

NMFS / Interagency Working Group Evaluation of CITES Criteria and Guidelines

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U.S. Department of Commerce
National Oceanic and Atmospheric Administration
National Marine Fisheries Service

NOAA Technical Memorandum NMFS-F/SPO-58
October 2002

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U.S. Department of Commerce

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This document is the result of several meetings and teleconferences of the NMFS / Interagency Working Group to evaluate CITES criteria and guidelines, held over a two-year period beginning in October 2000. The purposes were to evaluate existing CITES criteria and guidelines, to suggest improvements, and to evaluate the proposed improvements for a variety of marine and other taxa.

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Executive Summary

At present, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) criteria used to assess whether a population qualifies for inclusion in the CITES Appendices relate to (A) size of the population, (B) area of distribution of the population, and (C) declines in the size of the population. Numeric guidelines are provided as indicators of a small population (less than 5,000 individuals), a small subpopulation (less than 500 individuals), a restricted area of distribution for a population (less than 10,000 km²), a restricted area of distribution for a subpopulation (less than 500 km²), a high rate of decline (a decrease of 50% or more in total within 5 years or two generations whichever is longer or, for a small wild population, a decline of 20% or more in total within ten years or three generations whichever is longer), large fluctuations (population size or area of distribution varies widely, rapidly and frequently, with a variation greater than one order of magnitude), and a short-term fluctuation (one of two years or less).

The Working Group discussed several broad issues of relevance to the CITES criteria and guidelines. These included the importance of the historical *extent* of decline versus the recent *rate* of decline; the utility and validity of incorporating relative population productivity into decline criteria; the utility of absolute numbers for defining small populations or small areas; the appropriateness of generation times as time frames for examining declines; the importance of the magnitude and frequency of fluctuations as factors affecting risk of extinction; and the overall utility of numeric thresholds or guidelines.

The primary recommendations were:

- The historical extent of decline should be the ultimate criterion for triggering concern about the long-term viability of a species. The timeframe over which to examine the historical extent of decline should be as long as possible to enable a meaningful baseline to be chosen. Evaluations conducted by the Working Group suggested that declines down to 5-30% of historical or potential levels, with high productivity species being

nearer the 5% end, low productivity species being nearer the 30% end, and average productivity species being somewhere in the middle, worked reasonably well for exploited marine species.

- For the historical extent of decline, the time frame to examine should extend as far back into history as possible, regardless of the mean generation time of the species in question, in order to enable a meaningful baseline to be chosen (noting that the baseline need not be the highest historical point; depending on the species under consideration, the baseline may relate to some point in history, or to a *reasonable* or *potential* baseline given alterations to the environment that have affected current carrying capacity. Use of *reasonable* or *potential* baselines reflects, respectively, the reality that habitat changes have occurred in the past, and the possibility that such changes may be wholly or partially reversible. However, if the potential baseline is very small due to dramatic reductions in the carrying capacity of the habitat over time, it then becomes necessary to ask whether the current carrying capacity is adequate to ensure survival of the species).
- The "minimum sufficient data" is whatever usable quantitative, qualitative, or inferential data are available.
- Recent (5-10 year average) rates of decline should be considered in combination with historical extents of decline. It is suggested that threshold recent rates of decline be defined as the cumulative annual rate of decline that would drive a population down from its current level to the threshold extent of decline in the near future (e.g., 10 years).
- The extent and rate of decline of populations must be considered in light of various vulnerability factors that may be specific to one or a few taxonomic groups. Such factors may either increase the risk associated with a given decline, or mitigate it.

- The relative (e.g., percentage) historical extent of decline and the recent rate of decline should be preferred over absolute numbers as triggers for considering species for listing in CITES criteria. If absolute numbers of individuals or areas of distribution are to be used at all, they should be developed on a taxon-by-taxon basis.
- Development of generic thresholds based on percentage declines in relevant metrics should continue, recognizing that it may be necessary to have different numbers or ranges for different taxonomic groups.
- Although in some cases there may not be sufficient data to quantitatively evaluate the criteria recommended here, it will almost always be possible to use qualitative infor-

mation and analogies with other (related or cohabiting) species to develop an informed judgment about the likely status of a population with respect to the suggested criteria.

These recommendations were evaluated for the following taxonomic groups: demersal finfish, small pelagics, highly migratory species, sharks, deepwater species, salmonids, cold-water invertebrates, warm-water invertebrates, marine mammals, sea turtles, sturgeon and plants (trees and orchids).

The report also includes seven Appendices that augment the discussions and recommendations in the main body of the text. The most important of these is **Appendix I**, "Relationship Between Threshold Extent of Decline, Resilience and Productivity", which provides the justification for the 5-30% range suggested for historical extent of decline thresholds.

1. Summary of Terms of Reference

The following is extracted from a somewhat longer document compiled in September 2000, and approved by the National Marine Fisheries Service (NMFS) Science Board.

Background

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) adopted criteria and guidelines for listing species in CITES Appendices I and II at the 1994 meeting of the Conference of the Parties (COP9). The Parties also agreed at that time to review the criteria and guidelines and evaluate their applicability to various taxonomic groupings before COP12, scheduled to be held in the fall of 2002. The CITES review is currently underway, with various other groups (e.g., the United Nations Food and Agriculture Organization, FAO) contributing to the process by evaluating the criteria and guidelines for specific taxonomic groups. A joint meeting of the CITES Animals and Plants Committees will be [was] held to discuss progress on the review during 7-9 December 2000 at the National Conservation Training Center in Shepherdstown, West Virginia, USA. The NMFS / Interagency Criteria Working Group Chair, as well as other members of the WG, will [did] participate in the Shepherdstown meeting.

Role of NMFS / Interagency Working Group

The U.S. Fish and Wildlife Service (FWS) is the lead agency for the United States for CITES issues. The NMFS / Interagency CITES Criteria Working Group (the WG) will include staff from FWS and the Forest Service who will help formulate recommendations about CITES criteria and guidelines, primarily for those taxonomic groups for which NMFS is responsible, but also attempting to ensure the resulting criteria and guidelines can be applied to other taxonomic groups. The taxonomic groups, together with the experts assigned to them are:

- Demersal finfish (e.g., gadoids and flounders): Jay Orr, AFSC
- Small pelagic fishes (e.g., herring, sardines and mackerel): John Hunter, SWFSC
- Highly migratory species (e.g., tunas and

- swordfish): Pamela Mace, F/ST
- Low productivity finfish
 - most sharks: Nancy Kohler, NEFSC
 - deepwater species: Pamela Mace, F/ST
- Salmonids: Robert Kope, NWFSC
- Invertebrates (e.g., molluscs and crustaceans)
 - cold water: Robert Otto, AFSC
 - warm water: Margaret Miller, SEFSC & Andy Bruckner, F/PR
- Marine mammals: Tim Smith, NEFSC
- Sea turtles: Nancy Thompson, SEFSC
- Other - * Sturgeon: John Field FWS
 - * Plants: Julie Lyke FWS; Arthur Blundell USEPA
- CITES experts: Nancy Daves, F/PR; Sue Lieberman & John Field, FWS

* Sturgeons and plants were not included in the original terms of reference, but were included in workshop discussions and as case studies in this report, thanks to the participation and efforts of relevant experts.

The NMFS / Interagency WG will review existing CITES criteria and guidelines and other criteria proposed for similar purposes to determine their applicability to each of the taxonomic groupings above. Other proposed criteria/ guidelines include the World Conservation Union (IUCN) Red List criteria, Endangered Species Act (ESA) listing factors, American Fisheries Society (AFS) criteria proposed in *Fisheries* (Musick 1999), relevant sections of the report of the FAO CITES Criteria Technical Consultation held in June 2000, publications on population viability analysis (PVA), and other relevant scientific literature.

Specific questions to be addressed

At present, the CITES criteria used to assess whether a population qualifies for inclusion in the CITES Appendices relate to:

- A. Size of the population
- B. Area of distribution of the population
- C. Decline in the size of the population

The guidelines in Annex 5 of the CITES criteria

(CITES Resolution 9.24) suggest numeric values for a “small” population and subpopulation, a restricted area of distribution, and a rate of decline that might be cause for concern (see below for more details).

Some questions to initiate the WG discussions are [were]:

1. Which is more important, the rate of decline or the extent of decline?
2. Is it necessary to have a time frame for the extent of decline?
3. If so, is generation time an appropriate metric for the time frame?
4. Is it better to use absolute numbers for population size and area of distribution, or to use percentages of some historical and/or potential level?
5. Are different absolute numbers or percentages needed for different taxonomic groups?
6. Would it be better to use life history characteristics rather than taxonomic groupings (or both) to define appropriate absolute numbers or percentages?
7. Is it at all valuable to have numeric guidelines?

2. Background: CITES Resolution Conf. 9.24

CITES Resolution Conf. 9.24, "Criteria for Amendment of Appendices I and II", was adopted at the Ninth Meeting of the Conference of the Parties in Fort Lauderdale, USA in November 1994. The Resolution consists of a preamble and six annexes. Annex 1 concerns biological criteria for Appendix I; Annex 2 concerns criteria for the inclusion of species in Appendix II, with Annex 2a indicating listing criteria for species of concern, and Annex 2b covering the situation of look-alike and taxonomically related species; Annex 3 covers special cases such as listing of a species in more than one Appendix (split listings); Annex 4 outlines precautionary measures; Annex 5 contains definitions, notes and guidelines; and Annex 6 provides a format for proposals to amend the Appendices. Annexes 2b, 3, 4 and 6 are not addressed in this report, and will not be mentioned further.

Annex 1 lays out the conceptual basis for considering species for listing in CITES Appendix I. The conceptual basis includes four factors: (A) the wild population is small, (B) the wild population has a restricted area of distribution, (C) there has been a decline in the number of individuals in the wild, and (D) the species is likely to satisfy one of the previous three criteria within a period of five years. Each of criteria A, B and C also list additional relevant factors. Annex 2a is less specific, essentially stating that a species should be included in Appendix II if it is likely to meet at least one of the criteria in Annex 1 unless subject to strict regulation of international trade, or if harvest rates have exceeded sustainable levels over an extended period. There are no numeric thresholds or guidelines in Annexes 1 and 2a, but

both are prefaced with a statement that they must be read in conjunction with the definitions, notes and guidelines listed in Annex 5.

Annex 5 defines and discusses the terms: area of distribution, decline criteria, an extended period, fragmentation, generation time, large fluctuations, population, possibly extinct, subpopulations, and threatened with extinction. Numeric guidelines are provided as indicators of a restricted area of distribution for a population (less than 10,000 km²), a restricted area of distribution for a subpopulation (less than 500 km²), a high rate of decline (a decrease of 50% or more in total within 5 years or two generations whichever is longer or, for a small wild population, a decline of 20% or more in total within ten years or three generations whichever is longer), a small population (less than 5,000 individuals), a small subpopulation (less than 500 individuals), large fluctuations (population size or area of distribution varies widely, rapidly and frequently, with a variation greater than one order of magnitude), and a short-term fluctuation (one of two years or less). However, it is emphasized that each of these numbers are provided only as guidelines and not thresholds, and that there will be many cases where such numerical guidelines do not apply.

During initial meetings, the Working Group decided to focus primarily on Annex 5, rather than Annexes 1 and 2a, because modifications to the latter will depend on which modifications to Annex 5 are adopted. In addition, while some wordsmithing of Annexes 1 and 2a may be beneficial, it is hard to argue against them at the conceptual level.

3. Issues and Recommendations

The Working Group discussed several broad issues of relevance to the CITES criteria and guidelines. These included the importance of the historical *extent* of decline versus the recent *rate* of decline; the utility and validity of incorporating relative productivity into decline criteria; the utility of absolute numbers for defining small populations or small areas; the appropriateness of generation times as time frames for examining declines; the importance of the magnitude and frequency of fluctuations as factors affecting risk of extinction; and the overall utility of numeric thresholds or guidelines.

A. Decline

A decline is a reduction in some measure of, or proxy (surrogate) for, the abundance of a species; for example, numbers, biomass, or area of distribution. Decline can be considered in two fundamentally different ways: the overall long-term extent of decline or the recent average annual rate of decline. The Working Group asked the question, “which is more important, the recent rate of decline or the long-term extent of decline?”. It was concluded that the overall extent of decline will generally be more relevant to triggering concern about the long-term viability of a species than will the rate of decline; however, the recent rate of decline may also be important in certain situations such as in the case where a population has already experienced a pronounced extent of decline and is continuing to decline. Also, it was agreed that a given historical extent of decline and/or a given terminal rate of decline is more worrisome for a low productivity species than it is for a high productivity species.

Extent of decline

The historical *extent* of decline should be the ultimate criterion for considering a species for listing in the CITES Appendices. However, different baselines may be appropriate in different situations, and this needs to be evaluated on a case-by-case basis. Depending on the species under consideration, the baseline may relate to some point in history,¹ or

¹ Some experts argue that historical baselines are often impossible to estimate, yet when trying to determine whether a population is “small”, the question becomes “small relative to what”, and the mental gymnastics most experts would perform would involve comparing current

to a *reasonable* or *potential* baseline given alterations to the environment that have affected current carrying capacity. Use of *reasonable* or *potential* baselines reflects, respectively, the reality that habitat changes have occurred in the past, and the possibility that such changes may be wholly or partially reversible. However, if the potential baseline is very small due to dramatic reductions in the carrying capacity of the habitat over time, it then becomes necessary to ask whether the current carrying capacity is adequate to ensure survival of the species. The Working Group did not have time to fully address the issue of shifting baselines, but does consider this issue to be important and relevant (to what extent should or can changes in the baseline due to “development” or other factors be accepted, and expectations adjusted accordingly?).

Within reasonable limits, it is not necessary to have a time frame for the extent of decline.² However, the Working Group agreed that the threshold extent of decline should be a function of the resilience of a population. Unfortunately, resilience is not an operational concept. One possible proxy for resilience is population productivity,³ and the Working Group decided to adopt this as a tentative operational substitute for resilience. **Appendix I** of this document contains a more detailed discussion and brief literature review of the relationship between risk of extinction, resilience and productivity. Assuming productivity to be a reasonable substitute

population size or area of distribution to some estimate of, or informed guess about, historical size. Note that historical size can also be inferred.

² If the current population size is only a small fraction of its historical size, it should not matter whether this change took place over 3 generations or 10 generations. However, if the change took place over geological time, or over hundreds of generations, the species may have evolved adaptations to enhance survival at smaller population sizes.

³ Productivity is a complex function of fecundity, growth rates, age of maturity, generation time, and other relevant factors, commonly subsumed in a single parameter, r , the intrinsic rate of natural increase. Species with high productivity are said to be r -selected, whereas species with low productivity are said to be K -selected, where K is the symbol for environmental carrying capacity. Most species will fall between the

for resilience, for a very high productivity species (e.g., one with high fecundity and a rapid turnover of generations), consideration for listing in CITES Appendices might not be triggered until the species has declined to relatively low levels, while for a very low productivity species (e.g., one with low fecundity and a long period between generations), consideration for listing in CITES Appendices might be triggered at much higher levels of relative population size. The decline need not necessarily be continuing.

Rate of decline

The recent *rate* of decline may also be important in several respects: for example, when a population has already experienced a large extent of decline; as a surrogate for the extent of decline; or as a general indicator of the urgency of the need for remedial action. In addition, an unplanned rapid rate of decline might be indicative of a rapid change in environment, or a disease attack, or competition with an invasive species. The rate of decline is only relevant if a decline is still occurring (i.e., the population has exhibited a declining trend over the last several years) or has the potential to resume. A given recent (5-10 year) rate of decline is more problematic for species with higher historical extents of decline and for species with lower productivity. Thus, the (recent) rate of decline generally needs to be combined with the (historical) extent of decline and species productivity³. If it is not considered in conjunction with the historical extent of decline, the recent rate of decline may be largely irrelevant, except when it is extremely high. The importance of considering the historical extent of decline and the recent rate of decline in conjunction with one another is illustrated using a hypothetical example in **Appendix II** of this document. A potentially-useful method for combining the historical extent of decline, the recent rate of decline, and productivity, where species productivity is indexed only by generation time, is depicted graphically in Figures 1 and 2 of **Appendix III** of this document and discussed in Section 4B. The figures in **Appendix III** incorporate the concepts that species with lower productivity should have a higher threshold extent of decline, and a lower threshold rate of decline at any given extent of decline.

Recommendations: The historical extent of decline should be the ultimate criterion for triggering concern about the long-term viability

of a species. The timeframe over which to examine the historical extent of decline should be as long as possible to enable a meaningful baseline to be chosen. Recent (5-10 year average) rates of decline should be considered in combination with historical extents of decline.

B. Absolute Numbers, Biomass or Area

The Working Group considered and discussed at length the absolute numbers suggested as guidelines (not thresholds) in Annex 5; viz, 5,000 individuals for a population and 500 individuals for a subpopulation; and an area of distribution of 10,000 km² for a population or 500 km² for a subpopulation. The Working Group found it much easier to come up with examples for which the numbers would be either far too high, or far too low, or simply not meaningful (an example of the latter is an area of distribution for a highly migratory species), than it was to find examples where the numbers might be reasonable. For example, the number of 5,000 individuals was believed to be far too low for most commercially exploited finfish and invertebrates, and too high for large whales. Comparisons of large whales and small invertebrates (e.g., shrimp) suggested that, in fact, biomass may be a better metric than numbers, because it integrates body size. However, even at a taxonomic group as low as species, it does not make sense to apply the same absolute numbers across all populations within the species. Consider, for example, Atlantic cod (*Gadus morhua*). There are numerous separate stocks of Atlantic cod throughout the North Atlantic, with more than 25 stocks being of commercial importance. The range in maximum observed recent biomass levels is at least 2,000 to 1.6 million metric tons, and the degree of resilience of the stocks also appears to differ substantially between geographic areas. It was agreed that there were so many exceptions to the validity of these numbers that they were not useful, even as “a guideline, not a threshold”. Meaningful numbers may be able to be developed on a taxon-by-taxon basis for numbers of individuals (or related metrics such as biomass or percent cover) but not necessarily for area of distribution.

Recommendation: The Working Group recommends that the relative (e.g., percentage) historical extent of decline and the recent rate of decline should be preferred over absolute numbers as triggers for considering species for

extremes of the r-K spectrum. For more details, see **Appendix I** of this document.

listing in CITES criteria. If absolute numbers of individuals or areas of distribution are to be used at all, they should be developed on a taxon-by-taxon basis.

C. Generation Times⁴

There are two related ways in which generation times appear to have been interpreted and used in the CITES criteria and guidelines. First, generation times are explicitly used in the guidelines as the basis for time frames over which to consider rates of change in population sizes. Second, generation times are implicitly assumed to be inversely related to productivity and/or resilience (i.e., positively correlated with relative vulnerability). Long mean generation time is often considered to be correlated with relatively low productivity. However, the correlation is more likely to apply within taxonomic groups and may not hold when making comparisons across taxonomic groupings, due to the large diversity of combinations of life history characteristics.

The Working Group agreed that mean generation times are appropriate as ingredients of population rebuilding or recovery plans (in that long-lived, low productivity species can generally be expected to recover or rebuild at a slower rate than short-lived, high productivity species), but could not agree on their applicability as time frames for decline criteria. Oldfield *et al.* (1998) contend that "...it is very important to be clear that extinction times scale with a species' generation time, and not with absolute time. All other things being the same, a long-lived species takes longer to go extinct than a short-lived species. If one considers a time window that is short relative to the lifetime of the species, one might miss the fact that it is in trouble, even if it is heading inexorably to extinction. To take a simple example, if one counts the numbers of a short-lived species with four generations per year and declining at 50

⁴ There are several ways of measuring mean generation time. Here we assume the Resolution Conf. 9.24 Annex 5 definition of generation time; viz. "Generation is measured as the average age of parents in the population; except in the case of species that breed only once a lifetime, this will always be longer than the age at maturity." However, we note that this measure can vary considerably depending on the extent of exploitation of a population. It is suggested that generation time be measured or inferred as the average age of parents that would exist in an unexploited population.

percent per generation, the population will have declined to 6.25 percent of the starting number after one year. Conversely, a species with a long generation time (say 100 years), but also declining at 50 percent per generation will exhibit almost no detectable decline as measured over a one-year period."

In fact, in order for a long-lived species and a short-lived species to be declining at a rate of 50% per generation, it is not possible to have "all other things being the same". Continuing declines of 50% per generation would mean that the rate of harvest was above replacement levels for both species, but substantially lower for the long-lived species than for the short-lived species. It would mean that the people conducting the harvest were aware of the relative generation times and adjusted their harvest rates accordingly (this may be a sound idea, but people's needs or wants usually scale with absolute time and not with the generation time of the species they are exploiting). Regardless, scaling harvest rates to generation times does not fit with the assumption of "all other things being the same". Oldfield *et al.*'s argument is almost circular. It's like saying, "if decline is measured on a per generation basis, then it's important to consider generation time".

In fact, if all else is equal⁵, and in particular if the annual exploitation (harvest) rates (percentage of the population removed each year) were identical for the two species, the long-lived species would likely decline more rapidly than the short-lived species. (For empirical examples of the rapidity of depletion of long-lived marine species, see the case studies for deepwater species in Section 5). The reason is that the high productivity species will generally replenish itself faster. For the same reason, a large⁶ decline from one year to the next would be more worrisome for a low productivity species than for a high productivity species. Thus, for any specific extent rate of decline, one should become concerned sooner and act more quickly, the lower the productivity of the population. Based on these examples, and contrary to common usage, the time frame for decline criteria should actually be inversely related to mean generation time, as it is in the examples given in **Appendix III** of this document.

⁵ "If all else is equal" means, for example, that the population biomass and exploitation rates are identical in the two cases.

⁶ e.g. 20%.

An alternative viewpoint is that use of generation times as time horizons forces one to look further back in time for long-lived species, which may have exhibited a protracted, gradual decline, or may have been relatively stable for the past 10-20 years but experienced a substantial decline 2-3 generations ago. The basic premise is that it is not sufficient to look back (or forward) only 10 years or so for long-lived species. The flip side of this premise is that it must therefore be reasonable to look back (or forward) only 10 years or so for short-lived species. However, there are equally valid reasons for looking far back into history for highly-productive, short generation species, which often exhibit pronounced fluctuations over time (e.g., Pacific sardine and anchovy).

In fact, it may be difficult to interpret the current situation outside of the context of the magnitude and frequency of historical fluctuations. **Appendix IV** of this document describes two hypothetical examples that illustrate this point. In the first, a highly-productive, short generation species has declined substantially over the past three generations, but the most recent peak in population size is unusually high compared to the rest of the 100-year recorded history of the stock, and the stock is currently well above historical average levels. In the second, there has been a relatively slow decline over the past three generations, but the most recent population peak was well below the historical average, and the stock is currently lower than the lowest stock size recorded over the previous 100 years. In both cases, recent trends in population size could easily be misinterpreted if not considered in the context of the historical data.

Discussion of the utility of generation times also evoked discussion of the “minimum sufficient data” or the “minimum sufficient time horizon” for estimating or inferring a decline. It was quickly concluded that it would be dangerous to attempt to define “minimum sufficient”, and that the best approach would be to equate “minimum sufficient data” with “all available usable data” and encourage analysts to always take a historical perspective, to the extent possible.

Although threshold rates of decline of the form “x% in y years or z generations, whichever is longer” are currently a key feature of both the CITES and IUCN guidelines, the Working Group believes that alternative constructs may be more valid and more widely applicable. In particular, for the historical

extent of decline, the time frame to examine should extend as far back into history as possible, regardless of the mean generation time of the species in question, in order to enable a meaningful baseline to be chosen. It should be noted that the baseline would not necessarily be the highest point in the historical series (see Section 3A). For the recent rate of decline, a literal interpretation of the above construct could be taken to imply that one should wait for y years or z generations, or at least until the x% cumulative decline has accrued, before taking action (see example in **Appendix V**).

One way of simultaneously satisfying both sides of the debate about the utility of generation times is to express the rate of decline as an average annual rate and develop guidelines that embody the points of consensus that (a) a given average annual rate of decline should be more worrisome for a species that has already experienced a large extent of decline, (b) a given average annual rate of decline should be more worrisome for a low productivity species than for a high productivity species, and (c) recent rates of decline can be extrapolated forward in time to infer likely future reductions in population size, rather than waiting to see if they actually happen (**Appendix V**). It should also be noted that, although it may be reasonable to index productivity by some function of the inverse of generation time, it may be preferable to use other metrics that relate more directly to productivity, even though these may require more data to calculate; for example, the intrinsic rate of natural increase.³

Recommendations: For the historical extent of decline, the time frame to examine should extend as far back into history as possible, regardless of the mean generation time of the species in question, in order to enable a meaningful baseline to be chosen (noting that the baseline need not be the highest historical point). The “minimum sufficient data” is whatever usable quantitative, qualitative, or inferential data are available. For the recent rate of decline, generation time should not be used in a way that may preclude a species from consideration because it does not have a sufficiently long time series of data (**Appendix V**). Thus, generation time should not be used explicitly to set a minimum time frame of data to examine, but rather should be used implicitly to set threshold average annual rates of decline that are related to species productivity (the latter is

true only if it can be assumed that productivity is some inverse function of generation time). This reflects the concept that a given average annual rate of decline should be considered more problematic for a species with low productivity than for a species with high productivity. It is suggested that the time frame over which to consider average annual rates of decline should be about 5-10 years.

It should be noted that, by recommending that timeframes for assessing declines be as long as possible regardless of generation time, the Working Group is not attempting to negate or detract from the argument that it is necessary (to the extent possible) to look far back into the past for long-lived species; it is simply adding in the same requirement for short-lived species. If one considers a time window that is only three generations long for a short-lived species, one might miss the fact that it is in trouble, even if it is heading inexorably to extinction. Or, one might conclude that it is heading inexorably to extinction when it is simply exhibiting the same type of behavior it has exhibited for the last century or more (see the examples in **Appendix IV** of this document).

D. Fluctuations

The Working Group did not fully address the issue of fluctuations. The CITES criteria and guidelines imply that populations with large and rapid fluctuations in numbers of individuals or numbers of subpopulations are at greater risk of extinction than those that do not fluctuate as much. However, if a subpopulation is a semi-isolated reproductive unit, then rapid fluctuations in the number of subpopulations is unlikely (rapid declines perhaps, but not rapid increases). Large, high-frequency fluctuations in numbers and/or biomass are characteristic of highly-productive (r-selected) species that are able to take quick advantage of suitable conditions for reproduction. In theory, because of their higher variability, there may also be a greater risk that population numbers will fluctuate to dangerously low levels, even in the absence of continued exploitation. In addition, species with high turnover of generations tend to have relatively few mature age classes, which means that recruitment failure is more critical. On the other hand, some long-lived species may not have large and rapid fluctuations, but nevertheless have extremely high variability in recruitment that makes them vulnerable to overexploitation. For example,

there are several long-lived marine species with sporadic exceptionally large year classes with most other year classes being insufficient for population replacement; Atlantic redfish, Pacific bocaccio and Atlantic ocean quahog are examples.

The Working Group found it hard to envision a situation where large and rapid fluctuations would be a primary impetus for listing. In addition, CITES' definition of a "large fluctuation" as one of an order of magnitude or more does not take account of the different implications of order of magnitude fluctuations for an average population of 1,000 individuals compared to a population of a billion individuals. CITES' definition of a "rapid" fluctuation as one with a period of two years or less would seem to apply to very few species. More work and thought on this issue is needed.

Recommendation: It would make more sense to define a "large fluctuation" as a percentage of average numbers; for example, "a large fluctuation is one of plus or minus 50% of average numbers"; however, this may not fully alleviate the potential problem that a decline of 50% of average numbers will have greater implications for a small population compared to a larger one. In addition, the definition of "short term fluctuation" should be more flexible (i.e., longer); perhaps "5 years or less" rather than a fixed time of 2 years. However, more work and thought on the utility of fluctuations as an impetus for a CITES listing is needed.

E. The Utility of Numeric Guidelines

Despite the difficulty of developing numeric guidelines that are broadly applicable across a wide diversity of taxonomic groups and life history strategies, the Working Group felt there was merit in continuing the attempt to develop generic thresholds (based on percentage declines in relevant metrics, but not necessarily absolute numbers of individuals or absolute areas) that would trigger concern about the long-term viability of a species. Based on experience in fisheries, the development of numeric reference points has been instrumental in reversing overexploitation in many fisheries, even though specification and estimation of such reference points often stretches the limits of available data and methodologies. The guidelines and thresholds suggested in the next section appear to work reasonably well for the taxa considered in this report,

taxa considered in this report, although further work is needed to determine their applicability to an even broader array of taxa. It may well be that somewhat different numeric guidelines are needed for certain other taxonomic groups. Regardless of the specificity of any numeric guidelines developed, there is no escaping the need for informed and reasoned judgment as the ultimate criterion for considering a species for listing in the CITES Appendices.

Recommendation: Development of generic thresholds based on percentage declines in relevant metrics should continue, recognizing that it may be necessary to have different numbers or ranges for different taxonomic groups. In cases where no quantitative data exist, qualitative information and analogies with other (related or cohabiting) species should be used to develop an informed judgment about the likely status of a population with respect to the suggested criteria.

4. Proposal for Revised CITES Guidelines

Determination of CITES classifications must be based on the status of populations and their associated habitat. The pertinent conditions may vary across both taxa and habitat. The Working Group attempted to develop generic conceptual biological criteria, and generic numeric guidelines, that would trigger concern about the long-term viability of a species. It was concluded that quantitative, qualitative and inferential aspects of population status suitable for making CITES decisions that should apply to all taxa are (A) the historical extent of decline of a population or subpopulation, and (B) the recent rate of decline of that population or subpopulation. These are insufficient in themselves, however, and must be considered in the light of several (C) relevant modifying factors specific to individual situations. The historical extent of decline, the recent rate of decline, and the relevant modifying factors all need to be considered in conjunction with one another, not as a hierarchy. Of course, the Working Group recognized that the biological and other factors that “trigger concern about the long-term viability of a species” are only one of the considerations that will influence whether or not a species is or should be proposed for a CITES listing (other important considerations include the effects of international trade or trade demand and the existence and effectiveness of domestic or regional management plans).

A. Historical Extent of Decline

The extent of decline in populations should be considered relative to some appropriate baseline. However, different baselines may be appropriate in different situations. Depending on the species under consideration, the baseline may relate to some point in history¹, or to a *reasonable* or *potential* baseline given alterations to the environment that have affected current carrying capacity. Use of *reasonable* or *potential* baselines reflects, respectively, the reality that habitat changes have occurred in the past, and the possibility that such changes may be wholly or partially reversible. However, if the potential baseline is very small due to dramatic reductions in the carrying capacity of the habitat over time, it then becomes necessary to ask whether the current carrying capacity is adequate to ensure survival of the

species. Within reasonable limits, it is not necessary to restrict the time frame over which to examine the historical extent of decline. The entire known or inferred history of the species should be examined to enable a meaningful baseline to be chosen.²

Populations can be measured in many different ways, and appropriate units of measurement vary across taxa. For example, for some taxa, it may not make sense to refer to numbers of individuals (e.g., clonal organisms). The Working Group identified the following metrics as potentially relevant for measuring or indexing the extent of decline.

- numbers (of individual organisms in a population or subpopulation)
- biomass (total weight of a population or subpopulation)
- area inhabited (area of distribution)
- migratory range (for highly migratory species)
- percentage coverage (for sessile species)
- relative spawning per recruit (see **Appendix I**)
- numbers or biomass of new recruits (recruitment)

It may be appropriate to use more than one of these metrics for a given taxonomic group, but it is highly unlikely that all of these metrics are valid or useful for all taxonomic groups. The component of the population that should be considered is the life history stage that is most relevant to measure; except when considering recruitment, this will generally be the mature component.

Larger historical declines in one or more of the above metrics would suggest greater risk of extinction, and hence would trigger greater concern for consideration for a CITES listing. It is also likely that a given extent of decline may be considered more worrisome for a low productivity species than it is for a high productivity species, due to the likelihood that a high productivity species is probably able to rebound faster (see **Appendix I** of this document for a discussion of the relationship between risk of extinction, resilience and productivity). Therefore, the threshold extent of decline should be a function of the productivity of the species. The decline need not necessarily be continuing.

For a very high productivity species (e.g., one with high fecundity and a rapid turnover of generations), consideration for listing in CITES Appendices might not be triggered until the species has declined to relatively low levels; for example, about 5% of the baseline. For a very low productivity species (e.g., one with low fecundity and a long period between generations), consideration for listing in CITES Appendices might be triggered at much higher levels of relative population size; for example, a decline to about 30% of the baseline level. If productivity is indexed by generation time alone, then for exploited marine species the extreme of 5% of the baseline might be roughly applicable for species with mean generation times of 3-5 years or less (e.g., some squids and small pelagic fishes), while the extreme of 30% of the baseline might be applicable for species with mean generation times of 30 years or more (e.g., some sharks and deepwater species). For species between these two extremes, the threshold percentage decline should be interpolated. Here and in **Appendix III** of this document, an inverse function of generation time is used as a proxy for productivity because compared to most demographic variables it is relatively easy to estimate or infer. However, there are other metrics that may be superior indices of productivity; for example, intrinsic rates of natural increase calculated from life tables or surplus production models.

Support from the peer-reviewed fisheries scientific literature for the applicability of the 5-30% range and the role of productivity is summarized and discussed in **Appendix I** of this document. Most of the studies reviewed in Appendix I of this document were based on consideration of either population biomass or spawning biomass per recruit. However, the Working Group believed that the same or similar decline criteria could be applied for each of the metrics in the bulleted list above. Evaluations conducted by the Working Group suggested that the 5-30% range, with high productivity species being nearer the 5% end, low productivity species being nearer the 30% end, and average productivity species being somewhere in the middle, worked reasonably well for exploited marine species. Other ranges may be more appropriate for other taxonomic groups. For example, in a taxonomic group in which all species can be considered low productivity or “K-selected”, a narrower range excluding the lower end of the spectrum may be more appropriate.

B. Recent Rate of Decline

The *rate* of decline may also be important in several respects: for example, when a population has already experienced a large extent of decline; as a surrogate for the extent of decline; or as a general indicator of the urgency of the need for remedial action. In addition, an unplanned rapid rate of decline might be indicative of a rapid change in environment, or a disease attack, or competition with an invasive species. The rate of decline is only relevant if a decline is still occurring (i.e., the population has exhibited a declining trend over the last several years) or has the potential to resume. A given recent (5-10 year) rate of decline should be considered more problematic for species with higher historical extents of decline and for species with lower productivity. Thus, the (recent) rate of decline generally needs to be combined with the (historical) extent of decline and species productivity.

One straightforward, simple and coherent way of defining threshold average annual rates of decline that take account of the historical extent of decline and species productivity is to calculate the cumulative annual rate of decline that would drive a population down from its current level to the threshold extent of decline in the near future. The period used to represent “near future” needs to be sufficiently long that a decline can be detected, but not so long that the status of the population is likely to change dramatically. Here, it is assumed that a period of 10 years is a reasonable indicator of near future. The following text table summarizes the calculations for cumulative 10-year rates of decline (and corresponding average annual rates of decline) that will drive a population from its current level to the threshold extent of decline in 10 years. Values of % baseline greater than 50% should rarely cause concern for marine species, except perhaps in the case of small or endemic populations (see Section C), unless recent average annual rates of decline have been extremely high. Values of threshold average annual rates of decline are set to zero once the population is at or below the extent of decline threshold because once a population has fallen to such critical levels, a decline need not necessarily still be occurring. In the table, low productivity is equated with an extent of decline threshold of 30% of the baseline, medium productivity with a threshold of 20% of the baseline, and high productivity with a threshold of 5% of the baseline. For example, the

Current population as % unexploited	Productivity		
	low	medium	high
100%	70% (11.3%)	80% (14.9%)	95% (25.9%)
90%	67% (10.4%)	78% (14.0%)	94% (25.1%)
80%	63% (9.3%)	75% (12.9%)	94% (24.2%)
70%	57% (8.1%)	71% (11.8%)	93% (23.2%)
60%	50% (6.7%)	67% (10.4%)	92% (22.0%)
50%	40% (5.0%)	60% (8.8%)	90% (20.6%)
40%	25% (2.8%)	50% (6.7%)	88% (18.8%)
30%	0%	33% (4.0%)	83% (16.4%)
20%	0%	0%	75% (12.9%)
15%	0%	0%	67% (10.4%)
10%	0%	0%	50% (6.7%)
5%	0%	0%	0%

cumulative 10-year rate annual rate of decline that would reduce a high productivity population from 50% of its baseline to the threshold extent of decline of 5% is 90%, and the corresponding average annual rate of decline is 20.6% per year. The corresponding annual rate for a low productivity population would be only 5.0% per year. Note that the annual rates must be cumulated over time⁷ and therefore they cannot simply be multiplied by a number of years to calculate the total cumulative decline over those years.

⁷ The formula for cumulating annual rates over time is $F = (1-AR)^t$, where F is the fraction of the starting number remaining after t years, and AR is the annual rate of decline expressed as a proportion. This is equivalent to taking a starting number, multiplying it by the annual rate of decline, subtracting to determine the number remaining, then multiplying this number by the annual rate of decline, subtracting from the previous number, and continuing the process for the desired number of years. The number remaining after the desired number of years divided by the starting number is the fraction remaining (F). Subtracting F from 1 and multiplying the result by 100% gives the total percentage decline.

An alternative but related method for combining historical extent of decline, recent rate of decline and productivity (as indexed by an inverse function of generation time) is presented in **Appendix III** of this document. Figure 1 of **Appendix III** of this document suggests that, regardless of the recent rate of decline, for generation times of 10 years or less, concern for a population should be triggered when biomass falls below 5% B_0 ; for generation times of 30 years or more, concern for a population should be triggered when biomass falls below 30% B_0 ; and for generation times between these two extremes, the threshold extent of decline should be interpolated. Figure 2 of **Appendix III** depicts the threshold average annual rate of decline (expressed as a multiplier of the inverse of the natural logarithm of generation time) for various levels of historical extent of decline. Use of this graph would imply that the threshold rate of decline for a species with a mean generation time of three years would be an average recent rate of decline of about 13.7% per annum if it

had previously been reduced to about 50% of its baseline level, or about 5.5% per annum if reduced to 20% of its baseline level. The corresponding thresholds for a species with a mean generation time of 10 years would be about 6.5% per annum and 2.6% per annum. For a species with a mean generation time of 30 years, the corresponding thresholds would be about 4.4% per annum and 1.8% per annum. Although these annual rates of decline are more or less in line with the rates in the above text table (depending on the relationship assumed between generation time and high, medium and low productivity), and can easily be mixed and matched with several of the many decline rates currently used by IUCN, they are somewhat arbitrary and an alternative (but not too dissimilar) range of rates could be defended equally well. The method presented here is straightforward, eliminates artificial and arbitrary discontinuities in defining numeric guidelines, and allows use of variables other than generation times to index productivity.

The significance of both the extent and rate of decline should be evaluated using whatever data are available over as long a time period as possible. While longer time series are better, there is no basis for establishing either maximum or minimum lengths of time series. Further, while quantitative levels of population condition are preferable, qualitative information can be used where necessary, and even bounds on the likely degree of change inferred from such data can be useful. This is particularly relevant for species for which good quantitative data do not exist for periods prior to exploitation. Use of all available data, including qualitative data and inference, facilitates determination of appropriate reference levels to use as a baseline, as well as determination of expected rates of change of the population (as illustrated in **Appendix IV** of this document). In addition, examining as long a time series as possible may allow identification of causes of change; for example, changes in habitat availability or in other sources of mortality. Such changes may or may not be reversible, but need to be examined for their effect on risk of extinction, so that potential mitigating measures can be implemented.

C. Relevant Modifying Factors

The extent and rate of decline of populations must be considered in light of vulnerability factors that may be specific to one or a few taxonomic

groups. Vulnerability can be defined as the susceptibility of a species to over-exploitation or, conversely, the inverse of the capacity of a species to recover from overexploitation (i.e., the inverse of resilience). Vulnerability must be taken into account when evaluating a species against the CITES criteria. The Working Group believes that, in general, low productivity species are more vulnerable or susceptible to overexploitation than are high productivity species. Low productivity species (historically often referred to as “K-selected” species) are those that have one or a combination of long generation times, slow growth rates, or low fecundity; high productivity species (historically often referred to as “r-selected” species) have the reverse characteristics.³ K-selected species generally have lower resilience to exploitation because they have less potential to rebound as quickly. On the other hand, high variability may increase the risk that a population will fluctuate to dangerously low levels.

There are also several additional risk factors that cannot easily be incorporated into a generic risk evaluation because they are applicable only to particular taxonomic groups, certain life history strategies, or in specific instances. Such factors include population density (especially for sessile or semi-sessile species that rely on air, water, or other vectors to disperse and mix spawning products), migration, degree of endemism, habitat specificity, dietary specificity, and symbiotic relationships. These and other factors may increase (or decrease) the risks to the species, and may therefore necessitate appropriate modification to any thresholds suggested in these guidelines. For example, for a factor that increases vulnerability, the historical extent of decline and/or the recent rate of decline that would qualify a species for inclusion in the Appendices could be respectively higher and lower than it would in the absence of these factors. An alternative way of viewing vulnerability factors is that they increase the risk for a specific extent or rate of decline. For this reason, these additional risk factors are classified as modifiers of the decline criteria. The wide range of potential taxon- or case-specific vulnerability factors supports the contention that there is no escaping the need to consider each species on a case-by-case basis.

A non-exhaustive list of vulnerability factors (that would increase concern) and mitigating factors (that would decrease concern) that may be relevant to particular taxonomic groups is presented below. Those marked with an arrowhead are explained or

discussed further in **Appendix VI** of this document.

(i) Vulnerability factors that would increase concern

- Life history characteristics (e.g., low fecundity, slow growth rates, high age at first maturity, long generation time): these factors are discussed at length in the text and in **Appendix I** of this document
- Low absolute numbers or biomass
- Restricted area of distribution
- Selectivity of removals
- Distorted age, size or stage structure of a population
- Social structure, including sex ratio
- Low population density (especially for sessile or semi-sessile species)
- Specialized niche requirements (e.g., diet and habitat)
- Species associations such as symbiosis and other forms of co-dependency
- Strong aggregating behavior (e.g., schooling)
- Extensive migrations

- Secondary ecosystem-based effects
- Uncertainty
- Fragmentation
- Reduced genetic diversity
- Severe habitat loss
- Degree of endemism
- Existence of disease
- Existence of invasive species
- Existence of rapid environmental change (e.g., unfavorable climate regime shifts)

(ii) Mitigating factors that would decrease concern

- Life history characteristics (e.g., high fecundity, rapid growth rates, low age at first maturity, short generation time): these factors are discussed at length in the text and in **Appendix I** of this document
- High absolute numbers or biomass
- Existence of natural refugia
- Adaptations to small population size
- Selectivity of removals

5. Postscript: Developments since Initial Drafting of this Report

Subsequent to initial drafting of this report, several international meetings have been held. These include a meeting of CITES' own Criteria Working Group (CWG) held in Segenza, Spain on 21-23 May 2001; an FAO review convened from 11-15 June in Rome, Italy; and an FAO Technical Consultation on CITES criteria held in Windhoek, Namibia on 22-25 October, 2001 (FAO 2001). Material from the initial draft of this report was considered in all three meetings and many of the ideas contained herein were adopted or modified. Some of the key concepts were also presented and discussed at a joint Animal and Plants Committee meeting hosted by the United States in Shepherdstown, WV on 7-9 December 2000. Preparations are now underway for further work on the listing criteria to be conducted at CITES' 12th Conference of the Parties to be held in Santiago, Chile on 3-15 November 2002.

Evaluations conducted by the NMFS / Inter-agency Working Group concluded that a range for historical extent of decline to 5-30% of the baseline, depending on species productivity (with high productivity species being nearer the 5% end, low productivity species being nearer the 30% end, and average productivity species being somewhere in the middle), was likely to be appropriate for a wide range of taxa, including marine taxa. However, FAO (2001a, b) subsequently concluded that for the majority of exploited fish and invertebrates in marine and large freshwater bodies, a narrower range of 5-20% would be more appropriate. Although some species may fall outside this range (e.g., a percentage decline to less than 5% may be appropriate for some species such as the clupeids, sardines and anchovy, that are characterized by exceptionally high numbers and biomass; while a percentage decline greater than 20% may be appropriate for some species such as certain sharks and deepwater species, that are characterized by extremely low productivity), these were considered to be the exception rather than the rule. In particular, the range of 20-30% was believed to be overly conservative for all but a few commercially-exploited marine species.

FAO (2001b) further suggested that commercially-exploited marine species with high productivity would generally fall in the range of 5-10% of the historical baseline, species with medium produc-

tivity would be in the range 10-15%, and species with low productivity would be in the range 15-20%. FAO recommended that these percent historical extent of decline ranges be used for considering species for listing in Appendix I, and that 5% be added to each of the ranges for consideration for listing in Appendix II. In addition, it was recommended that consideration for an Appendix II listing would be triggered if the recent rate of decline has been sufficient to bring population size down from its current level to the threshold Appendix I level within 10 years.

FAO (2001a) also proposed a revision of Mulsick's (1999) table identifying ranges of numeric values of life history parameters associated with different levels of productivity (high, medium and low) and agreed with this report in rejecting the use of generation times as time frames over which to assess declines. CITES' own CWG initially omitted mention of generation times, but has now reinstated it based on comments received that disagreed with the change. This is at least partly because the current CITES proposal does not explicitly link rate of decline criteria with extent of decline criteria.

Both FAO and CITES' own CWG embraced the concepts of "modifying factors" as outlined in Section 4C of this report, although in both cases, they merged the lists of "vulnerability" and "mitigating" factors and rearranged and wordsmithed some of the items.

Another development that has taken place since the initial drafting of this report is the realization that most listings under CITES (and, incidentally, the U.S. Endangered Species Act; ESA) to date have been based on absolute numbers or restricted area of distribution, rather than decline criteria. Thus, even though the Working Group recommended

"...that the relative (e.g., percentage) historical extent of decline and the recent rate of decline should be preferred over absolute numbers as triggers for considering species for listing in CITES criteria... and if absolute numbers of individuals or areas of distribution are to be used at all, they should be developed on a taxon-by-taxon basis", further development of numeric

guidelines for these two factors is probably warranted. It is suggested that for absolute numbers, a range of 1,000 to 100,000 would cover most of the taxa likely to be considered for CITES listings. The lower end of the range would likely be appropriate to large vertebrates, while the upper end of the range might be appropriate to many smaller species. (In fact, it may even be preferable to use biomass, rather than numbers, since biomass integrates body size effects to some extent). Groups of experts on individual taxonomic groups should be challenged to choose narrower ranges within this spectrum that would likely cover most species within the taxonomic group. Values outside of the range could be used, but these would need to be justified on a case-by-case basis.

Regarding area of distribution, it is much more difficult to come up with a range of absolute numbers that would be at all meaningful, and it seems that any such range would still need to be extremely wide, even within a fairly low-level taxonomic grouping. Thus, it does not seem possible to recommend meaningful absolute areas of distribution at present and consideration of historical percent changes in area of distribution should be preferred.

The last issue to be mentioned in this section is that of Population Viability Analysis (PVA). This topic was not discussed in depth by the current Working Group or at any of the meetings listed in the first paragraph of this section, even though many participants were familiar with the use of this class of techniques for estimating extinction probabilities. Although PVA-based numeric guidelines (such as “the probability of extinction within 100 years is greater than X%”) could be added to the current criteria and guidelines, it is also necessary to recognize that PVA and extent of decline thresholds

represent fundamentally different philosophies in the approach to extinction risk. PVAs essentially try to model population dynamics at very low population sizes where it is known that several key biological features of the population may change dramatically (i.e., compensatory effects come into play). Although we know of the existence of Allee effects and other phenomena associated with small population size, we generally know little about the details of their dynamics, and experiments to learn more are potentially dangerous. Use of extent of decline thresholds recognizes this fact, and instead defines an upper bound on a “worry zone” that it would be best not to venture into. While both approaches may be incorporated into guidelines, it is unlikely that it will be meaningful to attempt to define numeric values that map the two approaches.

References for Sections 1-5

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6. Evaluations by Taxonomic Groups

The Working Group emphasized that the evaluations and discussions presented in this section do not necessarily imply endorsement by the Group of the need to include or exclude species from the CITES Appendices. Such decisions must be made through a separate process that also includes consideration of the implications of actual or potential international trade and the efficacy of current domestic or regional management plans.

A. MARINE FISHERIES

DEMERSAL FINFISH

Three major taxonomic groups make up the targets of demersal finfisheries of the United States and are subject to international trade: the flatfishes (pleuronectiforms), rockfishes (scorpaenids), and codfishes (gadoids). Nearly all members of the three groups are exclusively marine in habitat. Although some species are heavily overfished and some local populations are threatened with extirpation (e.g., Puget Sound rockfishes and gadoids and U.S. populations of Atlantic halibut), only one of the species throughout its range is considered vulnerable (cowcod, *Sebastes levis*) or threatened with extinction (Musick *et al.* 2000).

Flatfishes

The flatfishes demonstrate moderate productivity. Most of the approximately 88 species in U.S. waters exhibit high fecundity ($>10^4$) with early maturity (2-3 years) and a moderate life span (>10 years). Important exceptions are the halibuts (*Hippoglossus* spp.), which also exhibit high fecundity but mature at a later age (>10 years). All species are benthic broadcast spawners and generally are found over gravel, sand, or mud substrates.

Historical extent of decline

In 1999, fisheries management plans for Alaska called for the exploitation of flatfish stocks at a rate of $F_{40\%}$ with an overfishing limit set at $F_{30\%}$ or $F_{35\%}$ (NMFS 1999). With generation times in the range of 5-10 years, the 5% trigger appears most appropriate.

The later maturation of halibut species suggests their greater susceptibility to extinction risks and a more conservative value for extent of decline may be warranted.

Recent rate of decline

Flatfish stocks in Alaska exhibit strong changes in biomass over 5-10 years, ranging from 35 to 65% declines over the 10 years from the mid 1980s to mid 1990s. Some species decreased 15-20% in biomass estimates within two years, after the total biomass had been reduced below 50%, with subsequent increases in following years (McCaughran 1995, McConnaughey 1995). These Alaska stocks are all considered under-utilized, primarily because of management of halibut bycatch.

Rockfishes

The rockfishes demonstrate low to very low productivity. All of the approximately 70 species in U.S. waters exhibit high fecundity ($>10^4$) with late maturity (5-10 years) and a long life span (>10 -30 years). All species of *Sebastes* are viviparous, giving birth to live young; *Sebastolobus* species are oviparous, releasing eggs as a balloon-like mass. Rockfishes are found in a wide variety of habitats, from the intertidal to the continental slope at depths to 1500 meters. They are typically closely associated with the bottom or form semipelagic aggregations several meters above the substrate (Love *et al.* 2002).

Historical extent of decline

In 1999, fisheries management plans for Alaska called for exploitation of rockfish stocks at a rate of $F_{40\%}$ with the overfishing limit set at $F_{35\%}$ (NMFS 1999). Subsequently, the validity of this limit has been called into question, and is now considered to be too high (in terms of fishing mortality rates) for many rockfish species (Brodziak 2002; Dorn 2002). For the Washington-Oregon-California rockfish complex, management is currently based on $F_{50\%}$. With generation times in the range of greater than 10 years, a 15-20% historical extent of decline trigger appears most appropriate.

Recent rate of decline

Some rockfishes off the Washington, Oregon, California coasts have exhibited declines of over 90% during the last 10-15 years. One of these species, bocaccio, for which biomass is currently estimated to be at 2-4% of its estimated unfished level, is considered by some to be at risk of extinction, and a petition for listing under ESA is currently being evaluated. Some rockfishes, including several from Alaska, have experienced severe declines from overfishing in the past, but have since stabilized or are rebuilding (Musick *et al.* 2000).

Relevant modifying factors

Although all rockfishes are appropriately considered of low productivity, rockfish species differ widely in habitats by species, and therefore their susceptibility to fishing pressure and extinction may differ widely on a species-by-species basis. Dorn (2002) found that west coast rockfishes have a much lower steepness (implying much greater susceptibility to depletion) than their counterparts in Alaskan waters. Also there is growing evidence that reproductive success of rockfishes on the west coast is subject to strong interdecadal climate variability (MacCall, *in press*). Many nearshore species for which little stock information is available are subjected to intense recreational and commercial hook-and-line fisheries (e.g., off Oregon and California) and managed only as members of a "nearshore" complex (Parker *et al.* 2000).

Codfishes

The codfishes demonstrate moderate productivity. All of the species in trade exhibit high fecundity ($>10^4$) with early maturity (ca. 3 years) and a moderate life span (>10 years). All species are benthic broadcast spawners, and are found closely associated with substrate, as in Pacific cod, or in semi-demersal aggregations several meters from the bottom, as in walleye pollock.

Historical extent of decline

In 1999, fisheries management plans for Alaska called for exploitation of codfish stocks at a rate of $F_{40\%}$ with the overfishing limit set at $F_{35\%}$ (NMFS 1999). With generation times in the range of 5-10 years, the 5% trigger appears most appropriate.

Recent rate of decline

Some Pacific and Atlantic cod (*Gadus* spp.) stocks have experienced declines of 80-98.9% (Musick *et al.* 2000). Other populations may be experiencing overfishing but are not at risk of extinction.

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SMALL PELAGIC FISHES

The small pelagic fishes: sardines, menhadens, anchovies, mackerels, scads, herring, capelin, sand lance, eulachon and others, provide about a third of the world's fish catch and are a key forage for many species of fishes, marine mammals and birds. All species are highly productive; for example, Murphy (1967) estimated r (the intrinsic rate of nature increase under a stable age distribution) for the Pacific sardine to be 0.338. The age of first spawning in clupeoid fishes varies from one year old for some tropical and temperate clupeoids, to 4-5 years old in some herring stocks (Blaxter and Hunter 1982). Pelagic spawners produce hundreds of thousands of small pelagic eggs each year, while the demersal spawners from high latitudes such as herring and capelin produce fewer but larger demersal eggs. Most clupeoids have a short life span and rarely live longer than 5-10 years, except for some herring stocks. Most stocks show striking interannual or decadal variation in productivity and abundance. Fisheries for clupeoids can vanish for 50-100 years then undergo a remarkable recovery with the population growing as fast as 40% per year.

A risk exists that stocks of small pelagic fishes might become extinct, or might never recover, if fishing continues after the biomass has collapsed to about 5% of estimated unfished biomass, despite the fact that recoveries have been sustained from lower biomass levels. Declines to 30% of the unfished biomass fall well within the normal range of variability for small pelagic fishes (Smith *et al.* 1992). Hence, a 30% decline would be an inappropriate

trigger for CITES listings for this group of fishes. Evidence based solely on recent rates of decline over the last 10 or so years is a poor indicator of the current survival potential given the great population variability of stocks of small pelagic fishes. The extent of the total decline measured over the last 20-60+ years is far preferable measure. Long-term persistent trends in low productivity seem to be the major biological risk involved in sustaining small pelagic fish stocks under heavy fishing. Such trends are best identified using as long a time series as possible (see **Appendix IV** of this document for hypothetical examples, based on sardine-like dynamics, that demonstrate the importance of considering long-term trends).

The tendency in many species of small pelagic fishes to be distributed into a number of semi-isolated stocks probably enhances the survivorship of the species. Given sufficient time, one regional stock may become the source of replenishment for other depleted or lost stocks. For example, paleoecological evidence indicates that the temperate northern anchovy vanished from the Gulf of California around 1930 where it had been prevalent for the previous 200 years (Holgren and Baumgartner 1993). No anchovy were detected in numerous surveys of the Gulf over the ensuing 55 years until 1985 when the first reappearance was noted, and the population has been increasing ever since. Presumably, the population was re-seeded by Pacific coast immigrants migrating around the tip of Baja California through tropical waters. Such processes of local extinction and recolonization may decrease genetic variability.

The level of genetic variability in Pacific sardine is much lower than that of northern anchovy and most other clupeoids. Simulations by Gaggiotti and Vetter (1999) indicated that moderate differences in life history such as those that exist between northern anchovy and Pacific sardine can lead to substantial differences in genetic variability. The low heterozygosity in Pacific sardine is most likely due to life history traits and the recent arrival (founder effect) of the Pacific sardine population in the California Current. Despite the huge reduction in population numbers, the effect of the sardine fishery collapse on heterozygosity is relatively slight. The authors pointed out that, for a fishery collapse to have an effect on heterozygosity similar to that of a bottleneck, the population has to reach such low levels that extinction due to demographic or environmental stochasticity is likely. An important characteristic of

small pelagic fishes is that are small forage fishes that school for nearly all their lives. Only during the first 1-4 weeks as eggs and early pelagic larvae are they not in schools. Their survival depends upon the existence of schools of some minimum size. Thus, the size of the stock is better characterized in terms of the number and sizes of schools or total biomass rather than by the number of individuals in the population. Even at extremely low and unstable levels, much below our ability to measure such things, small pelagic stocks would be composed of thousands if not millions of individuals.

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HIGHLY MIGRATORY SPECIES

Highly migratory fish species include many of the tunas, swordfish, marlins, and spearfish. Usually, they have relatively rapid initial growth, reach large size, undertake semi-predictable migrations that may span entire oceans, and are opportunistic predators. Other factors affecting productivity and resilience may vary considerably. For example, skipjack tuna (*Katsuwonus pelamis*) mature at ages 2-3, have a maximum life span of about 6-8 years, and spawn opportunistically throughout the year in vast areas of the Atlantic

Ocean; while western Atlantic bluefin tuna (*Thunnus thynnus*) mature at about age 8-10, have a maximum life span of 20-30 years, and reproduce at most once per year in one or two known discrete locations. With the possible exception of western Atlantic bluefin tuna, most other highly migratory species can be considered to have high, or moderately-high productivity.

There is considerable variation in current stock status. The Atlantic highly migratory species under the jurisdiction of the International Commission for the Conservation of Atlantic Tunas (ICCAT) will be used as examples below. Most of the following information is extracted from ICCAT (2001).

Historical extent of decline

Current Atlantic yellowfin tuna (*Thunnus albacares*) biomass is estimated to be near or slightly above B_{MSY} , and fishing mortality is estimated to be near the level associated with MSY (viz, F_{MSY}). For bigeye tuna (*Thunnus obesus*), the 1998 biomass was estimated to be 57-63% of B_{MSY} and the 1998 fishing mortality was estimated to be 150-182% F_{MSY} . For Atlantic skipjack, the current status is unknown but French and Spanish vessel owners have voluntarily ceased fishing for three months of each fishing season out of concern for the long-term viability of the Atlantic stock. For albacore (*Thunnus alalunga*), the North Atlantic stock is estimated to be about 68% of B_{MSY} with a fishing mortality of 110% F_{MSY} ; the South Atlantic stock is estimated to be 160% of B_{MSY} with a fishing mortality of 57% of F_{MSY} . The western Atlantic bluefin tuna stock is estimated to be at 19-21% of the 1975 biomass and 10-36% of B_{MSY} , with a fishing mortality rate of 137-222% of F_{MSY} . The eastern Atlantic and Mediterranean bluefin tuna stock is estimated to be at about 19% of the 1970 biomass. Blue marlin (*Makaira nigricans*), white marlin (*Tetrapturus albidus*), and sailfish (*Istipophorus albicans*) / spearfish (*Tetrapturus pfluegeri* and *T. belone*) are estimated to be at 40%, 15%, and 60-90% of B_{MSY} , respectively. Atlantic swordfish (*Xiphias gladius*) are estimated to be at about 65% of B_{MSY} in the North Atlantic and 110% of B_{MSY} in the South Atlantic.

In summary, in most cases, stock size is below B_{MSY} , but is not below 10-20% B_{MSY} , which would be of the order of 5-10% or less of the unexploited baseline for most of the Atlantic highly migratory

species assessments. The two possible exceptions are Atlantic bluefin tuna and white marlin, both of which have been placed under increasingly strict management.

Recent rate of decline

Western Atlantic bluefin tuna declined rapidly in the mid to late 1970s, but have been relatively stable for the past two decades and are currently the subject of a rebuilding plan. North Atlantic swordfish were also exhibiting a rapid decline a few years ago, but catches have been substantially reduced as part of a rebuilding plan. There are also efforts underway to reduce commercial bycatch of blue marlin, white marlin, sailfish and spearfish.

Relevant modifying factors

Moderate to high productivity and oceanic distributions should be factors that mitigate risk of biological extinction. However, this is tempered to some extent by the high value of many highly migratory species, which makes it cost effective to continue exploiting them even when stocks are depleted. In addition, the relatively recent adoption of fishing techniques utilizing fish aggregating devices (FADs) for capturing skipjack, bigeye and yellowfin tunas, which greatly increase catch rates and also result in larger catches of juveniles, may have increased the vulnerability of the species caught by this method.

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SHARKS

The chondrichthyian or cartilaginous fishes are comprised of elasmobranchs (sharks and rays) and holocephalans (chimaeras and elephant fishes). They are one of the oldest living groups of jawed vertebrates and have evolved independently for at least 400 million years. Sharks are classified into three superorders and eight orders in the subclass Elasmobranchii and comprise approximately 394 extant species inhabiting continental and insular shelves,

open oceans, continental and insular slopes, and freshwater (Compagno 1990, Cortes 2000). Sharks give birth in a variety of habitats from the deep ocean floor to coral reef environments, but many of the commercially important species have pupping and nursery areas in estuaries, bays and the shallow near shore waters. Most sharks are apex predators feeding on the upper trophic levels (*Carcharodon carcharias*), but a few are planktivorous (*Cetorhinus maximus*). Many of the larger species have worldwide ranges; some make frequent trans-oceanic migrations (*Prionace glauca*); and others have more localized distributions (*Ginglymostoma cirratum*).

Although sharks exhibit considerable taxonomic, morphological, ecological and behavioral diversity (Compagno 1990), they share common life history traits and strategies that make them more similar to marine reptiles and mammals than bony fishes. These biological characteristics include: slow growth rate, large adult size, late age at maturation, low fecundity (small numbers of relatively large, precocial young), extended reproductive cycles, lengthy gestation periods, and long life-spans. In general, this combination of factors gives rise to long generation times and low reproductive potentials for many species of sharks.

Fecundity varies by species, with the number of embryos ranging from 1 or 2 (*Alopias superciliosus*) to 300 (*Rhincodon typus*). Reproductive cycles and gestation periods each are usually 1 to 2 years and may run concurrently or consecutively (Castro *et al.* 1999). Known gestation ranges from 70-80 days (*Chiloscyllium griseum*) to 23 months (*Squalus acanthias*). Size at birth can be over 150 cm (Alopiidae) with offspring size expressed as a percentage of maternal size ranging from 3% (*Cetorhinus maximus*) to 49% (*Loxodon macrorhinus*) and averaging 27% (Cortes 2000). Typically, species of large sharks mature at approximately 200 centimeters and bear 4 to 16 young that are 30 to 50 centimeters long at birth (Pratt and Castro 1990). Sexual maturity generally occurs at about 75% of maximum size for both males and females. Most shark species grow slowly and take years to reach sexual maturity with female maturity ranging from 1 year (*Rhizoprionodon taylori*) to 29 years of age (*Squalus acanthias*). Age at maturity is generally reached at about 50% of the maximum age in both males and females. Sharks are among the longest-lived fishes with a reported maximum age of 70 years or more (*Squalus acanthias*). Longer-lived

species of sharks tend to complete their growth at a slower rate than shorter-lived species (Cortes 2000).

Elasmobranchs produce relatively few young and the level of recruitment is largely determined by the time they are born (Holden 1977, Bonfil 1994). This results in a stock-recruitment relationship that is linear or slightly curvilinear. Sharks and rays cannot exploit favorable environmental conditions to the same degree as species with higher levels of reproductive output (teleosts) (Fogarty *et al.* 1990). Elasmobranchs appear to make up for low fecundity by investing in large offspring with relatively high survival rates (Frisk *et al.* 2001).

Vulnerability to excessive mortality is inversely proportional to the annual rates of increase (r) with groups that have r less than 10% being particularly at risk (Musick 1999). The annual intrinsic rate of population increase for shark species calculated from a variety of sources generally ranges from 1-10% (Hoenig and Gruber 1990, Sminkey and Musick 1996, Cortes 1999, Heppell *et al.* 1999) with a maximum of 22.8% (Liu and Chen 1999) and 32.7% (Cortes and Parsons 1996).

Historical extent of decline

Elasmobranchs are particularly vulnerable to over-exploitation due to K-selected life history characteristics such as slow growth rates and low rates of reproduction, and the fact that stock and recruitment are tightly coupled. Historically, directed fisheries for sharks have been characterized as “boom and bust” enterprises. Most targeted shark fisheries where there has been no regulation or management have been short-lived. The general course is that after initial exploitation, there is a rapid decline in catch rates, followed by a collapse of the fishery. Recoveries of population numbers from severe depletions caused either by natural phenomena or human action takes many years for the majority of elasmobranchs (Holden 1977).

The history of the porbeagle shark (*Lamna nasus*) fishery in the western North Atlantic is an example of a typical “boom or bust” fishery when a shark species is directly targeted. In 1961, a longline fishery was established by the Norwegians on a previously unexploited population of porbeagle sharks and ranged from the Newfoundland Grand Banks to waters off the Mid-Atlantic states of the east

coast of the United States. Vessels from the Faroe Islands also began fishing in 1961 in this same geographic area and combined landings from 1961 to 1964 increased from 1,924 to 9,281t, then declined sharply to 625t in 1967. A decrease in catch per unit effort (CPUE) and average size of the fish also occurred during this time period. The Norwegian fishery lasted from 1961-1966 with small catches in 1968, 1972, and 1984. By 1967, the fishery had almost disappeared with the Faroese fishery continuing at a very low level until 1994 (Anderson 1990, Campana *et al.* in press). Average catches of about 4,500t per year caused the fishery to collapse after only 6 years, while the recovery of the stock took another 20 years. Low and apparently sustainable catches of about 350t in the 1970s and 1980s allowed the stock to rebuild before a new Canadian fishery arose in the early 1990s. Catches of 1,000-2,000t throughout the 1990s have lowered catch rates, reduced the numbers of large sharks, and markedly lowered the numbers of mature females. Population dynamics analysis suggests that the porbeagle population in the western North Atlantic has again declined with stock abundance at about 15-20% of the size of the unexploited population that was present in the 1960s. Calculations show that even $F_{0.1}$ fishing target is inappropriate for the porbeagle shark and will eventually lead to stock collapse (Campana *et al.* in press).

Other examples of historical “boom and bust” shark fisheries include the harpoon fishery for *Cetorhinus maximus* off Ireland, the California drift-net fishery for *Alopias vulpinus*, the British Columbia *S. acanthias* fishery, the *Galeorhinus galeus* fishery of southern Australia, California, South Africa, southern Brazil, Uruguay, northern Argentina, and New Zealand (Walker 1998).

Annual catches from the United States east coast shark fishery (North Carolina to Texas) increased rapidly to more than 9,500t during the late 1980s and early 1990s. Catch rates of many of the species and species groups declined by about 50 to 75% from the early 1970s to the mid 1980s but that rapid rate of decline had slowed significantly in the 1990s. The 1998 stock size for the large coastal aggregation was estimated to be 30 and 36% of MSY levels. Stock size was estimated to be 58 to 70% and 44 to 50% of MSY levels for the two primary species in the fishery (*Carcharhinus plumbeus* and *Carcharhinus limbatus*, respectively) (NMFS 2000).

Overall, productivity and intrinsic rates of increase are low for shark species. In practice, smaller, fast growing, early maturing, and more fecund species are the basis of sustainable catches in managed shark fisheries in some parts of the world. Cortes (1999) cautions, however, that the effect of these management strategies on population abundance should be tracked for at least a generation. Based on the known ranges of k- and r-values for shark species, a historical decline percentage of the baseline level should vary by species or species group.

Recent rate of decline

The recent rate of decline might be more relevant for localized populations or for less highly migratory shark species. The more oceanic species should be better able to withstand fishing pressure and be less prone to depletion because of the greater likelihood of continual “seeding” by conspecifics from other areas within their extensive oceanic ranges (Smith *et al.* 1998).

Relevant modifying factors

Smith *et al.* (1998) used a demographic technique to compare the intrinsic rates of population increase in 26 shark species hypothetically exposed to fishing mortality and classified shark species into three general groups. These groupings are similar to those described by Cortes (2000) based on life history patterns and correlations. Sharks with the highest value for rebound capabilities were smaller, inshore coastal species that mature early and tend to be comparatively short-lived (e.g., *Rhizoprionodon terraenovae*). Those with the lowest recovery capabilities tended also to be coastal species but were generally medium to large-sized sharks, slow growing and late to mature (e.g., *Carcharhinus obscurus*). The sharks within the mid range of rebound values were mostly large (> 250 cm maximum size) pelagic species, relatively fast growing and early maturing (e.g., *Prionace glauca*).

Smith *et al.* (1998) cautions that these intrinsic rebound potential values should be used within a broad context of considerations covering the vulnerability of a stock and that many other factors such as innate plasticity of growth and regeneration rates, extent of geographic range, abundance, extent of stock mixing, and vulnerability to fishing on

pupping, nursery and feeding grounds should also be taken into account when developing management regimes.

Vulnerability Factors

Low absolute numbers or biomass

Sharks are apex predators and, therefore, their abundance is relatively small compared to other trophic levels. Some species, e.g., *Carcharodon carcharias*, may exist at very low levels even in an unexploited state. In addition, sharks have unique life history characteristics as K-strategists, limited compensatory mechanisms, a tightly-coupled stock and recruitment relationship, and generally lack validated age estimates.

Social structure / Age, size, or stage structure of a population

Many species segregate by age, size, sex, and reproductive state that could make a particular life stage vulnerable to exploitation. Juveniles of some species have been found to be more susceptible to overexploitation. The high mobility of many species, which often involves trans-boundary migrations, makes determination of stock structure difficult. Determination of the reproductive cycle and gestation time is also complicated by the shoaling and migratory activities of most sharks that may result in certain stages being inaccessible for sampling.

Reduced genetic diversity

Allozymes and mitochondrial DNA typically reveal levels of genetic variation within sharks that are significantly lower than those in marine teleosts. Sharks exhibit little genetic heterogeneity across wide geographic ranges (e.g., *Prionace glauca*, *Squalus acanthias*), which may be due to their age structure and long generation times (Heist 1999).

Specialized niche requirements

Pupping and nursery areas in estuaries, bays and shallow near shore waters for commercially important species offer the young some protection from predators and an abundant food supply, but also make them accessible to subsistence and modern fishing operations (Branstetter 1990, Castro *et al.* 1999) and

susceptible to impact by pollution, encroaching industrialized development, and overburdening recreational use (Pratt and Casey 1990). Site affinity, homing ability, and/or a home range has been established for some shark species. Dietary specificity has been reported for some species of shark that demonstrate selective feeding at least in some parts of their geographic range (e.g., *Isurus oxyrinchus*; Stillwell and Kohler 1982).

Selectivity of removals

Finning (i.e., cutting off the fins from the shark and dumping the carcass) occurs in some fisheries, causing high shark mortality. In general, fin traders select for larger fins, white fins over black fins, and fins with higher needle content, thereby increasing fishing pressure on larger fish of all species and certain species with choice fins. Selective removal of large females in the northwest Atlantic *Squalus acanthias* fishery has decreased their average size in the catch and has contributed to the overexploitation of the females (Rago *et al.* 1998).

Uncertainty

There is uncertainty about the status of many species of shark in terms of abundance, recruitment, and exploitation rates. There are species identification problems with some groups of sharks (e.g., Carcharhinids) that partially contribute to the fact that catch information and fishery statistics are not species specific and are often lumped into aggregate categories. Species could be at high risk of depletion without even being identified in the catch (Walker 1998). Sharks are often not caught in directed fisheries, but as incidental or bycatch in multispecies and multigear fisheries. Baseline information on fisheries that catch shark, historical abundance levels, time-series of catch and effort data, and information on the effects of trade is generally lacking. Lack of historical time series (i.e., less than 15-20 years) in a particular region might not reflect the overall stock trends, and short time series for limited areas do not necessarily indicate historical status of the stock.

Mitigating Factors

Adaptations to small population size

Elasmobranchs have complex reproductive life histories, specialized organ systems for reproduction

and complex precopulatory and copulatory behaviors including sexual segregation, mating displays, female refugia, sexual selection, social hierarchies, sexual dimorphism in skin thickness and of teeth, and sperm storage and competition (Pratt and Carrier 2001). In addition, their high parental investment in relatively few, large, and precocial young may give them a competitive advantage over most teleosts in the more uniform marine environment (Compagno 1990). Many species are highly migratory (Kohler *et al.* 1998) which would allow them to travel out of an area where the habitat is less than favorable. This is especially true for the adult sharks and also younger juveniles and pups that may not migrate long distances, but can move within a localized area. Even in the smaller species with relatively limited geographic ranges, complex movements can occur within given populations, influenced by ontogenetic stages, local current patterns and annual and long-term water temperature cycles (Compagno 1990). Sharks have large livers which store high-energy, fatty acids to provide buoyancy and an energy reserve which can be drawn upon during times of low food availability. Recent studies have shown that shark populations have limited compensatory mechanisms and varying ability to recover from exploitation.

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DEEPWATER SPECIES

Deepwater species are usually classified as those that spend most of their time below a depth of about 300 m (J.A. Moore, pers. comm.). In addition, the distribution of some species that primarily reside in shallower waters may become progressively deeper as the fish become older and larger. Some deepwater species, particularly those classified as mesopelagic fishes, have relatively high productivity and may have extremely high biomass. Mesopelagic fishes such as the lanternfishes (myctophids) and pearlsides (sternoptichids) form a deep scattering layer around 200-600 m during the day and migrate to the surface at night (Moore 1999). Fisheries for mesopelagics have been limited by the small individual size and high content of oils and wax esters. By far the majority of deepwater fisheries have been focussed on deepwater demersals. For most deepwater demersals that have received adequate study, it has been demonstrated that productivity, as indexed by growth rates, age of maturity, fecundity, and longevity is extremely low. For example, Mace *et al.* (1990) estimated that for orange roughy (*Hoplostethus atlanticus*) populations in New Zealand, the age of maturity was in excess of 20 years, natural mortality was less than 0.05, and the Brody growth coefficient, K , was about 0.06. Subsequent studies have suggested an age of maturity of about 28 years and a maximum age of 100-200 years.

Historical extent of decline

Most deepwater fisheries have exhibited a now-familiar evolution over time: discovery, rapid development, depletion of the resource, and slow recovery (Moore 1999) or, in some cases, no recovery. Several examples of this pattern, or

variations thereof, are given below.

Atlantic halibut

A directed fishery for Atlantic halibut (*Hippoglossus hippoglossus*) began in the United States in the early 1800s as a shallow-water fishery, and gradually moved into deeper waters off Georges Bank and Nova Scotia as the near-shore areas were depleted. By 1884, fishing was almost exclusively conducted in the 200-640 m depth range. The fishery started showing signs of severe depletion around the turn of the 20th century and was commercially extinct by the 1940s. In the Georges Bank / Gulf of Maine area, landings dropped from 4000-5000 mt in 1895 and 1896 to an annual average of 475 mt from 1900-1940, 144 mt from 1941-1976, and 95 mt from 1977-1999. Landings were not recorded prior to 1893, although they were probably substantial. According to A.B. Howe (Massachusetts Division of Marine Fisheries, pers. comm.), by 1900 landings had already declined 95% from the 1879 levels. Thus, in contrast to modern-day deepwater fisheries, it took more than 100 years to deplete the fishery to the point of commercial extinction. The historical and current size of the population is not known, even in relative terms; however, based on landings information, it appears likely that the stock has been reduced to well below 5% of historical abundance, and that there has been little if any evidence of recovery for more than 50-60 years, despite substantial reductions in landings.

U.S. west coast Pacific Ocean perch

Fisheries for Pacific Ocean perch (*Sebastes alutus*) were prosecuted at depths of 150-450 m from Oregon to British Columbia beginning in 1946. Along the U.S. west coast, annual catches of about 2000-6000 mt were being recorded by the early 1960s, but the arrival of Soviet and Japanese fleets in the mid-1960s resulted in catches escalating to more than 23,500 mt in 1966, 34,000 mt in 1967, and 19,300 mt in 1968. By 1970, the stock had been severely depleted and catches had been reduced to 5000-6000 mt. Since 1979, landings have invariably been less than 2000 mt per annum and since 1994, landings have been less than 1000 mt per annum. An active program to rebuild the stock has been in place since 1983 when the first fisheries management plan was implemented. The most recent stock assessment (Ianelli *et al.* 2001) indicates that the stock continued

to decline until the early 1990s, when it reached about 16% of the unexploited level, and may now be slowly recovering 30 years after being severely overfished. In fact, the west coast stock of Pacific Ocean perch may never be able to support a sustained harvest of more than a few hundred or a few thousand metric tons per year. Recent analyses (Dorn 2002) suggest that the compensatory response of the stock to harvesting is unusually weak. Dorn also notes that there are four other stocks of Pacific Ocean perch, all of which were depleted by foreign fleets in the mid-1960s to 1970s, and that those in the middle of the range have subsequently rebuilt strongly while those at the southern and northern limits of the distribution (U.S. west coast and eastern Bering Sea, respectively) have been slow to rebuild and have exhibited low compensation.

New Zealand orange roughy

The initial commercial successes of deepwater fisheries for orange roughy in New Zealand and Australia, combined with increasing levels of depletion of nearshore resources, may have spurred the current spate of explorations by several countries seeking to develop new commercially-viable deepwater fisheries (Moore 1999). However, with one possible exception, the history of the development of orange roughy fisheries within and near New Zealand (and Australian and other) waters has been one of unintentional serial depletion. The possible exception is the largest of the New Zealand fisheries, which occurs on the Chatham Rise and surrounding areas in what is called management area 3B. This area is extremely large and it is not known how many true populations or stocks of orange roughy actually exist there. Recent assessments assume three primary areas: northwest Chatham Rise, northeast Chatham Rise, and South Chatham Rise; and four surrounding areas that are considered part of the same management unit: Puysegur Bank, Auckland Islands, the Arrow Plateau, and the Antipodes Islands (Annala *et al.* 2001). In each case, the biomass associated with maximum sustainable yield (B_{MSY}) is estimated or assumed to be 30% of the unexploited level.

For the northwest Chatham Rise, catches peaked at 8400 mt in the second year of the fishery (1980-81), gradually declining to current levels of about 2000-2500 mt. The biomass in 1999-2000 was estimated to be near or somewhat above B_{MSY} , but is

continuing to decline, and current catches are not sustainable.

For the northeast Chatham Rise, which supported by far the largest unexploited biomass of any of the New Zealand (or other known world-wide) orange roughy stocks, landings peaked at 29,100 mt in the second year of the fishery (1979-80), declining to 12,200 mt in 1990-91. Subsequently, the distribution of orange roughy catch within management area 3B has been affected by a series of agreements with the fishing industry. The catch within the northeast Chatham Rise subarea was 4600 mt in 1999-2000. A recent assessment (Annala *et al.* 2001) indicates that the unexploited stock (late 1970s) was about 350,000-400,000 mt, that it declined to 25-35% of this level in the early 1990s and has since rebuilt to about 40-50% of the unexploited level. Current catch levels are predicted to result in further rebuilding.

For the south Chatham Rise, catches peaked at 11,000 mt in 1989-90 and have subsequently decreased rapidly to 1100 mt in 1999-2000. This fishery has exploited a series of seamount complexes on which catches and catch rates are usually extremely high initially and decline rapidly thereafter. For example, for the seamount complex dubbed "Graveyard", annual catches from the start of the fishery in 1992-93 to 1999-2000 were 3300, 2200, 1500, 1800, 900, 800, 900, and 600 mt, respectively. Corresponding average catch rates over the same period were 11.0, 6.1, 4.2, 5.1, 3.6, 2.7, 5.0 and 2.6 mt per tow. Overall, the current south Chatham Rise stock is estimated to be at about 24% of the unexploited biomass level of 95,000 mt, but may be in the process of rebuilding.

On Puysegur Bank, catches peaked at 6950 mt in the third year of the fishery (1991-92), but declined to 550 mt within a further 5 years, and the fishing industry has now agreed to a voluntary quota of 0 mt. On the Macquarie Ridge off the sub-Antarctic Auckland Islands, the fishery peaked at 1250 mt in its third year (1994-95), declining rapidly to 200 mt in 1999-2000. On the Arrow Plateau east of the Chatham Rise, catches peaked at 850 mt in the third year of the fishery (1994-95) and declined to 200 mt in 1999-2000. A new fishery to the west of the Antipodes Islands began in 1995-96 with a catch of 3400 mt; however, catch dropped in 1996-97 to 700 mt, despite almost double the number of targeted tows, and has decreased subsequently to 370 mt in 1997-98, 120 mt in 1998-99, and to only 2 mt in

1999-2000.

The orange roughy fishery on the Challenger Plateau off the west coast of New Zealand is perhaps the most extreme example of stock depletion for this species. Catches peaked at 10,000-12,000 mt per year for the 1986-87 to 1988-89 fishing years more or less in accordance with an “adaptive management” quota in place at the time. The following year, the quota was dropped to 2500 mt, although more than 4000 mt was caught. In 1990-91, the quota was dropped to 1900 mt but was undercaught in most subsequent years. The quota was further reduced to 1425 mt for 1998-99 and 1999-2000. The recorded catch in 1999-2000 was 629 mt. The most recent assessment (Annala *et al.* 2001) indicates that the unexploited biomass (late 1970s) was about 91,000 mt, but that current biomass is only about 3% of this level. Average catch per unit effort has declined rapidly from 15.8 mt per tow in 1982-83 to 0.9 mt per tow in 1998-99. If current levels of catch continue, the stock is expected to decline further.

The East Cape stock (east coast of New Zealand, relatively nearshore) is currently estimated to be about 14-18% of its unexploited level of 18,000-19,000 mt, while the mid-east coast stock is estimated to be about 10% of the unexploited level of 119,000 mt (Annala *et al.* 2001). The west coast South Island stock is estimated to be about 12% of the unexploited biomass of 12,700 mt.

In summary, New Zealand orange roughy should be classified as a very low productivity species on the basis of life history characteristics. However, it has been estimated that B_{MSY} is about 30% of the unexploited level, so this is not an appropriate trigger for highlighting concern about risk of extinction. The only New Zealand orange roughy stock that appears to be exhibiting signs of recovery (the northeast Chatham Rise stock) seems to have rebounded from a biomass of about 25-35% of the unexploited level. However, many other New Zealand orange roughy stocks or sub-stocks have been depleted to much lower levels and it remains to be seen whether or not and how rapidly they will recover.

Other deepwater fish

The following examples are extracted from Moore (1999) and Moore and Mace (1999).

- In 1882, just after the commencement of a deepwater fishery for tilefish (*Lopholatilus chamaeleon-ticeps*) off the northeastern United States, an influx of very cold water caused a massive die-off of an estimated 1.5 billion or more tilefish. It took 33 years for the species to recover to levels sufficient to support a viable fishery but even so, catches have subsequently been below 1500 mt in most years.

- From 1969-75, landings of pelagic armor-heads (*Pseudopentaceros wheeleri*) in the southern Emperor Seamounts and northern Hawaiian Ridge averaged 22,000 mt per year, but since 1977, catches in all but one year have been in the range of 1000-2000 mt.

- A Soviet fishery for roundnose grenadier (*Coryphaenoides rupestris*) in the northwest Atlantic recorded peak landings of 82,000 mt in 1971 but, since the early 1980s, catches of this species have declined to less than 3000 mt per annum.

- A Soviet fishery for slender alfonsino (*Beryx splendens*) in the eastern Sargasso Sea recorded its highest catch in the first year of the fishery, collapsed in the second year, and only began to recover ten years later.

Deepwater invertebrates

Although few deepwater invertebrates have been targeted for harvest (the exception being a few small fisheries for deepwater crabs, shrimps, and antipatharians and precious corals), some deepwater fisheries may severely impact non-target invertebrate fauna that provide habitat structure. In particular, deepwater bottom trawls can locally deplete gorgonian and scleractinian corals that grow so slowly that it may take over a century with no further disturbance for them to rebuild back to the undisturbed condition. Recolonization and recovery rates of deepsea benthos also appears to be much slower than in shallow habitats, and after disturbance of deepsea sediments, it may take several years to replace the invertebrate fauna (Moore and Mace 1999).

Recent rate of decline

For many deepwater fish stocks, the recentness

of discovery and exploitation and the subsequent rapid decline mean that the (historical) extent of decline and the recent (5-10 year) rate of decline are almost synonymous. Pacific ocean perch, some stocks of New Zealand orange roughy, pelagic armorhead, and wreckfish (*Polyprion americanus*) off the southeast coast of the United States are all examples of fisheries that went from very low catch levels to levels far in excess of sustainable catches within five years. In almost all cases, the rate of decline at any point during the active phase of the fisheries has probably been sufficient to drive the stocks to a level of about 20% of the unexploited baseline within 5-10 years. However, for most deepwater fisheries, the stocks have been rapidly depleted to the point of commercial extinction so that fleets have either voluntarily left the fishery or a management authority has imposed a moratorium or near-moratorium on directed fishing for the species. Some of the depleted stocks are believed to be slowly recovering but the rate of recovery is so small that it is difficult to detect.

Relevant modifying factors

Most mesopelagic species are small (< 15 cm in length) and have relatively high turnover rates, reaching sexual maturity in 1-2 years (Moore 1999). On the other hand, deepwater demersal species are generally extremely long-lived and slow-growing with a somewhat larger maximum size of 40-100 cm. Species that frequent shallower waters at younger ages may grow larger than those that spend most of their lives in deep waters.

Although there are insufficient data to enable generalization, it is possible that many deepwater species are characterized by highly variable recruitment with extremely large year classes occurring infrequently, and most other year classes being inadequate to replace the population. This appears to be the case for the ovoviparous Atlantic redfish, *Sebastes fasciatus* and *S. mentella*, in the Gulf of Maine and Flemish Cap. For example, *S. fasciatus* in the Gulf of Maine has experienced about four pulses of recruitment in the past 35 years (about one pulse every seven years), with much lower levels of recruitment between, although it appears that recruitment may have been larger and more even in the 1960s and earlier. Many U.S. Pacific coast *Sebastes* species also appear to have high recruitment variability (Dorn 2002). There is also weak evidence

for high recruitment variability in New Zealand orange roughy.

Very deep waters may also provide a refuge for the larger adults of some species that are heavily fished in shallower waters. Technology that has enabled progressively deeper fishing may ultimately destroy such "natural" refuges, thereby increasing the vulnerability of several important commercial marine species.

History has shown that depletion rates for many long-lived, slow turnover deepwater species can be extremely rapid – so rapid in fact, that it seems unlikely that an organization such as CITES could usually react sufficiently fast to avert severe depletion. The saving grace for many deepwater stocks may be that the economic arguments against overexploitation commonly advanced and subsequently disproved in the past for shallow water species may work much better for deepwater species, given current technological constraints. Due to the high capital costs of constructing offshore vessels, the operating costs of traveling to offshore locations, the huge volumes of water that need to be searched, the time and labor involved in deploying gear to great depths, the expense of developing new markets for novel products, and long time lags before stocks can be replenished, fishing for deepwater species may often cease to be profitable well before a species approaches biological extinction.

Conclusion

As a generalization, and based on very few case studies to date, it appears that deepwater demersal fishes may be capable of recovering from declines to biomass levels as low as 10-30% of the unexploited level, but that recovery may be extremely slow. However, how the relatively rapid removal of large proportions of the biomasses of dominant species affects deepwater ecosystems is unknown.

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SALMONIDS

Salmonids under NMFS' jurisdiction in the United States include five species of Pacific salmon, steelhead trout, and Atlantic salmon. While there is considerable variability in life history, all are anadromous, relatively short-lived, and fast growing. Lifespan varies from 2 to 8 years and fecundity ranges from about 2,000 to 6,000 eggs per female. Pacific salmon are all semelparous, while steelhead trout and Atlantic salmon can be iteroparous. As a group, salmonids should probably be considered highly productive species.

There is international trade in all species, although only Pacific salmon have extensive capture fisheries. Worldwide, most production of Atlantic salmon, steelhead trout, chinook salmon, chum salmon, and coho salmon for commerce is currently from aquaculture and artificial propagation. Only fisheries for sockeye salmon and pink salmon rely primarily on natural production

Historical extent of decline

The extent of decline in abundance has varied widely by species and location. Many populations in the contiguous United States have been extirpated, primarily by the loss of habitat resulting from construction of impassable dams. Many less productive populations have been extirpated or driven to the brink of extinction by overharvest. In Alaska, pink salmon, chum salmon and sockeye have been at or near record abundance levels as indicated

by fishery landings data.

Distinct population segments that have been listed under the U.S. Endangered Species Act have experienced declines in natural populations that range from reductions down to <0.1% to something on the order of 15% of historical abundance. Listings with smaller declines have been influenced by additional risk factors.

Recent rate of decline

Recent rates of decline also vary widely on a population by population basis. For populations in Washington, Oregon, and California, recent (10 year) average rates of change range from -30% per year to + 50% per year.

Relevant modifying factors

Anadromy coupled with fidelity to natal stream results in an extremely complex stock structure. For example, the state of Washington recognizes 435 individual management units of salmon and steelhead, most of which are composed of many individual spawning populations. This complexity precludes complete assessment of all populations, resulting in monitoring efforts that focus on the most abundant and productive populations.

Abundances of Pacific salmon populations are influenced by changes in marine regimes that occur on decadal scales. However, the effect of regime shifts varies with location and species. In general, since the mid-1970s prevalent conditions in the north Pacific have been unfavorable for chinook and coho salmon in the contiguous United States and Canada, and have been favorable for pink, chum and sockeye salmon in Alaska.

Because of their anadromous life history, salmonids are vulnerable to loss and degradation of freshwater habitat. The availability and quality of freshwater spawning and rearing habitat is often the limiting factor in stock productivity. Individual populations differ widely in both size and productivity. Healthy populations in good habitat can sustain harvest rates in excess of 90% under favorable marine conditions, while populations in marginal habitat may not be able to sustain any harvest.

To mitigate losses of freshwater habitat and en-

hance fisheries, extensive hatchery programs have developed. In many locations, hatchery fish predominate in fishery landings and in some locations stray hatchery fish predominate over naturally spawning populations. Hatchery production increases the risk to natural production by masking declines in natural populations, diverting attention from habitat loss and degradation, disrupting the genetic integrity of wild populations adapted to local conditions through outbreeding, and leading to overexploitation of natural stocks in fisheries targeting commingled hatchery stocks.

Because of their short life history, fishery landings are dominated by one or two age classes, which are generally maturing fish on their spawning migrations. This abbreviated age structure provides little buffering of recruitment failures or management mistakes. It also results in very high annual exploitation rates; e.g., an $F_{35\% SPR}$ harvest policy translates into a 65% annual exploitation rate when a single age class is harvested.

The status of all salmonid species under NMFS jurisdiction outside of Alaska has been reviewed in response to petitions to list them as threatened or endangered under the U.S. Endangered Species Act. To date, 51 distinct population segments of Pacific salmon and steelhead trout have been identified, and 26 of these have been listed as threatened or endangered. All harvest is closely monitored and regulated, and all harvest impacts to listed populations are permitted as incidental take under no jeopardy biological opinions or reasonable, prudent alternatives.

COLD-WATER INVERTEBRATES

Here we consider North American marine invertebrates that are found north of Cape Hatteras on the Atlantic Coast and north of Point Conception on the Pacific Coast. While there is a very large, taxonomically diverse, group of marine invertebrate fauna in these areas, a much smaller and less diverse group of species is potentially relevant to CITES. These species are harvested in commercial, recreational, or subsistence fisheries as sources of food, chemicals or bait. Collecting activities may cause localized depletion of various invertebrates and can result in a considerable trade but have not involved endangered species or the serious depletion of species within the

area of interest. Fisheries for invertebrates may involve major economic endeavors, serious socio-economic considerations and harvests that represent considerable portions of the total mortality of various species. At present, none of the species harvested in the area are listed under the Endangered Species Act or CITES and fisheries management practices frequently close fisheries when stocks are in low abundance, which precludes trade. A large international trade exists for some of these species. Major taxa that are harvested include decapod crustaceans (shrimps, crabs and lobsters), bivalve molluscs (clams, mussels, oysters, scallops, snails and abalones), cephalopods (octopus and squids), echinoderms (sea urchins and sea cucumbers) and polychaete worms (blood worms and clam worms). All of these species are common animals in their respective habitats although abundance in some of them is extremely variable or even so low as to be designated as "overfished" in the sense of the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA).

Life history characteristics are diverse but all species considered here are dioecious, sexually reproducing species. Some are serial hermaphrodites (e.g., pandalid shrimps and oysters), others are semelparous (e.g., squids and octopus), and most are annually spawning multiparous animals. Some are the basis of important biocoenoses (e.g., oysters and mussels) or zones. Modes of reproduction range from broadcast spawning (e.g., clams, oysters, mussels and abalones), through copulation with external fertilization (e.g., crabs, lobsters and shrimps) to copulation with internal fertilization (e.g., whelks and other snails). The success of broadcast spawning may be limited by decreasing the density of individuals in dioecious species (e.g., some abalones and clams). The success of copulation may be influenced by the ability of females to store sperm and produce multiple clutches from a given mating (e.g., some crabs). The trophic levels are diverse including detritus feeders (e.g., some sea cucumbers), filter feeding herbivores (e.g., mussels, clams and oysters), grazing herbivores (e.g., abalones, sea urchins and some snails), benthic scavenger-predators (e.g., crabs, shrimps, lobsters, whelks and octopus) and pelagic piscivores (e.g., squids)

Examples from Alaska

There are four species (but six major stocks) of

king and tanner crabs in the eastern Bering Sea that support large fisheries and have been intensively managed over periods from 20-40 years. Population abundance information is derived primarily from an extensive bottom trawl survey conducted by NMFS each summer.

St. Matthew Island blue king crab (*Paralithodes platypus*), Tanner crab (*Chionoecetes bairdi*) and snow crab (*C. opilio*) are currently considered to be overfished under guidelines pursuant to the MSFCMA, while Bristol Bay red king crab (*Paralithodes camtschaticus*), Pribilof Island red king crab and Pribilof Island Blue King crab are not.

In computing MSY's for Bering Sea crabs under MSFCMA, a fixed fraction of the annual total mature biomass (TMB) was considered as sustained yield (SY) for that year and the average of SY's over a suitable period of time is considered as MSY. That suitable period of time (1983-1997) was specifically

chosen to exclude the late 1970s to early 1980s, a period when the five crab populations were apparently unusually large. The late 1970s to early 1980s were excluded because "the MSY computations required that environmental conditions remain reasonably constant over the period during which the SYs are averaged". It was felt that environmental conditions in the Eastern Bering Sea shifted in the late 1970s and early 1980s (crabs out, fish in), and the more recent conditions would not favor such large crab populations.

The table below shows how populations of these crabs have fluctuated over the last two or three decades. The numbers approximate the number of mature animals in the populations (in millions) from the NMFS trawl survey. The number of consecutive survey years for the various populations ranges from 21 to 31. (Pribilof red king crab has been excluded due to unreliable data).

POPULATION	CYCLES OF MAX & MIN POP SIZE. HI NUMBER/YEAR : LO NUMBER/YEAR	MSFCMA (1983-1997) MEAN POP SIZE AND ITS % OF MAXIMUM POP SIZE	% CHANGE IN POP SIZE OVER THREE GENERATIONS. KING CRABS = ~ 27 YRS CHIONOECETES CRABS = ~21YRS
Red king crab	252/1977 : 21/1993	30/12%	-73%
Blue king crab, Pribilofs	46/1978 : 1/1985	4/9%	-87%
Blue king crab, St Matthew	10.1/1982: 0.8/1986	4/40%	-60%
Tanner crab	two cycles in 25 years: 466/1976 : 29/1985 268/1991 : 21/1998	115/25%	-82%
Snow crab	two cycles in 21 years: 4260/1980 : 198/1985 4059/1990 : 568/1999	2078/49%	-51%

Population fluctuations of an order of magnitude or more in scale occur in periods as short as four or five years among these populations. All of the Eastern Bering Sea crab populations would probably qualify as low productivity in Musick's (1999) Table 3 by virtue of age at maturity and maximum age. Given the extent and rate decline values discussed by the Working Group as "triggers for concern about the

long-term viability of a species", several of these crab populations would probably qualify. Where they escape that fate is if the "harvesting program reductions" and/or "natural fluctuations" exemptions in the current Annex 5 definition of decline are invoked.

Some species have also declined dramatically in

the Gulf of Alaska in the last couple of decades presumably (largely) as the result of a climate driven regime shift. These include pandalid shrimps (*Pandalus borealis*, *P. goniuris*, *P. dispar*; ~95% decrease), Tanner crab, and red king crab. Because some stocks have declined to very low levels, fisheries for them have been closed and there is very limited possibility of international trade.

Other examples

Abalones are harvested from Alaska (one species) to California (5 species). They are also harvested in Canada (one species) and Mexico (several). There is international trade between Canada, Mexico and the United States and between North Americans and the far east, principally Japan. The white abalone is a trans-boundary species (United States and Mexico) that is severely depleted and may be involved in international trade where species may not be identified. A status review for the white abalone indicated the population was less than 0.01% of its pre-exploited size and current densities are far below those needed for successful reproduction. On the basis of this report, white abalone was listed as an endangered species by the Department of Commerce in May of 2001. Other species of abalone, most notably the black abalone, also may be severely depleted. Landings of California abalones declined sharply from the 1950's onward due to overfishing, spread of disease (withering syndrome), predation by re-introduced sea otters and poaching after the fisheries closed. Commercial fisheries are now closed in California but a limited recreational fishery exists in some areas.

Marine worms (e.g., blood worms and clam worms) are harvested extensively in Maine and although the weight of landings is low, the price is very high. Extensive harvest over many years has caused depletion in the intertidal portion of many bays. These worms are used as bait in sport fisheries and as specimens in teaching. They are shipped extensively. While it is unlikely that these species are in any danger of extinction, there have certainly been severe local depletions that have closed fisheries in extensive areas. The closure of fisheries in Maine may increase trade with Canada.

Horseshoe crabs are ancient organisms that are only distantly related to crabs. They were harvested for fertilizer many years ago and severely depleted in

several areas of the Atlantic Coast. Their main use now is as bait in whelk fisheries and for serum in certain medical applications. The fishery has recently been curtailed, in part because its spawn on beaches is thought to be important to migratory shore birds. There is at least some international trade of medical research products.

Sea urchins are rarely eaten in the United States but are esteemed in Japanese and Korean cuisines. They are considered keystone species because they may regulate kelp abundance. They are internationally traded and have become important fisheries in Maine and California where they are harvested in competition with sea otters. Should urchins become rare in California, they may be of interest to CITES.

WARM-WATER INVERTEBRATES

The taxonomic grouping of "warm-water invertebrates" covers a huge range of taxonomic diversity, even at the phylum level (i.e., from Phylum Porifera to Phylum Chordata). Thus, the group includes incredible life history diversity. They potentially have a large number of larval stages which may require distinct habitat types during different periods in their lives; they may exhibit early or late reproduction, their life-span may range from less than one year to many centuries, they exhibit vast reproductive strategies ranging from broadcast spawning, to brooding, to species that deposit eggs or species with direct development, to species with a clonal body plan and others that are asexual. The continua (both within and among taxonomic groups such as shrimps or corals) of productivity, clonality, and mobility are of great importance in determining the appropriate application of the proposed CITES criteria and guidelines. For organisms with internal fertilization, low density or population size is problematic, as difficulties in finding a mate result in reduced reproductive success. In particular, for sessile or semi-sessile broadcast-spawning species, absolute abundance or range criteria may be inadequate to assess threat because, at low density, fertilization success may drop to zero. In addition to the importance of density, fertilization success is affected by spawning group size, degree of aggregation, concentration of eggs and sperm, position within an aggregation (for species with separate sexes) and water flow. Also, many of these invertebrate groups are clonal, which severely complicates the application of criteria involving numbers of individuals,

traditional means for quantifying species productivity and, in some cases, biomass. For instance, biomass would not work well for stony corals or other sessile animals that produce a calcium carbonate skeleton, unless surface area can be correlated with biomass.

Warm-water invertebrates are subject to potential international trade in a wide range of sectors. Currently, the most common sectors include food (e.g., various crustaceans, echinoderms and molluscs) and the ornamental aquarium/curio trade. However, there are several ongoing bio-prospecting initiatives aimed at detecting and developing pharmaceutical or other biotech applications of natural products from marine organisms. It is plausible to expect that many of the ultimate products from these efforts will involve species in the "warm-water invertebrate" group, as it contains a large proportion of the total marine biodiversity. This means that international trade for biotech/natural product development must be considered as potentially relevant for all species in this group.

Relevant modifying factors

Warm-water invertebrates utilize a huge range of habitat types, many in tropical coastal areas that are particularly vulnerable to human destruction (e.g., seagrass, coral reefs and mangrove fringe). Also, the predominance of species in this group that are sessile or have limited mobility makes them particularly prone to habitat destruction on local scales.

Although certain marine invertebrates, including many crustaceans, are highly mobile as adults, the great bulk of species have adults with limited or zero mobility. However, these have planktonic larval stages, many of which are presumed to travel great distances away from the adult, thereby providing a means of gene dispersal. In these situations the area occupied by a genetic breeding unit, or "stock", is likely to be very large. In many species from the same and other groups, the lifespan of larvae is short, dispersal is much less, or larvae have little control over the distance or direction of travel. In these species, stock size is likely to be low, and hence they are much more vulnerable to overfishing (Thorpe *et al.* 2000).

Clonality is a particularly problematic consideration in the application of risk of extinction criteria. Clonal invertebrates often exist in colonies of

genetically identical modules (i.e., polyps); hence growth of the colony involves asexual reproduction at the polyp level. The colony is most often considered the appropriate unit for population ecology. To complicate things further, most clonal species are able to reproduce at the colony level by either asexual (fragmentation) or sexual (either broadcast spawning or brooding planktonic larvae; either hermaphroditic or gonochoric) means. The relative importance of these two reproductive modes will vary greatly by species. Species for which fragmentation is the dominant mode of reproduction might be expected to possess relatively lower levels of genetic diversity than an otherwise comparable species which has a dominance of sexual reproduction because a small number of successful clones (with many ramets) could come to dominate a population over time if less successful clones were gradually lost to disturbance and sexual recruitment was rare.

One of the largest limitations of any criteria used to define extinction risk is the lack of "historical" or baseline data for most warm-water invertebrates (with the exception of a few commercially important food species). For instance, in a coral reef ecosystem limited information is available on the status and trends of populations for isolated reefs over the last 10-20 years and very few data are available for longer time periods. For sessile organisms it may be more appropriate to define a "potential" abundance/cover or density. This could be an estimate of the possible cover or density that could be achieved in the particular habitat or zone where a particular species normally occurs. As our knowledge of the location of reefs improves, and better maps are available, it may be possible to obtain a more reliable estimate of the area of available habitat for at least some of the dominant species.

HIGH PRODUCTIVITY, INDIVIDUAL SPECIES

Penaeid shrimp - crustacean

Penaeus species are heavily exploited (9 million pounds annual harvest of pink shrimp in South Florida alone), omnivorous, high productivity species with generation time about one year and juvenile growth rates for pink shrimp of up to 1.7 mm per day. Pink shrimp populations are characterized by large fluctuations in recruitment and abundance that are primarily environmentally determined (e.g., by

salinity and temperature fluctuations). They display 11 larval stages utilizing a range of habitats from inshore estuarine marsh/mangroves to offshore soft bottoms down to 100 m depth. These species possess many of the classical r-selected life history characteristics such as high fecundity, short generation time, and aggregating behaviors that enhance reproductive success. Hence, a decline to 5% of the historical baseline of number of individuals or of some parameter of reproductive potential would be appropriate for this species. However, modifiers to this decline criterion would include habitat threats based on its complex life cycle that involved both estuarine and offshore habitats, and vulnerability to environmental change due to high sensitivity to environmental conditions such as salinity.

Mangrove oysters - bivalve molluscs with high productivity

Small bivalve molluscs such as oysters are sessile species, most with separate sexes, that broadcast gametes into the water for external fertilization. They reach sexual maturity in a relatively short period of time (approximately one year), each mature female produces millions of eggs, larvae have high survival, and juveniles grow rapidly. In response to natural fluctuations (e.g., die-off of sea grass, hurricanes, outbreaks of predators, and disease), populations of adults have declined by up to 90-95% in certain areas, and several decades were required for recovery (but they did recover); in most cases these die-offs were relatively restricted and the species as a whole was not threatened with extinction.

The number of individuals is most likely the best parameter for estimating the extent of decline, but populations could sustain a large decline without triggering concern. Reduction of a population to 5% its historical abundance is a reasonable trigger for concern. First, the number of offspring produced is largely independent of the size of the spawning stock (above some minimum undetermined number) due to niche specializations. At low density of adults, individuals produce large numbers of seed, whereas a high density of adults produces lower numbers of seed. The most relevant modifiers of the decline criteria include an abrupt change in environmental

conditions, including changes in temperature or salinity in the water column (this affects larval survivorship), and conditions in the clam beds (e.g., predator abundance and habitat degradation) that affect survival of juveniles. The only other modifier that may affect a species is the degree of endemism or habitat specialization; most bivalves have a widespread distribution and a good potential to recover through long-distance dispersal, but some that occur in restricted bays and estuaries may have a lower potential for dispersal and these populations would be more vulnerable to anthropogenic disturbances, and hence less resilient.

Sea urchins - echinoderms with moderate to high productivity

Sea urchins have separate sexes and they broadcast their gametes into the water for external fertilization. Once they reach sexual maturity, they continue to reproduce for many years, and may have a life span of 10 or more years. These species are mobile, but tend to have a relatively small home range. Some urchin species (e.g., *Diadema antillarum*) have displayed substantial fluctuations in abundance due to disease outbreaks. Recovery from extremely low levels of abundance has occurred in some areas over a decadal scale, but has not been observed in other areas. Thus, resilience appears to be moderate for urchins.

The area of distribution and abundance are the two most appropriate parameters to consider when evaluating the historical extent of decline. Small or rare urchins may aggregate, but fertilization success will be limited at low population density. Although the minimum density necessary for fertilization is unknown, and is likely to vary among species, urchins might be able to sustain a decline to <30% of their historical abundance. Modifiers of the decline criteria of primary importance are density of individuals, severe habitat loss or environmental change (e.g., loss of grass beds, change in algal composition on reefs, and disease), and environmental parameters (e.g., current regime) that will limit fertilization success. Selective removals of adults could also influence the ability of urchins to recover from a large extent of decline.

LOW PRODUCTIVITY INDIVIDUAL SPECIES

Queen conch: a gastropod with low to moderate productivity (currently included in CITES Appendix II)

Queen conch species have internal fertilization, but they deposit egg masses in soft bottom habitats. Queen conchs first reproduce at about 2-3 years. Individual fecundity is independent of age after maturity because somatic growth ceases at sexual maturity. Females lay 7-9 egg masses during the summer months, each with several hundred thousand eggs. Historical abundances (1970s-1980s) in unfished areas of about 1500-1800 conch per hectare have been reported. Today, in fished areas, density ranges from <0.1 to a maximum of about 50 conch per hectare. Some depleted areas that have been closed to fishing for many years (e.g., Florida) have shown little recovery suggesting that larval stages may be inhibited by slight water quality degradation. At a density of about 56 individuals per hectare, mating and spawning was not observed; above 56 animals, reproductive output increased until about 200 conch per hectare when it reached an asymptotic level (Stoner and Ray-Culp 2000).

The overall extent of decline is more important than the rate, and the percent reduction from a historical numbers of adults is most relevant, as depensation occurs below a critical population size and density. Conch populations could sustain about an 80% decline in abundance from unexploited levels before concern for the long-term viability of the population would be triggered. The density of adults will be the most relevant modifier of the decline criteria, as individuals would have difficulty in locating a mate at low density, which would negatively impact reproductive success. Habitat and water quality degradation may also increase vulnerability of this species.

Giant Clams (*Tridacna gigas*): bivalves characterized by low productivity (currently included in CITES Appendix II)

Giant clams are the largest bivalves in the world and may grow to 200 cm or more. Reproductive maturity is reached within a year for some small species, but requires many years for the larger *T. gigas*. Giant clams may be simultaneous hermaphro-

dites (although *T. gigas* and a few others are reported to be sequential hermaphrodites, with the largest individuals being female), but individuals do not self fertilize, and spawning is dependent on the presence of other individuals. Spawning is triggered by a chemical released in the water by a clam; thus, in areas where the density is too low, spawning does not occur.

The number of individuals is most likely the most important parameter in estimating the effect of the extent of decline, but the absolute abundance will vary depending on the species. For instance, the largest clams (*T. gigas*) are reported at densities of 5-100 individuals per hectare, whereas a small clam (*T. crocea*) can be found at up to about 1500 individuals per hectare. These species could have probably sustained a decline to 30% of historical (unexploited) levels. The smaller species could sustain a much larger percentage decline than the larger species, partly because they are naturally more abundant, and they reach maturity more rapidly (i.e., they have higher productivity). The most important modifier of decline criteria is density, as these species broadcast their gametes and a critical number of individuals per unit area (or clumped distribution) is also necessary. Other vulnerability factors affecting giant clams are degree of fragmentation of populations (they have been extirpated in many areas, and recovery depends on recruitment from distant reefs), rapid environmental change (e.g., increasing sea water temperatures increase the likelihood of bleaching and subsequent mortality), and selectivity of removal (the preferred individuals for food are the largest animals which may be primarily female). The recent rate of decline would be most relevant for localized populations that were still dense enough to sustain reproduction. In most locations, an ongoing decline is likely to occur for the species, as their densities may already be too low for recovery.

Sea Cucumbers - moderate to low productivity echinoderms

Sea cucumbers (holothurians) reproduce sexually or asexually. Sexes are separate and they broadcast their gametes into the water column for external fertilization. Holothurians are susceptible to over-exploitation due to their limited mobility, late maturity, density-dependent reproduction, and low rates of recruitment. Also, even though they have a wide distribution, most species have very specific

habitat preferences, including a specific zone within reef habitats, algae, or grassbeds, and some species have been found to exhibit restricted gene flow among populations, due to relatively limited dispersal of larvae.

The effect of commercial fisheries on sea cucumber populations has been examined in a few south Pacific locations. For instance, average densities of one of the most valuable species, *Holothuria nobilis* in the Torres Straits, Papua New Guinea (PNG) ranged from 9.4-18.4 individuals per hectare in the late 1980s, with maximum reported densities of 275 individuals/ha (Preston 1993). Peak catches occurred in 1991-1992, and subsequently declined. As sites were serially depleted, fishing effort shifted to more distant locations, until the fishery was closed. Surveys conducted in PNG several years after closure indicated little recovery; both adults and the recruiting year class were notably absent. On the Great Barrier Reef in Australia, densities of *H. nobilis* were found to be four to five times higher on reefs protected from fishing when compared to 16 reefs open to fishing. In addition, the average weight of individuals was substantially smaller (1763 g) on fished reefs than on unfished reefs (2200 g). The reduced densities combined with lower average weights resulted in a reduction of biomass by more than 75% on fished reefs. In fished populations, the combination of lower density, lower average weight, smaller number of mature individuals and reduced likelihood of fertilization was thought to greatly reduce output of larvae in fished populations (Uthicke and Benzie 2001).

The number of mature individuals, biomass and area of distribution are the most important parameters related to the extent of decline. The species could probably sustain a decline to 30% of their pre-exploited stock size; however, there are few data available to quantify this baseline. The modifiers of decline criteria appropriate for sea cucumbers include density (the primary factor), distance of larval dispersal, and habitat loss or fragmentation (some species are found in very specific habitats and these are vulnerable to degradation from land-based pollutants and sedimentation).

(RELATIVELY) HIGH PRODUCTIVITY, CLONAL, SESSILE SPECIES

Branching Bryozoan

While there is no known international trade in these species at the present time, there are numerous efforts to discover and develop potential pharmaceutical and other natural products from marine invertebrates. One of the promising candidates, an anti-cancer agent nearing the clinical trial stage, is derived from the bryozoan *Bugula neritina*. The number of colonies or percentage cover would be appropriate abundance parameters. However, the "decline to 5% of baseline" criterion would likely be overruled by concerns regarding population density (in order to maintain effective fertilization) because it is sessile, and concerns regarding maintenance of genetic diversity because it is clonal with substantial reproduction by fragmentation. Unfortunately, information regarding minimum density and genetic diversity requirements in such a species is extremely limited and a precautionary approach is required. There might be additional concern regarding depletion of coastal reef habitat.

LOW-PRODUCTIVITY CLONAL, SESSILE SPECIES

Scleractinian coral (currently included in CITES Appendix II)

Scleractinian corals are generally slow-growing (e.g., one to a few cm per year), long-lived (up to centuries, or at the level of the genet, potentially immortal) species with generally low rates of sexual recruitment. Appropriate abundance parameters are particularly difficult to find. Declines in percentage cover or in the number of colonies are likely the most applicable as this is most often how coral abundance is surveyed. While there are few solid data with which to quantify productivity parameters (e.g., generation time, r , and spawning potential ratio) for corals, a "decline to 30% of baseline" is probably a reasonable starting point. However, because of clonal life histories and sessile lifestyles, density is an important consideration to ensure potential for sexual reproduction. In addition to density, a modifier of decline criteria appropriate for corals is size. In most clonal corals, size and not age is the most important determinant of sexual maturity. Once a colony achieves a certain minimum

size, it becomes sexually mature. However, that same colony may regress in size due to partial mortality, to a level below which it is incapable of sexual reproduction. For species with a dominance of fragmenting reproductive modes, genetic diversity might be naturally low, making them more vulnerable to disturbance events, and loss of existing genetic diversity might be a more important consideration than decline *per se*. Because many coral species are extremely long-lived, the rate of recent decline is an important consideration if it is ongoing. Also, many coral reef habitats are subject to water quality degradation and other anthropogenic habitat disturbances. Thus, increased vulnerability due to habitat threats and, in the case of some species in the ornamental trade, endemism are also relevant modifiers. Other relevant modifiers include cases where the size structure of populations would increase concern, as many types of disturbance tend to reduce colony size.

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MARINE MAMMALS (many species currently included in CITES Appendix I or II)

Life history characteristics

Marine mammals comprise a wide range of taxa, including cetaceans, pinnipeds, sirenians, sea otters and polar bears. All of these are large bodied

animals with life histories typical of K-selected species. Maximum life spans are long, ranging from decades to over a century. Sexual maturity is delayed, and females usually produce single offspring at most annually. Intervals between reproduction greater than five years are known; for example, in social toothed whales such as killer whales and sperm whales. Offspring are cared for, receive substantial nutrition from mothers, and increase in size very rapidly.

Measuring populations

Direct estimates of numbers of animals in one or more population components are often possible. These are frequently based on direct counts of animals. For example, counts of pinniped pups in breeding areas and individual or groups of cetaceans during sighting surveys are often used. Estimates have also been based on rates of resighting of marked animals, especially where animals can be distinguished individually based on natural markings or on genetic analysis of tissue samples. The levels of precision vary greatly depending on the approach used, making determination of rates of change difficult from short time series of estimates.

Population reference levels have often been taken to be pre-commercial exploitation levels. These have been estimated based on population modeling techniques for some species, and in some coastal breeding species can be qualitatively estimated based on historical breeding habitat occupied.

Population productivity is generally constrained by the life history to relatively low levels. Maximum intrinsic rates of increase are typically less than 10% per year, and observed rates are rarely that high. Maximum rates are higher for pinnipeds than for cetaceans.

Alaska fur seal

Fur seals breeding on the Pribilof Islands of St. George and St. Paul were exploited very intensely over two periods, and reduced to very low levels by the early 1900s, the end of the first period. Population size is most easily measured by counts or estimates of numbers of pups on breeding beaches. The St. Paul Island breeding population recovered from pup production of less than 50,000 in 1911, to over 150,000 by 1922, to nearly 400,000 by mid-

century. The rate of increase from 1911 to 1922 was greater than 8% per annum, and the annual increments were remarkably consistent. The St. George population had a similar history, although with lower absolute numbers. The second period of harvesting was selective, focusing only on juvenile males from the 1920s until the mid-1900s, when an adult female harvest was also initiated. The population decreased markedly after the 1960s under this regime, and continued to decline after the cessation of all commercial harvesting. It is now at levels similar to those of the 1920s. The reasons for the post-harvest decline are not well understood, although entanglement of young animals in discarded fishing gear may have been a contributing factor.

Population size risk levels

The estimated pup production at the low point of population size in 1911 was roughly 12% of the peak pup production levels in the 1950s. The population was able to recover from this low level. The steady increase from 1911 to 1922 suggests that environmental fluctuations had little influence at low abundance levels.

Relevant modifying factors

Despite a large relative reduction in the size of the St. Paul population, the population size as measured in terms of numbers of pups born appears to have never been below several tens of thousands of individuals. Because females give birth annually after maturity at age 5 or 6, the number of pups produced is roughly equal to the breeding female population, suggesting that the total population size was more than twice the pup counts and, accounting for the males and immature females, probably more than three times that number. Thus, the total population size must have been about three times greater than the observed 50,000 pups.

Regarding selective removals, social structure and habitat loss, more breeding beaches were used historically than in the 20th century. These have not been recolonized with the recovery of the population. This may be related to social structure disruptions associated with the differential harvest of young males; for example, in reduced exploratory behavior associated with young animals.

North Atlantic humpback whale

The humpback whale in the North Atlantic breeds in the winter in two distinct areas, the West Indies and the Cape Verde Islands. Animals migrate across the central Atlantic to at least five summer feeding grounds across the North Atlantic, almost always occupying the same grounds as their mothers. Whales were harvested in ten to fifteen localized fisheries over their entire range from early in the 19th century, and continuing at very low levels today. Individual fisheries tended to be short lived, likely due to local reductions in abundance or availability. Whales breeding in the West Indies have increased since at least 1980 at several percent per annum, but the spatial distribution on the breeding grounds appears to be different than it was historically, based on the location of historical breeding ground catches. Whales have been shown to be increasing in one feeding ground, but data are insufficient to determine this for others.

Population size risk levels

The relative extent of decrease of the two breeding populations can be inferred to have been large based on the long catch history; certainly below 50% of pre-exploitation levels and probably below 30%. The short-lived fisheries suggest reductions to uneconomic levels in many localized areas, especially as some of these fisheries were moved to other areas. The western North Atlantic population has apparently recovered substantially, and analyses are underway using population models to estimate recovery based on the catch history. The eastern North Atlantic population remains at relatively low levels even after several decades of protection. Some of the feeding aggregations were likely decreased to quite low levels, and even the one known to be increasing likely numbers less than 1000 animals.

Relevant modifying factors

The absolute numbers of humpbacks in the North Atlantic at present is on the order of 12,000 animals, with strong evidence of increase in at least one of the populations. The animals in one feeding area are increasing, but number less than 1000, while the numbers in another are roughly 300 and apparently not increasing. Estimates for other feeding areas are being developed.

The population structure in the North Atlantic is complex, with mixing of animals from the two breeding grounds on five feeding grounds. Animals on the breeding grounds are of primary concern, because they represent the breeding potential. However, the status of animals on the various feeding grounds must also be considered because of maternal site fidelity. Reductions of abundance in either the breeding grounds or any of the feeding grounds would increase risk of extinction.

Determination of the appropriate component for CITES consideration is difficult. The two breeding ground populations could be considered independently. Alternatively, the feeding ground populations could be considered independently, but some of these would contain a mixture of the two breeding groups. The relative reductions and the rates of increase are undoubtedly different among these components, making any classification problems more or less complicated depending on the approach taken.

SEA TURTLES (all currently included in CITES Appendix I)

There are seven species of sea turtles in the world and they are found in the Pacific, Indian and Atlantic Oceans and their bays, sounds and other coastal waters. Only Mexico supports either nesting or foraging of every species (NRC 1990). In some parts of the world, turtles and eggs are fully protected on nesting beaches or in foraging areas, but in other regions eggs, laying females and foraging animals are harvested (with varying levels of control). The life cycle starts with hatching from the eggs and nests and moving *en masse* into the ocean. While moving down the nesting beach and into the ocean, juvenile sea turtles are often subjected to high levels of natural mortality from crabs, raccoons, foxes, birds, coastal fish, sharks and other species. They move from coastal waters rapidly into the open ocean and again may be subjected to high levels of natural mortality. They spend several years in the open ocean, possibly ranging from about 3 to 15 years. Though few small turtles have been observed in sargassum, it is assumed, based on currents, that this is the primary pelagic developmental habitat for turtles in the Atlantic Ocean.

Upon achieving a certain size - about two feet in carapace length (except for leatherback turtles, which

may be larger, and hawksbill turtles, which may be smaller), most species return to coastal waters where they interact with coastal human activities. This life history stage is referred to as the beginning of the coastal benthic stage and they remain in these waters as juveniles through the adult stage. Of seven species of sea turtles, only the Olive ridley remains pelagic during its lifetime and only the adult females range into coastal waters when they return to the beach to nest. Most species are tropical or subtropical in their range, but leatherback, loggerhead and Kemp's ridley turtles normally range to temperate waters. Because leatherback turtles demonstrate some degree of endothermy, they range into colder waters more than any other species. For example, in the western North Atlantic Ocean, leatherback turtles are found seasonally in Canadian waters. Leatherback turtles have also been recorded in Canadian waters in the eastern North Pacific Ocean. In the coastal benthic life history stage, they demonstrate omnivory (loggerhead turtle, Kemp's and Olive ridleys), herbivory (green turtles) and very specialized feeding regimes (sponges for hawksbills, and jellyfish for leatherback turtles). Habitat changes and loss or degradation of sea grass beds can have a direct impact on the survivorship of green turtles.

All sea turtle species are listed as endangered or threatened under the U.S. Endangered Species Act because of their reduced population sizes compared to historical levels. The number of females nesting on a beach is a common index of population levels and, for most species, this is the only metric available to evaluate stock status over any period of time. The age of first nesting may range from 7 years (e.g., ridleys) to as much as 20-30 years (e.g., green and loggerhead turtles). Aging of sea turtles is difficult and can only be achieved by tagging turtles as hatchlings. Currently, age and growth are inferred from data that tend to be incomplete over all life history stages and thus must be viewed with caution. Generation time is estimated to be about 10 years for some species (ridleys) and may be as long as 30 to 40 years for other species. While it is assumed that sea turtles live many years, data are insufficient over any length of time to verify this at present. The longest turtles have been maintained in captivity is about 30 years. Adult female ridleys return to nest on natal beaches. Kemp's ridley only nests along the beaches of Mexico in the Gulf of Mexico. Loggerhead turtles and green turtles also appear to return to natal beaches to nest. Leatherback turtles may shift nesting beaches depending on beach availability.

Genetic analyses indicate that while all turtles within a species look similar, some nesting populations may demonstrate sufficient reproductive isolation to be functional subpopulations (Bowen and Witzell 1995). Thus, the loss of subpopulations may have an impact on genetic diversity. Nesting beach requirements can be specific as well. Some species nest mainly on island beaches and some only on mainland beaches. For example, hawksbill turtles require beaches with vegetation on the dune and nest mainly on island beaches. This habitat specificity in the face of possible declines in suitable nesting beaches suggests that this may be more of an issue in the future as coastal habitat is developed worldwide. For most species, females nest every 2-4 years and lay about 100 eggs per clutch or nest, and may lay between 2-7 clutches per nesting year. Thus, fecundity can range from about 200 to 700 eggs every two to four years. Because nesting females can be from 7-30+ years of age, the number of females nesting may reflect the stock size and conditions from 7 to 30 years previous to the nesting year. This delay makes it very difficult to assess the current status of a stock and argues that extremely long time series must be evaluated to determine trends in nesting females. The recognition that the measured signal may represent the conditions turtles were subjected to many years previous to emergence as adults on the nesting beach has prompted many countries to implement surveys to index abundance of smaller and younger life history stages. There is a single study in Australia with a time series beginning in the 1960s (Hirth 1997). For other areas including the United States, the longest time series for these data is generally about 10 years and again, given the long developmental period for these species, much longer time series must be established before conclusions regarding trends in indices can be made (TEWG 1998, TEWG 2000).

Historical extent of decline

The lack of long time series for these species, especially as these time series do not come near to approaching even a single generation, makes it somewhat difficult to assess the historical extent of decline. The status and condition of these species focuses on the number of nesting females, as this is the easiest way to index population abundance. However, while for some coastal areas the recent (10 years or less) change has been measured, for many

species, this is not representative of the population or stock as a whole.

Kemp's ridley

Over the centuries, turtles have been valued as food or for other products (e.g., skin for boots, shells for ornaments and jewelry, and eggs as food or aphrodisiacs). The impact has been to reduce population levels (as indexed by number of nesting females) to very low levels. For example, the Kemp's ridley turtle, which likely numbered at least in the tens of thousands of nesting females at least through the 1940s, were harvested along with their eggs in Mexico until the mid 1970s when Mexico imposed strict protection on beaches and in coastal waters. The only historical estimate is from a film recorded in the 1940s showing a single day in the three-month long nesting season. The estimate of 40,000 nesting females resulted from extrapolations from this image. It is likely that this number is very low and that given nesting intervals of two years and a larger nesting area than that which was photographed, translates to a likely estimate of at least 100,000 adult females at this time. By the 1980s, the nesting population was reduced to hundreds of females or less than 1% of the historical nesting population (TEWG, 1998; TEWG, 2000).

Olive ridley

In the Pacific, there are reported to be very large nesting aggregations for this species particularly in Mexico where as many as 200,000 turtles have been reported nesting in a single year. The historical legal take of these animals has been controlled by the setting of quotas. This level of nesting has been reported for many years and at least into the 1970s. Two additional nesting aggregations occur in Costa Rica and together include another 500,000 turtles nesting per year. In the Atlantic, while the nesting populations are relatively small, they numbered as much as 2000 or so females per year along the coast of Suriname alone (Reichert 1993, NMFS / USFWS 1997, Salazar *et al.* 2000).

Leatherback turtle

The leatherback turtle demonstrates the greatest range in nesting. In the 1980s, the total number of turtles nesting along the Pacific coast of Mexico was estimated to be about 2000 turtles. How this com-

compared to numbers previous to this time is not known. In Indonesia in 1984 there were between 2000-3000 females reported as nesting and in recent years this has declined to 2 females in 1994 (Spotila *et al.* 1996). In Malaysia, there were reportedly thousands of turtles nesting in the late 1960s. In the Atlantic, the largest leatherback rookeries along the northern South American coast was relatively stable from the 1960s through the 1980s at about 10,000 females per year (NMFS/SEFSC 2001).

Hawksbill turtle

The hawksbill turtle was common on the eastern Pacific rim as recently as 50 years ago (NMFS/USFWS 1998). Prized for its shell, it has been exploited for hundreds of years, and heavily exploited for trade up to 1992. In the Atlantic, there is no information on historical levels except that they occurred in sufficiently high numbers to support the making of jewelry and other artifacts from the shell and this occurred for many years in Japan. In the Atlantic, most nesting populations appear to be continuing to decline. Important exceptions include Mexico and Mona Island, P.R. which represent about 25% of the total nesting in the Atlantic basin (Garduno-Andrade 1999, Garduno-Andrade *et al.* 1999). Estimates of nesting females tend to be lacking for this species, as it prefers to nest on remote islands and in small numbers at each site.

Loggerhead turtle

Nesting loggerhead turtles are found only in the western Pacific and in Japan and Australia. From the mid 1970s through the 1980s, the number of females nesting in Australia declined from about 3500 turtles to less than 1000 turtles for a total decline of about 75% over this period. While there are about 2000-3000 turtles nesting in Japan, these numbers as well appear to be declining but the rate is not known. Historical estimates for the loggerhead turtle in the western Atlantic are from the 1970s through the 1980s and showed declines over this period, although the rate is not known (TEWG 1998, TEWG 2000).

Green sea turtle

Australia continues to be the largest nesting area for the green turtle in the Pacific and the current nesting distribution is the result of heavy fishing over

the past centuries (Hirth 1997). While some nesting beaches have been stable or increasing over the past few years, many have demonstrated significant declines as well to point of extirpation or near extirpation. In the Atlantic, again as a result of aggressive protection in some areas, there have been some recent increases in numbers of nesting females (Tortuguero, Costa Rica as an example) while in other areas, nesting has been extirpated (Bermuda). The fishing logs from the vessels in the Caribbean through the early 1900s suggest that green sea turtles were abundant throughout the region. Jackson (1997) quotes the log from Columbus' 2nd voyage in 1494, which states: "But in those twenty leagues, they saw very many more, for the sea was thick with them, and they were of the very largest, so numerous that it seemed that the ships would run aground on them and were as if bathing in them". Valued for food as large turtles, they were harvested through the early 1900s and heavily fished throughout their ranges.

Recent rate of decline

Kemp's ridley

With the imposition of protective measures on the beaches and through the use of devices which allow the release of turtles from some fishing gear, it appears that since the mid 1980s the trend in numbers of nesting females has been increasing and there are now about 1500-2000 nesting females. While the trend over the past 15 years or so has been increasing (10-12% increase in number of females nesting per year), this number is small compared to the historical levels dating back to the 1940s. The current level is likely no more than about 1% of the historical level (TEWG 1998, TEWG 2000).

Olive ridley

Over the past 10 years, the number of nesting females in Mexico has reportedly been increasing but the rate is not known. In Costa Rica on the other hand, the number of females nesting annually has reportedly been declining, again at an unknown rate. Over the past 10 years, the numbers of female turtles nesting along the coast of Suriname in the Atlantic has declined to about 100 females from the historical estimate of 2000 females for a decline over this period of about 95% (Reichert 1993).

Leatherback turtle

Along the eastern Pacific coast, the numbers of turtles since the mid 1980s have been reduced from about 2000 nesting per year to about 100, or a decline over this period of about 95%. Over the same period, the number of females nesting in Indonesia has been reduced to about half of the 2000-3000 estimated in the mid 1980s. By 1994 the number of females nesting in Malaysia declined from thousands in the 1960s to 2 females, a decline of more than 99% over about 30 years (Spotila *et al.* 1996). In the Atlantic, the largest rookery along the northern South American coast, which once supported about 10,000 females per year, had shown declines in nesting females of about 15% per year. On the U.S. Florida and Caribbean beaches, which have not been considered a large rookery for this species, for the past 10 years, the numbers of females nesting has been increasing at about 8-12% per year (NMFS/SEFSC 2001).

Hawksbill turtle

Trend information for hawksbill turtles in Australia suggest that this stock has been declining at about 3% per year for the past 10 years. Trends in some areas of the Atlantic appear to be positive, but have not been well-quantified. This species has been noted as not commonly seen in the eastern Pacific (Meylan and Donnelly 1999).

Loggerhead turtles

The rate of decline in the western Pacific has continued over the past 10 years and it is now estimated that no more than 500 females nest annually in Australia which is a decline of about 50% over this period. For the past 10 years, in the Atlantic and along the U.S. southeast coast, loggerhead turtle nesting numbers have been relatively stable in Florida but declining at about 3-4% per year from Georgia to Virginia (TEWG 1998, TEWG 2000).

Green sea turtle

Recent changes in nesting numbers are not well known. In some areas, declines continue while in other areas, there appear to be some increases in numbers of nesting females. Any trend at this time is difficult to substantiate (Hirth 1997).

Relevant modifying factors

The effect of reduction to such small numbers on the genetic diversity of the Kemp's ridley and other species is not known. While for some nesting aggregations the trend in number of nesting females is upward, current levels for some species may only be a very small percentage of the historical level. A similar scenario of harvest of eggs and interactions of large juveniles and adults with fishing gear appears to have caused significant declines in Pacific leatherback nesting populations to the extent that thousands of nesting turtles have been reduced to tens of turtles or to single turtles nesting in any year. Current population levels are a small portion of the presumed very large numbers nesting prior to this heavy exploitation. The proportion of reduction from historical levels has not been determined as the baseline information is often not available. However, it is likely given the level of harvest and the life history strategies of turtles that make them especially vulnerable to depletion, that these reductions have been very large for some species and that current levels may be as low as 1% of historical levels. Conversely, there may be nesting populations that have not been well reported or recorded which could increase these numbers though it is likely that it would not result in large increases in current estimates. Thus, even the current baseline in abundance for some species is not known with confidence.

The life history of turtles, which includes relatively low fecundity compared to finfish, long generation times, and specific habitat requirements, coupled with limited time series of data of less than even one generation, lack of age and growth information, and populations that have been depressed to very low levels continues to equate to concerns about recovery for these species. Most species tend to be very specific in their food requirements; e.g., the green turtle is an herbivore and preferentially feeds in seagrass beds. The hawksbill turtle appears to be spongivorous and targets specific sponge species. The leatherback turtle feeds on jellyfish, coelenterates and tunicates. This specificity of resources adds to its vulnerability as habitats are degraded or change. As a result, throughout the world, there have been protected areas established for sea turtles; for example, the hawksbill turtle foraging grounds of the U.S. Caribbean Sea (e.g., Puerto Rico). The approach of protecting nesting beaches and foraging grounds by Mexico, coupled with the implementation

of restrictions on take either as harvest or accidental takes from fishing throughout its range, appears to have reversed the decline of the Kemp's ridley turtle. This experience suggests that aggressive protective approaches may be able to reverse declines in these species worldwide.

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B. OTHERS

STURGEONS

The Order Acipenseriformes (sturgeons and paddlefish) is an ancient order of fishes, comprising 23 species distributed across temperate freshwater, estuarine and marine ecosystems. These species are either restricted to freshwater systems (e.g., American paddlefish and lake sturgeon) or are anadromous (e.g., beluga and shortnose sturgeon). The order is generally characterized by long life-spans, low fecundity, and late maturation and thus most Acipenseriformes are vulnerable to overexploitation. Indeed, the valuable roe (for caviar), high catchability, and large sizes of many species resulted in lucrative, intense, but short-lived industries that collapsed at the beginning of the 20th century from

recruitment overfishing. Recent estimates of F_{MSY} for most mature sturgeons and paddlefish are only 0.03 - 0.05, and lower still for fisheries that exploit both adults and sub-adults. Historical fishing mortality rates in many commercial sturgeon fisheries are an order of magnitude higher than this.

Historical extent of decline

Although historical abundance estimates are difficult to calculate for most sturgeon species, experts believe that habitat loss (through damming, water removal and pollution) combined with chronic overfishing and bycatch has resulted in current populations that are a small fraction of pre-exploitation biomass. Estimates vary by species, but the following values have been calculated in recent studies:

Species	Decline
<i>Acipenser oxyrhincus</i>	99% from 1850 to present (based on landings; ASMFC 1998)
<i>Acipenser gueldenstaedtii</i>	95% from 1977 - 1999 (Caspian Sea, based on landings; CITES 2000)
<i>Huso huso</i>	92% from 1961 - 1997 (Volga River spawning population estimates; Khodorevskaya <i>et al.</i> 1997)

Given these levels and the taxon's low productivity, the Working Group's 70% decline benchmark (i.e., 30% B_0) is a relevant criterion that indicates that the Acipenseriformes are in fact appropriate species for CITES protection. As indicated by recent analyses, sturgeons and paddlefish populations that are at or below 30% B_0 are in need of timely and effective rebuilding plans that are augmented by international trade controls provided by an Appendix II listing. In fact, as indicated below, uplisting to Appendix I may be indicated for certain species that continue to experience overfishing and habitat degradation.

Recent rate of decline

The marked decline of sturgeon populations around the world is locally exacerbated by continued overfishing, rampant poaching, and declining habitat quality. This is particularly apparent in the case of the Caspian species of Beluga (*Huso huso*). Beluga sturgeon continue to exhibit poor recruitment, and commercial catches wane in the presence of increasing effort and demand. Recent information suggests that

Beluga populations are continuing to decline, falling 75% from 1991-1997 (Khodorevskaya *et al.* 1997). This situation indicates that current harvest and trade controls are insufficient for rebuilding *Huso huso* in the wild, and stronger measures are warranted. The Animals Committee, range countries, and other CITES Parties are currently investigating domestic management regimes for the species, and export quotas have recently been set to zero by the CITES Significant Trade Review process (www.cites.org, 2002).

Relevant modifying factors and additional considerations

In addition to historical population collapses and continuing declines in many Acipenseriform populations, reviewers should consider several exacerbating factors when determining the appropriate CITES listing for various species. These include truncated age structure, skewed sex ratios (from directed fishing on females), recruitment failure (from recruitment overfishing and poor habitat), selective removal of females, habitat specificity (e.g., anadromy and

spawning substrate), long generation times, and bycatch losses (i.e., undocumented mortality).

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PLANTS

The traditional focus of CITES on animals has created difficulty in applying criteria and guidelines to the life history characteristics of plants. Life history characteristics of plants that are atypical of most animals include:

1. Trophic level – most plants do not rely on prey
2. High fecundity; plants mostly have *r*-selected reproductive strategies (with some notable exceptions)
3. Many can reproduce vegetatively
4. Absolute population size is generally large
5. Species with long-lived seeds and juveniles can persist in the absence of adults
6. No social structure – for many species, however, individuals are self-incompatible
7. Sessile; dispersal occurs during the seed stage
8. Some species create their own habitat, or even monospecific stands
9. Most plants are symbiotic; they rely on specialized fungus (mycorrhizae) on their roots

Historical extent of decline (relevant variables)

1. Human consumption and market value may generate overharvest
2. Exotic diseases, pests and invasive competitors
3. Habitat destruction and disturbance
4. Climate change

Compared to plants, animals may be more vulnerable with respect to resilience and risk of extinction, due to low productivity, compromised social interactions, and the need for suitable habitat distributed over large areas. For fish, empirical and theoretical evidence (see Appendix I of this document) suggests that a reduction to 5 - 30% of the historical baseline abundance should trigger concern about the long-term viability of the species. Because of their greater resilience, the trigger for many plants may be less than 5–30%. However, such a decline in a plant species may have dramatic effects on other plants or animals that rely on it for food or habitat. This may generate a negative feedback that decreases the resilience of the plant species due, for example, to a loss of pollinators or seed dispersers.

Recent rate of decline

With respect to “severe” habitat loss and plants (other than rare or highly valued species), the rate of loss is important because high rates suggest that other (human) factors exist that may soon eliminate required habitat (e.g., logging or conversion to agriculture). Being sessile, rapid rates of loss may be more of a concern for plants than animals.

The recent rate of decline in abundance of plant species is highly variable for different parts of the world. For species associated with certain ecosystems, the rate of decline has been very rapid (e.g., North American prairie and tropical forests) and it may be inappropriate to wait for a 70 – 95% reduction in abundance before triggering concern and consideration for a CITES listing.

Relevant modifying factors

There are few empirical examples for plant species whose population size was reduced and then either rebounded or became extinct. However, many U.S. prairie species have persisted despite extreme reductions in habitat (although many have also become extinct). Likewise, island endemics have become extinct due to grazing pressure from introduced herbivores (e.g., palms on Indian Ocean islands). Furthermore, primary succession may take centuries before recolonization is complete (e.g., Krakatoa or Mount St. Helens volcanoes). Even reforestation can take decades (e.g., New England). In summary, plants may persist at very low abundances, but they require a long time to recover.

TREES

The world's tree flora is estimated to total around 100,000 species, nearly nine percent of which are documented as globally threatened with extinction, according to the 1994 IUCN criteria (Oldfield *et al.* 1998).

Historical extent of decline

Noss *et al.* (1995) conducted a literature review and summarized the extent of decline for numerous ecosystems in which trees play a dominant role. The following examples represent a partial list from their review. Although the original sources are cited, the full references are not repeated here, and readers should refer to Noss *et al.* (1995) for further details.

50 United States

- 85% of original primary (virgin) forest destroyed by late 1980s (Postel and Ryan 1991)
- 90% loss of ancient (old-growth) forests (World Resources Institute 1992)
- 12% loss of forested wetlands from 1940 to 1980 (Abernethy and Turner 1987)

48 Conterminous States

- ca. 95-98% of virgin forests destroyed by 1990 (estimated from map in Findley 1990 and commonly estimated by other authors, e.g., Postel and Ryan 1991)
- 99% loss of primary (virgin) eastern deciduous forest (Allen and Jackson 1992)
- >70% loss of riparian forests since pre-settlement time (Brinson *et al.* 1981)
- 23% loss of riparian forest since the 1950s (Abernethy and Turner 1987)

International Comparisons

Worldwide

- 76% of original primary forest worldwide destroyed by late 1980s (Postel and Ryan 1991 from various sources)
- ca. 50% of original area of tropical forest worldwide destroyed (Postel and Ryan 1991 from various sources)

- 55% of original coastal temperate rainforest worldwide logged (Kellogg 1992)

North and Central America and Caribbean

- 41% of original coastal temperate rainforest in North America logged (Kellogg 1992)
- 48% of original primary forest destroyed in Canada by late 1980s (Postel and Ryan 1991 from various sources)
- 60% of old-growth forests in Canada lost to logging (World Resources Institute 1992)
- 57% of original coastal temperate rainforest in British Columbia logged (Kellogg 1992)
- >90% of southern Mexico's rainforest destroyed (Ross 1992)
- 45% of Mexico's remaining forest significantly disturbed (The Nature Conservancy 1986)
- 60% loss of primary forest in Guatemala (The Nature Conservancy 1989b)
- 98% loss of dry forest in western central America (Jordan 1987; McLarney 1989)
- Virtually all dry forest in West Indies destroyed (Ray 1992)
- >99% of original forest in Puerto Rico destroyed by 1900, although coffee plantations covering 9% of island contained remnant individual dominant trees (Brash 1987; Weaver 1989)
- 75% loss of primary forest in Jamaica (The Nature Conservancy 1989c)

South America

- 37% of original primary forest destroyed in Brazil by late 1980s (Postel and Ryan 1991 from various sources)
- 10% of Brazilian Amazon forests destroyed (Ryan 1992)
- 98.5% of Brazilian Atlantic coastal forests destroyed (McNeely *et al.* 1990)
- 40% of original primary forest destroyed in Peru by late 1980s (Postel and Ryan 1991 from various sources)
- 29% of original primary forest destroyed in Venezuela by late 1980s (Postel and Ryan 1991 from various sources)
- 74% of original primary forest destroyed in Columbia by late 1980s (Postel and Ryan 1991 from various sources)

- ca. 50% of mangroves cleared in Ecuador (Ryan 1992)
- 58% of original coastal temperate rainforests in Chile and Argentina logged (Kellogg 1992)

Africa and Madagascar

- 65% of original wildlife habitat lost in Africa south of the Sahara (IUN/UNEP 1986a)
- 44% of original primary forest destroyed in Zaire by late 1980s (Postel and Ryan 1991 from various sources)
- 70-80% of original forest, savannah, and wetlands in Nigeria destroyed (World Resources Institute 1992)
- 70% loss of mangrove forests in Mozambique over last 20 years (World Resources Institute 1992)
- >90% of natural vegetation of Madagascar destroyed (Raven 1986)
- 75% loss of forests in Madagascar (World Resources Institute 1992)

Europe

- >99% of original primary forest destroyed in Europe by late 1980s. (Postel and Ryan 1991 from various sources)
- >99% of original coastal temperate rainforest logged (Kellogg 1992)
- 99.2% of the Caledonian forest of Scotland cleared (Watson 1992)

Asia

- 67% of original wildlife habitat lost in tropical Asia (IUCN/UNEP 1986b)
- 94% of original vegetation of Bangladesh destroyed (WRI, IUN, UNP 1992)
- 58% of original primary forest destroyed in Papua New Guinea by late 1980s (Postel and Ryan 1991 from various sources)
- 57% of original primary forest destroyed in Indonesia by late 1980s (Postel and Ryan 1991 from various sources)
- >75% of mangrove forests destroyed in India, Pakistan, and Thailand (Ryan 1992)
- 99% of original primary forest destroyed in China by late 1980s (Postel and Ryan 1991 from various sources)

Australia and New Zealand

- 95% of original primary forest destroyed in Australia by late 1980s (Postel and Ryan 1991 from various sources)
- 15% of original coastal temperate rainforest in Australia logged (Kellogg 1992)
- 76% of original primary forest destroyed in New Zealand by late 1980s (Postel and Ryan 1991 from various sources)
- 72% of original coastal temperate rainforest in New Zealand logged (Kellogg 1992)

Recent rate of decline

To qualify as globally threatened based on population reduction, the population of a species should have an observed, estimated, inferred or suspected reduction of at least 20 percent over the last 10 years or three generations, according to the 1994 IUCN criteria. Generation time is the average age of parents in the population. Defining the generation time of a tree species is very difficult given that the capability of reproduction in tree species varies widely according to the age/size class of individuals. For evaluation purposes, the guidelines for trees suggest that where no information is available, the generation time should be taken as 50 years for most tree species, 10-20 for pioneer species or small trees, 100 years or more for slow-growing trees.

Following this estimation, very many tree species potentially fall within at least the IUCN Vulnerable category given rates of deforestation, and therefore implied population decline, or the rates of exploitation of mature timber trees, over the past 150 years (Oldfield *et al.* 1998). In particular, over 250 species of dipterocarp of southeast Asia are believed to have lost over 80 percent of their forest habitat over the last three generations and are therefore listed as Critically Endangered (Oldfield *et al.* 1998).

ORCHIDS

The Orchidaceae is among the largest families of flowering plants, with an estimated 20,000 species or more. Orchids grow in all terrestrial ecosystems except the poles and extremely dry deserts, but their greatest diversity is found in the tropics. They can grow on the ground on many soil types, on rocks, or as epiphytes. There are also well-documented cases of

fully subterranean orchids (IUCN/SSC Orchid Specialist Group 1996).

Despite the fact that they are perennial, orchids in general have a short life expectancy because: 1) they inhabit niches in successional flux, and 2) they are subject to numerous catastrophic events. They are characterized by a relatively large investment in reproduction, with large inflorescence, large, attractive flowers, and/or large numbers of small seeds that can be dispersed over long distances. Their strategy appears to be to colonize new, temporary patches of habitat quickly (IUCN/SSC Orchid Specialist Group 1996). In one of the earlier North American studies documenting population changes, Curtis and Greene (1953) referred to the changes in orchid populations as “explosions” which in many cases were followed by extinction.

Geographical distribution, habitat specificity, and population size provide a basis for estimating the relative rarity of orchids and other plant species. Many known orchids would qualify as rare by one or more of these criteria. For instance, most of the several hundred species of the neotropical genus *Lepanthes* have restricted geographical distributions and are obligate inhabitants of montane cloud forests. In the Antilles most *Lepanthes* species are found on only one island, and 52 of the 60 species recorded in Mexico are restricted to a single mountain range, although their populations usually include large numbers of individuals. A good example of extreme rarity is provided by *Phragmipedium xerophyticum*. A careful search in the only known locality turned up only seven plants (IUCN/SSC Orchid Specialist Group 1996).

Historical extent of decline

Habitat alteration, including total destruction, modification, and fragmentation, threatens orchids, especially in the tropics where diversity is the greatest. Estimates of the extent of deforestation for various parts of the world are identified above in the tree section.

Most tropical orchids are found exclusively in primary forests that are largely undisturbed, although a lesser number of species thrive in marginal or disturbed sites, such as forest edges or “gaps”. Species belonging to the latter group, including twig epiphytes, are more tolerant of modification and fragmentation of the original forest and are, in fact, favored by disturbance, successfully colonizing secondary or introduced vegetation (e.g., citrus and coffee plantations).

However, populations of many of the less-tolerant taxa associated with mature forests usually decline as a result of disturbance, apparently being unable to cope with the increased isolation and reduced relative humidity, among many other potentially adverse factors (IUCN/SSC Orchid Specialist Group 1996).

Data suggest that there are more orchid species than those expected in certain managed areas. The absence of extinct orchid species in densely populated countries such as Mexico is also surprising. This could mean that orchid species have mechanisms that enable them to survive at very low densities or with very small population sizes. However, studies in this area are not available (IUCN/SSC Orchid Specialist Group 1996).

Recent rate of decline

Removal of unsustainable levels of plants from wild populations for trade is a major cause of the decline of many showy orchids. Well known examples of taxa impacted by collection from the wild are the slipper orchids in the genus *Paphiopedilum*, which have long been a major target of orchid collectors in southeast Asia. Many of the species of this genus have been overcollected, and populations have been extirpated even from protected areas (IUCN/SSC Orchid Specialist Group 1996). Most *Paphiopedilum* species are naturally rare due to restricted geographical distributions and narrow habitat preferences. Cribb (1987) estimates that 25 of the 60 species recognized are seriously endangered in the wild. A similar situation prevails in species of the genus *Phragmipedium* distributed throughout the American tropics (IUCN/SSC Orchid Specialist Group 1996).

A well-documented case of an orchid threatened by collection is *Laelia speciosa*, a Mexican endemic restricted to the southern limits of the Central Plateau. Although its geographical distribution is relatively extensive, populations are mostly local and strictly associated with a very specific habitat. Every year, many thousands of plants are collected in flower for sale within Mexico. A demographic study showed that in a population that is heavily harvested every year, recruitment of new individuals is non-existent, and the population will apparently become extirpated as the remaining old plants die (Hernandez 1992).

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APPENDICES

APPENDIX I. Relationship between Threshold Extent of Decline, Resilience and Productivity

The concept that thresholds for the extent or rate of decline should be inversely related to productivity may be a novel idea with respect to CITES (or IUCN) criteria and guidelines, although the use of generations times in the current guidelines suggests that there was appreciation of the implications of vast differences in productivity between species and populations.

In fact, rather than productivity, the demographic variable of greatest relevance to the risk of extinction is probably population resilience, which can be defined as the “ability to rebound after perturbation” (Holling 1973). The problem with the concept of resilience is that it is not an operational concept. There is no reliable way of measuring the ability to rebound, except empirically. However, for populations and subpopulations that have been reduced to low levels and recovered, the sample size is rarely greater than $N=1$ which is hardly sufficient to infer that a stock will rebound again if reduced to a similar level in the future (although some fisheries seem to be attempting replication). In addition, extinction is not a repeatable experiment, and so whatever is learned from isolated extinctions may provide little information relevant to the risk of extinction of other species.

Due to the lack of operationality of the concept of resilience, the Working Group used population productivity as a measurable proxy for resilience. Textbook definitions of productivity include:

Ricker (1975): “the total elaboration of new body substance in a stock in a unit of time, irrespective of whether or not it survives to the end of that time.”

Krebs (1972): “the amount of energy (or material) formed by an individual, population or community in a specific time period.”

Odum (1971): “gross primary productivity is the total rate of photosynthesis, including the organic matter used up in respiration” ... “net primary productivity is the rate of storage of organic matter used up in plant tissues in excess of the respiratory utilization by the plants” ... “secondary productivity is the rate of energy storage at consumer levels” ... “The total energy flow at heterotrophic levels which is analogous

to total production of autotrophs should be designated as assimilation, not production.”

Productivity at heterotrophic levels is more akin to the net productivity of autotrophs.

“Productivity = rate of production”

“Standing biomass or standing crop present at any given time should not be confused with productivity.”

Thus, productivity is a complex function of fecundity, growth rates, survival rates, age of maturity, and longevity. More productive species tend to have high fecundity, rapid growth rates, and high turnover of generations. It follows that species with high natural mortality must generally be more productive because they must produce energy or matter at a higher rate to compensate. The question is, “Is species or population productivity positively correlated with resilience?” On average, species with high fecundity, rapid growth and high turnover of generations will have greater ability to rebound from low numbers because they can quickly take advantage of conditions suitable for re-establishment or re-colonization. But such species also tend to have higher variability and, therefore, greater risk that population numbers may fluctuate to dangerously low levels, even in the absence of continued exploitation. In addition, species with high turnover of generations tend to have relatively few mature age classes, which means that recruitment failure is more critical. On the other hand, for the same reduction in percentage of the unexploited level, a long-lived species will have a more seriously truncated age distribution than will a short-lived species and, if egg viability, larval survival and related factors increase with maternal age, the ability of such species to rebound or to sustain further exploitation may be seriously compromised. Further, there are several examples of long-lived marine species with very high recruitment variability (e.g., sporadic exceptionally large year classes with most other year classes being well below the level required for population replacement; Atlantic redfish, Pacific bocaccio and Atlantic ocean quahog are examples).

These considerations must be borne in mind when adopting the working hypothesis (assumption) that population productivity is an operational proxy for population resilience. Musick (1999) also made this assumption. In his view, the intrinsic rate of natural

increase (r) is the real key to resilience because it incorporates all of the other components of productivity. He further noted that late-maturing, long-lived animals have low intrinsic rates of increase and, therefore, very low resilience to extraordinary mortality. However, there are many species for which there are insufficient data to create life tables, surplus production models, or other methods of estimating r . In these cases, a simpler proxy for productivity may be desirable. Since high productivity species are often characterized by low age at maturity and low maximum life spans, or more generally, shorter generation time, some function of the inverse of generation time or maximum life span may provide a rough index of productivity (as in the example provided in **Appendix III** of this document).

Another widely-used indicator of the risk of extinction in fisheries is the fishing mortality corresponding to the slope at the origin of a stock-recruitment relationship (i.e., the extinction threshold, called F_τ by Mace 1994 and F_{crash} by ICES 1997). Using a simple age-structured population dynamics model, Mace (1994) showed that F_τ increased with increasing natural mortality and with individual growth rates, both of which are positively related to productivity. F_{MSY} (the fishing mortality rate that results in maximum sustainable yield, MSY), also increased with increasing natural mortality and growth rates. Punt (2000) also showed that both F_τ and F_{MSY} increased with increasing population productivity. In Mace's (1994) studies, F_{MSY} was usually well below F_τ , ranging from about 16% F_τ to 43% F_τ over all tested parameter combinations for Beverton-Holt stock-recruitment relationships, and 38% F_τ to 48% F_τ for Ricker stock-recruitment relationships. Both Mace (1994) and Punt (2000) found that the ratio F_{MSY} / F_τ is a decreasing function of stock productivity (i.e., as productivity increases, F_{MSY} and F_τ become relatively closer together). (Note that this statement about agreement between the two studies is based on the results presented in Punt's tables, not statements in the text which appear to say the opposite of the results in the table). Punt (2000) also found that incorporation of depensatory effects substantially increased the ratio F_{MSY} / F_τ (1.0 being the maximum value), meaning that depensation causes the optimum and extinction fishing mortalities to be close together.

Mace (1994) also found that the ratio of B_{MSY} (the average biomass associated with MSY) to B_0 (the unexploited biomass) declined, but only very slightly, over the range of natural mortalities and growth rates considered ($M=0.1-0.3$ and $K=0.1-0.3$, respectively). More importantly, F_τ and F_{MSY} both increased

substantially with increasing slope at the origin of the stock-recruitment relationship, while $\%B_{\text{MSY}}/B_0$ exhibited a pronounced negative relationship with the slope at the origin. This implies that stocks with higher productivity, as indexed by either high natural mortality, high growth rates, or high slope at the origin of a stock-recruitment relationship, can sustain higher harvest rates at lower relative biomass, and that the harvest rate corresponding to the extinction threshold also increases with productivity.

Development of biological reference points defining overfishing thresholds

The development of biological reference points (BRPs) for use as fishing *targets* has a long history, and many useful reference points have been produced (e.g., MSY, F_{MSY} , and fishing mortality rates associated with spawning biomass per recruit levels of about 30-50% of the maximum possible level). More recently, the importance of specifying overfishing *thresholds* or *limits* has been recognized. These thresholds or limits are usually expressed in terms of a fishing mortality rate (F) that is so high that it will ultimately drive a stock to unacceptably low biomass levels, or a biomass (B) that is unacceptably low, or both. One of the most commonly assumed thresholds is 20% of the unexploited biomass; i.e., 20% B_0 (Beddington and Cooke 1983 and many others). The theoretical or empirical basis for 20% B_0 as a threshold is not particularly strong but it has become generally accepted, at least for temperate-water teleosts of "average" productivity (e.g., many gadoids), as a biomass level that it is best not to venture below. The main justification is that, based on empirical observations, average temperate-water teleosts appear to be able to rebuild from levels above 20% B_0 , but below this level there is much greater uncertainty. The main problem with allowing fish stocks to become too low is a "fear of depensation", wherein a depensatory process is defined as one that has a proportionately greater effect as biomass decreases (predation is often used as an example of a potentially depensatory process). Thresholds higher or lower than 20% B_0 have sometimes been employed in risk analyses, with lower values being used for species believed to have high resilience or high productivity, and higher values for species with low productivity or potentially low resilience (e.g., Quinn *et al.* 1990).

Myers *et al.* (1994) examined eight methods for estimating minimum spawning stock biomass thresholds corresponding to the level at which recruitment to a fish stock is seriously reduced. They

considered three classes of thresholds defined by: (1) the stock size corresponding to 50% of the maximum predicted average recruitment; (2) the minimum stock size that would produce a good year class when environmental conditions are favorable; and (3) the stock size corresponding to 20% of various estimates of virgin stock size. The authors concluded that estimators of the first type are generally preferable because they are easily understood, relatively robust if only data at low stock sizes are available, and almost always result in higher levels of recruitment above the threshold. For this class of estimators, the estimates of the threshold spawning stock biomass defining recruitment overfishing for the 64 iteroparous stocks examined were generally in the range of 2-25% B_0 (excluding extreme outliers with biologically impossible values). By far the majority of these estimates fell below 20% B_0 . However, there are no data to evaluate the validity of any of these thresholds as cutoffs delineating increased risk of depensation.

In fact, depensation is difficult to demonstrate analytically. Myers *et al.* (1995) examined data sets from 129 fish stocks for evidence of depensation. Of these, 29 had an estimated statistical power over 0.95 and, of the 29, only three appeared to have significant levels of depensation. However, in a reanalysis of these data using a different approach (a hierarchic Bayesian meta-analysis rather than a likelihood ratio test), Liermann and Hilborn (1997) demonstrated that distributions of a parameter associated with depensation had broad tails extending well into the depensatory range. They concluded that there is a significant amount of uncertainty about whether depensation exists and to what degree, and suggested that not allowing for the possibility of depensation in fish population dynamics is a poor assumption.

Over the last 20 years, an important new class of reference points associated with overfishing thresholds has been developed based on percentiles of survival ratios estimated from stock-recruitment (S-R) observations. This work began with Shepherd (1982) who showed how a standard spawning per recruit (SPR) analysis could be combined with S-R observations to generate reference fishing mortality rates. The relationship between the two types of information is straightforward (Gabriel *et al.* 1989; Mace and Sissenwine 1993): for any constant F , there is a corresponding SPR level that can be inverted and used as the slope of a straight line through the origin of the S-R data. Points along the line represent the average survival ratio (R/S) required to support that particular constant F . Percentiles of observed survival ratios can therefore be used to define threshold and target levels

of F , which can then be translated back to the SPR scale and expressed as a percentage of the maximum (which occurs at $F=0$). Gabriel *et al.* (1989) provide a lucid description of the methodology including computational details.

Two percentiles that have been advocated as reference points for overfishing thresholds are the 90th percentile (denoted F_{high} ; Shepherd 1982) and the median (denoted F_{med} ; Sissenwine and Shepherd 1987). Both are intended as indicators of *recruitment overfishing*, defined by Sissenwine and Shepherd (1987) as occurring when fishing mortality rates are so high that a stock is unable to replace itself on average. The tangent through the origin of an S-R relationship corresponds to $F_{extinction}$ (also referred to as F_{τ} by Mace and Sissenwine 1993 and Mace 1994, and F_{crash} by ICES 1997 and Punt 2000). F_{high} may overestimate the tangent since the highest survival ratios may simply reflect anomalously favorable environmental conditions, not the ability of the population to sustain fishing under average environmental conditions. On the other hand, F_{med} may underestimate the slope if the data exhibit compensation (concavity). It is more correct to use F_{med} as an estimate of F_{rep} ($F_{replacement}$), the fishing mortality rate corresponding to the observed average survival ratio. Thus, F_{rep} is the fishing mortality rate that, on average, allows for replacement of successive generations over the observed range of S-R data. F_{rep} may be a valid approximation to the slope at the origin in the case where observations are restricted to low stock size, or where there is very little evidence of compensation in the relationship. However, F_{rep} may be a substantial underestimate of the extinction threshold if the stock has a history of light exploitation.

Mace and Sissenwine (1993) surveyed 83 well-studied European and North American fish and invertebrate stocks with sufficient data to construct stock-recruitment plots and conduct yield per recruit and spawning per recruit analyses to obtain estimates of various BRPs including F_{med} and the corresponding %SPR (hereafter referred to as the threshold %SPR). The estimated threshold %SPR ranged from 2.0% to 65.4%, with a mean of 18.7%. Although the range is large, this is to be expected because F_{med} and the corresponding threshold %SPR are dependent on the range of data available for analysis. If the data are restricted to low stock size, F_{med} may provide a reasonable estimate of $F_{extinction}$; otherwise it will be a conservative estimate.

Despite the large range, there were clear and statistically significant differences among the species

groups considered. Of the fish species, Atlantic cod had the smallest mean threshold %SPR (6.8%, N=14), suggesting that it has relatively the greatest resilience to fishing. Mean values for other groups were: other gadoids 25.7% (N=23), non-Baltic Atlantic herring 17.4% (N=9), other clupeids 37.5% (N=10), flatfish 14.5% (N=19), Perciformes 22.4% (N=6), and sea scallops 2.5% (N=2). Multiple linear regression analyses indicated that, of the life history parameters considered as independent variables (natural mortality, average body weight at 50% maturity, the maximum average body weight, and the maximum observed spawning biomass), the maximum average body weight was the most important.

However, it was believed to be unlikely that the correlation with body weight could be extended across all taxa involved in commercial fisheries. First, the two stocks of sea scallops were clearly "outliers" in the regression, having relatively very low values of threshold %SPR (i.e., high resilience). Second, two stocks of swordfish (which have much larger individual size than any of the other species considered) were clearly separated from the rest of the species included in the regression. Third, the remaining species included in the analysis represent a relatively homogeneous group compared to the huge diversity of life histories exhibited by marine species. For example, elasmobranchs (which were not included in the analysis) have large body size but generally do not exhibit high resilience to fishing and would probably have moderate or high values of threshold %SPR. The authors believed that rather than a relationship with body size *per se*, "body size is probably a proxy for fecundity, which may be more strongly tied to resilience and therefore to [the threshold %SPR]".

Based on these analyses, Mace and Sissenwine (1993) advocated use of 20% SPR (approximately the mean of values calculated) as a recruitment overfishing threshold for stocks believed to have average resilience, and 30% SPR (approximately the 80th percentile of the values calculated) for little-known or low resilience stocks, with the disclaimer that "the 80th percentile of the observations included in this survey will be an overly-conservative threshold for most stocks". Goodyear (1990, 1993) also concluded that %SPR levels below about 20% should be avoided except in cases of strong density-dependence in the stock. Gabriel *et al.* (1989) calculated an overfishing threshold of 20% SPR for Georges Bank haddock. Similar values were obtained for Canadian gadoid stocks (Maguire and Mace 1993).

Between about 1990 and 1998, U.S. Fishery Management Councils were required to develop definitions of overfishing based on thresholds signifying recruitment overfishing. (Since 1998, more stringent definitions have been required). In the majority of the 117 cases reviewed by Rosenberg *et al.* (1994), definitions of recruitment overfishing were based on spawning biomass per recruit analysis. Of the cases using this methodology, about half defined recruitment overfishing as occurring at a spawning biomass per recruit of 20% of the maximum, where the maximum corresponds to the situation for an unfished stock. Almost one-third defined recruitment overfishing as occurring at a spawning biomass per recruit of 30% of the maximum. The primary reason for such consistency is that most of the overfishing definitions were set by analogy with results for well-studied ("data-rich") stocks and theoretical analyses presented in papers by Goodyear (1980, 1990, 1993), Gabriel *et al.* (1989), Clark (1991), and Mace and Sissenwine (1993). In the Rosenberg *et al.* (1994) review, values as high as 40% and as low as 10% were also used. Subsequent studies have calculated threshold values even lower than 5%. For example, Ennis and Fogarty (1997) calculated a threshold value of 2.5% for American lobsters. Myers and Barrowman (1995) conducted a meta-analysis based on the slopes at the origin of a Ricker stock-recruitment curve. Since many of the stocks included in their analysis were the same as those on Mace and Sissenwine (1993), their results are similar. However, since they estimated the slope at the origin based on a particular type of stock-recruitment relationship rather than the median observed recruits per spawner, their estimates tend towards slightly lower percentages. For example, of the 85 estimates for iteroparous species 7 (8.2%) were below 2% SPR.

Unfortunately, the percent spawning biomass per recruit does not translate directly into the same percentage of the unfished biomass. The translation between %SPR and %B₀ depends on the shape of the stock-recruitment relationship (Appendix B in Mace 1994). In general, the correspondence will be best for relatively low (0-20%) threshold %SPR, but as the threshold %SPR increases, the corresponding %B₀ will increase at a slower rate. For example, in the cases tabulated in Mace (1994), Table B1, a threshold %SPR of 5% corresponds to a %B₀ of about 5-8%, a threshold %SPR of 20% corresponds to a %B₀ of about 14-17%, a threshold %SPR of 30% corresponds to a %B₀ of about 19-23%, and a threshold %SPR of 50% corresponds to a %B₀ of about 33%. Thus, even though threshold %SPR values as high as 40-60%

have been calculated, this translates into a substantially lower %B₀.

In summary, there seems to be strong consensus amongst fisheries scientists that 20% SPR and 20% B₀ are applicable overfishing thresholds for marine stocks with "average" productivity. Defining the extremes for species with very high or very low productivity may be more problematic but, for most of the recruitment overfishing studies based on spawning per recruit, the estimates have been based on a conservative estimate of the slope at the origin of a stock-recruitment relationship (viz. F_{med}, the median fishing mortality corresponding to the median survival ratio from observed stock-recruitment data; see Gabriel *et al.* 1989, Mace and Sissenwine 1993). The 5% level appears not to be overly risky for highly productive species given that replacement %SPR values as low as 2-3% have been calculated for some species (Mace and Sissenwine 1993, Ennis and Fogarty 1997), and that there are instances where fish stocks reduced by as much as 1/750 have exhibited recovery or are in the process of recovering (Musick 1999). In fact, Musick (1999) suggested 1% as a threshold for species with high productivity,¹ but the Working Group felt that waiting until a stock was reduced to 1% of its unexploited level could be risky. For low productivity species, threshold %SPR values as high as 40-60% have been calculated for some species but, depending on the stock-recruitment relationship, these values of %SPR will translate into considerably lower values of %B₀. Based on the translations from %SPR to %B₀ presented in Mace (1994), an upper bound of 30% B₀ seems reasonable. This is also the value suggested for very low productivity species by Musick (1999).¹

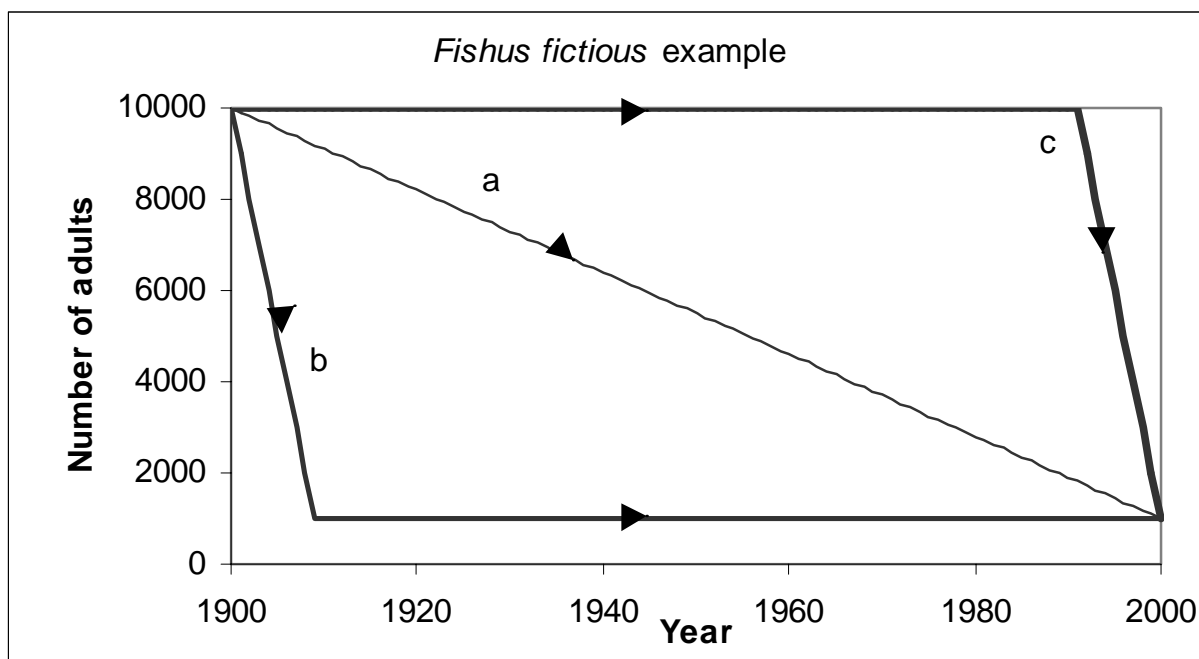
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¹ The decline thresholds suggested by Musick (1999) were intended to operate over the longer of 10 years or 3 generations and to designate a species or population as "vulnerable", which would then trigger consideration for further analysis and possible action.

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**APPENDIX II. Hypothetical Example Illustrating the Importance of
Considering the Historical Extent of Decline and the
Recent Rate of Decline in Conjunction with One Another**



Consider a hypothetical long-lived species, *Fishus fictious*, that could have exhibited three different patterns of decline over the past century: (a) a steady decline from a historical unfished state to 10% of the unfished level, (b) a rapid decline over a ten-year period beginning a hundred years ago to 10% of the unfished state, and fluctuating around this level ever since, and (c) a 90-year record of stable stock size with no fishing, followed by a rapid decline to 10% of the unfished state over the past 10 years. All three trajectories begin and end up at the

same place, but the three situations should not be treated equally. For (b), the rate of decline over the period 90-100 years ago is largely irrelevant; only the overall extent of decline now matters. Trajectory (c) is "worse" than either (a) or (b) for two fundamental reasons: higher uncertainty (there is greater uncertainty about the ability of the stock to persist near 10% of the unfished level), and higher risk (projecting recent trends into the future gives the worst scenario for this trajectory).

APPENDIX III. Graphs Combining Numeric Guidelines for the Historical Extent of Decline, Recent Rate of Decline, and Productivity

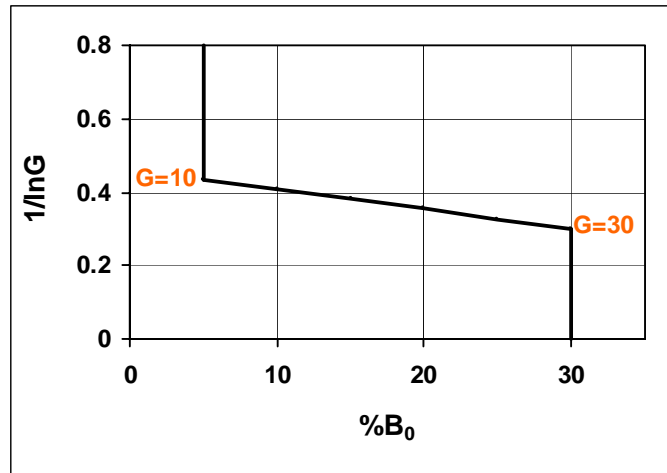


Figure 1. The threshold historical extent of decline (x-axis) expressed as a percentage of the baseline unexploited biomass (B_0) for various levels of productivity, where productivity is indexed by the inverse of the natural logarithm of generation time (y-axis). This figure indicates that, regardless of the recent rate of decline, for generation times of 10 years or less, concern for a population should be triggered when biomass falls below 5% B_0 ; for generation times of 30 years or more, concern for a population should be triggered when biomass falls below 30% B_0 ; and for generation times between these two extremes, the threshold extent of decline should be interpolated.

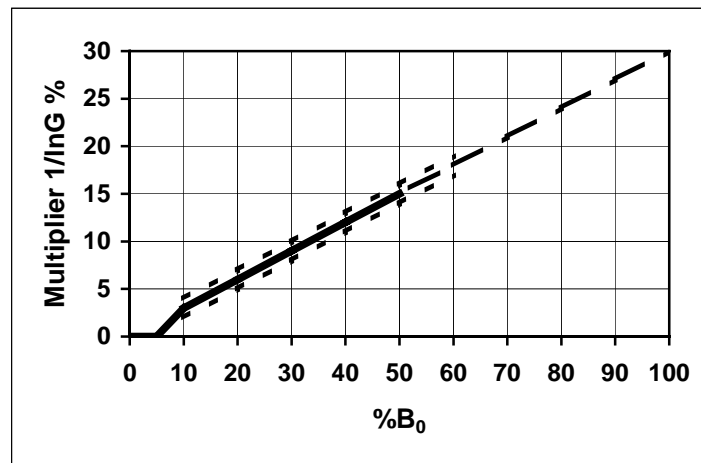


Figure 2. The threshold average annual rate of decline (expressed as a multiplier of the inverse of the natural logarithm of generation time) for various levels of historical extent of decline ($\%B_0$). The solid line bends towards the x-axis at low $\%B_0$, indicating threshold levels of depletion that are likely to be of concern regardless of the recent rate of decline. The dashed line indicates the range of extent of decline where one would not normally be concerned about the threat of extinction unless the average annual rate of decline was very rapid. The dotted lines indicate that the actual numeric values have uncertainty associated with them. Figure 1 overrides Figure 2.

APPENDIX IV. The Importance of an Historical Perspective

The following two hypothetical examples illustrate the importance of considering the longest possible time series, even in the case of short generation, rapid turnover (sardine-like) species.

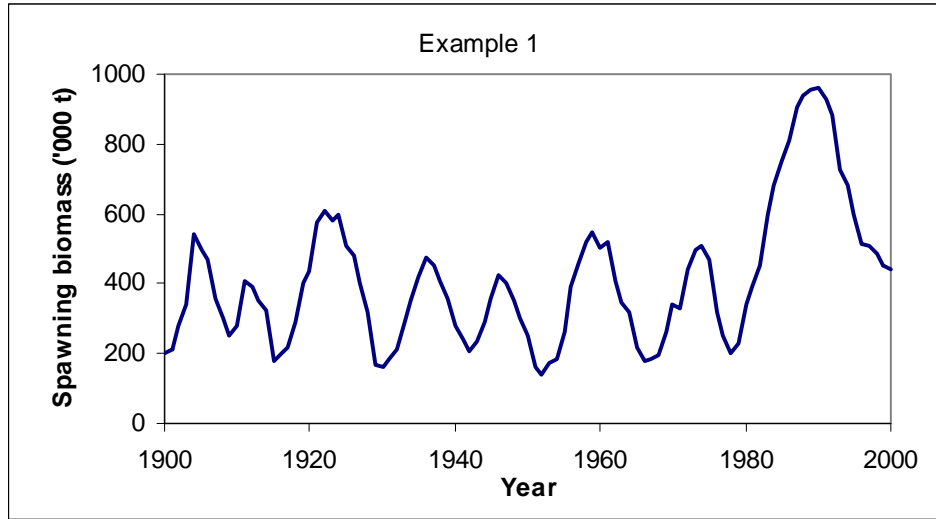


Figure 1. A highly-productive, short generation species (mean generation time of 3 years) has recently declined substantially (a 54% decline over the most recent ten years) but the most recent peak in population size was unusually high compared to the rest of the recorded history of the species and the stock is currently well above historical average levels.

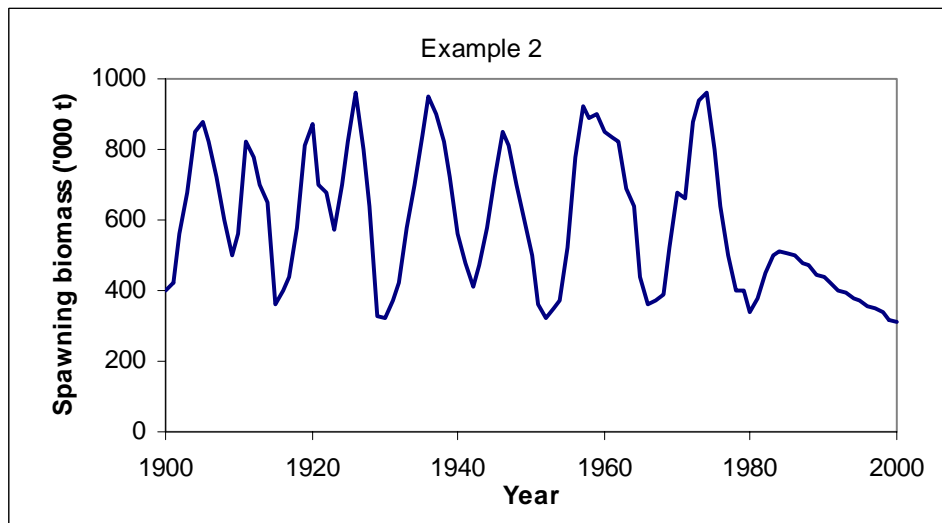
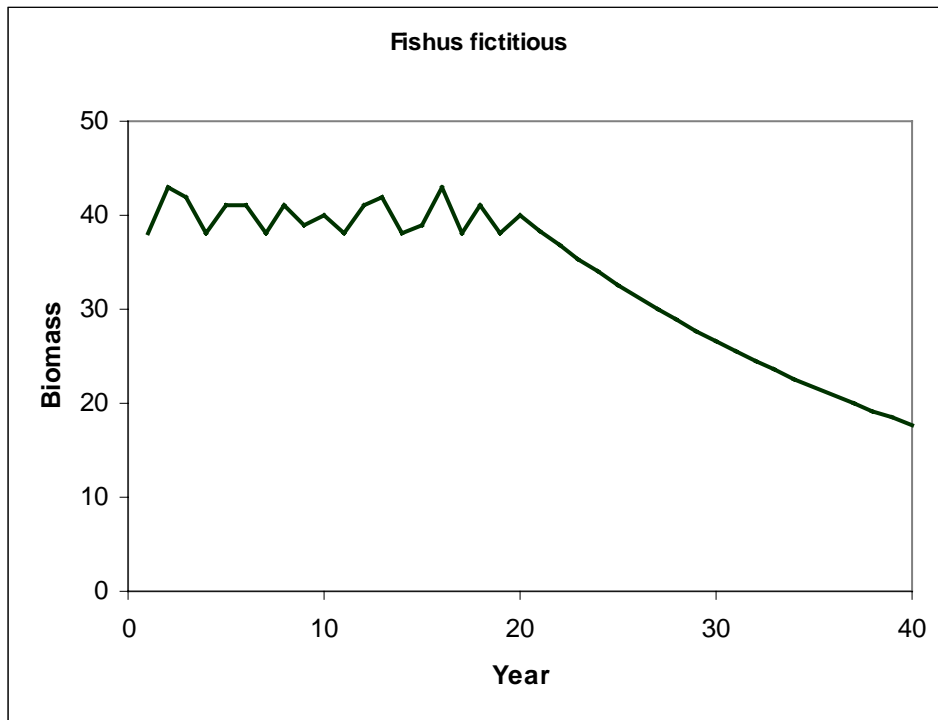


Figure 2. A highly-productive, short generation species (mean generation time of 3 years) has been declining at a relatively slow rate over the past two or three generations (actually a 39% decline over the most recent 16 years and less than this over 2-3 generations), but the most recent population peak was well below the historical average, and the stock is currently below the lowest point over the 100-year recorded history.

Appendix V. Another Problem with the Use of Generation Times as Time Horizons



Fishus fictitious has a mean generation time of about 30 years. It has been well-managed for several decades with biomass fluctuating around 40% of the baseline, unexploited size. Due to a sudden increase in demand, *F. fictitious* starts declining steadily at an average rate of about 4% per year.

If the trigger for concern is “a 50% decline in 10 years or three generations, whichever is longer”, this implies waiting at least until the cumulative decrease is 50%. In the current example, where there is a steady 4% decline per year, the “wait period” would not necessarily be three generations, but it would likely be at least 17 years. Shouldn’t an average annual decline of 4% per year for a low productivity species that is already at 40% of the baseline act as a trigger for concern much sooner than 17 years?

This problem can be moderated if estimated or inferred annual rates are projected into the future. In that case, use of generation time as a basis for the projection makes more sense because species with long generation times should not be allowed to decline as low. Thus, the trigger should be modified to read “an estimated or inferred annual rate of decline that is projected to lead to a cumulative 50% decline in 10 years or three generations, whichever is longer”. In the current example, this would likely result in concern being triggered as soon as it became apparent that the species was declining monotonically – likely much sooner than 17 years.

The above recommendation is not taken up in the main body of this report, because there it is overridden by the recommendation that the historical extent of decline and the recent rate of decline need to be considered in conjunction with one another.

APPENDIX VI. Explanations and Discussion of Selected Modifying Factors

This appendix provides explanation or discussion of several of the modifying factors listed in Section 4 of the main body of the report. There are two types of modifying factors: vulnerability factors that would increase concern about a given extent or rate of decline, and mitigating factors that would decrease concern about a given extent or rate of decline.

(i) Vulnerability factors that would increase concern

► Low absolute numbers or biomass

If absolute numbers or biomass of a population or subpopulation are low, this could override the decline criteria. For example, some species may exist at low levels even in an unexploited state. This modifier applies particularly to some endemic species, and to other species that could be considered to have been rare throughout their recorded history. The working group was unable to come up with a single number that could apply across all taxonomic groupings and recommends that experts in different taxonomic groupings derive specific numbers appropriate to that taxonomic group.

► Selectivity of removals

International trade in plants and wildlife is often conducted for commercial purposes, and in these instances it involves products for human consumption or enjoyment. Consequently, harvesters are often targeting specific types of individuals that exhibit special characteristics such as a unique biological product, color variety, or desirable size. This means that exploited populations often experience selective removals rather than indiscriminate harvest that equally affects both sexes and all age/size classes.

Such “artificial selection” can have profound effects on the subject population’s ability to reproduce, withstand environmental variabil-

ity, maintain genetic diversity, or retain consistent structures (e.g., primate social groups, whale pods, etc.). Although selective removal can differentially affect various populations of the same species (e.g., one color variety of reptile is more desirable than another in the pet trade), it probably has the most serious effects when it compromises reproductive potential (e.g., targeting mature adults, removing nestlings, etc.). Such scenarios can be exacerbated when non-targeted individuals are sacrificed to obtain the target product (e.g., discarding male sturgeons in caviar fisheries, killing parents to obtain offspring).

► Distorted age, size or stage structure of a population

A distorted age, size or stage structure in a population or sub-population may raise concern about that population or subpopulation sooner than indicated by the decline criteria. Examples of distorted age, size and stage structures include those where the structure is distorted towards young individuals due to prolonged high harvest rates, and those where the number or biomass of young individuals is sufficiently low that it is unlikely that they will be able to match or exceed the mature portion of the population in the near future. Such distortions may be caused by factors such as selective harvesting of one or more ages, sizes, or stages, or recruitment failure for one or several years due to harvesting or to environmental factors. Which of the three variables (age, size or stage) is considered will depend primarily on data availability and which of these metrics is the most applicable to the life history characteristics of the species of concern.

► Social structure, including sex ratio

Social structure of populations, and social interactions between individuals within a population, can influence reproduction and survivorship, and thus should be taken into consideration in evaluating vulnerability. Factors to consider include social hierarchies,

social dominance, highly skewed sex ratios, matriarchal social structures, etc.

➤ Low population density

Many marine species are broadcast spawners and external fertilization takes place in the water column. Because the volume of the ocean is large, dilution of gametes occurs rapidly in time and space. Hence, fertilization success can be a primary bottleneck in the life history of many marine species. Many mobile marine species have behavioral adaptations such as spawning aggregations to enhance fertilization. However, aggregative behaviors are limited for sedentary and impossible for sessile species. Hence, for sessile marine species or species of limited mobility, declines in population density may represent a more relevant scale of extinction risk (than abundance per se) by eliminating potential for sexual reproduction. Unfortunately, quantitative information on the relationship of density and reproductive success is likely limited for most species (exceptions include conch and some sea urchins).

➤ Specialized niche requirements

There will be some species that during some portion or all of their lifetimes will have very specific requirements that must be met to complete their life cycles. This could be very specific or unique habitat requirements that provide food resources or are necessary for successful reproduction. This special need could add to a species risk of extinction when subjected to removals. These types of specialized niche requirements could in fact be limiting to a population and maintain numbers at a fixed or constant level which could be small. For a species that has a long generation time and low fecundity otherwise identified as low productivity, any removals could depress population levels to very small numbers.

➤ Species associations such as symbiosis and other forms of co-dependency

When information is available on special interactions between different species (both the mechanism and the effects of the interaction), it should be taken into consideration in evaluating the vulnerability of the species. Vulnerability increases for species for which the interactions

are more obligatory and more species-specific. The effects on populations of these interactions can be complex.

- *Mutualism* occurs when two species live in close association with one another, to the benefit of both species. This is a positive reciprocal relationship, at the individual or population level, between two different species, in which both species benefit. The relationship enhances the survival, growth, reproduction, and fitness of both species' populations. Mutualism can be either symbiotic or non-symbiotic.
- *Symbiosis* occurs when individuals of two different species interact physically, in an obligatory relationship. The interaction can either benefit both species, or only one of the two species. Examples of mutual benefit include algae and fungi (forming lichens), mycorrhizae (plant roots and fungi), and coral anthozoans. Symbiosis can also occur in parasite/host relationships, where two species live in an obligatory association in which the parasite depends on the host.
- *Non-symbiotic mutualism* can be obligatory or facultative. Many plant/pollinator relationships are examples of non-symbiotic mutualism, and are vital to understanding the vulnerability of species to harvesting pressures. An example of obligatory non-symbiotic mutualism is the relationship between fig wasps and the figs they pollinate; such relationships increase the vulnerability of species to exploitation. An example of non-symbiotic facultative mutualism is the role of elephants as seed dispersers in Africa. Declines in elephant populations could also impact these plant species.

➤ Strong aggregating behavior

Strong aggregating behavior such as schooling in fish and herding in mammals may have benefits in terms of protection from predators and location of food, but species exhibiting such

behavior are likely to be more vulnerable to exploitation by humans, particularly humans using modern technologies (e.g., purse seine nets).

► Extensive migrations

Species that undertake extensive migrations probably have substantial energy requirements for surviving the migration, and also may be vulnerable to exploitation by the inhabitants of several different countries along their often-predictable migratory paths.

► Secondary ecosystem-based effects

Compromising the role of a species in its ecosystem creates a secondary effect that could threaten the survival of the species itself. For example, the severe decline of a plant species that negatively impacts the status of a species-specific pollinator may inhibit the ability of the plant to rebound.

► Uncertainty

In many cases there is great uncertainty about the status of a species or the true risk of extinction. This arises for several reasons; for example, (i) lack of data, (ii) data of questionable quality or validity, and (iii) extinction is not a repeatable experiment, and therefore the amount that is learned from isolated extinctions may not provide much information relevant to the risk of extinction in other species. In the face of uncertainty, it is prudent and precautionary to err on the side of species conservation.

(ii) Mitigating factors that would decrease concern

► Absolute numbers or biomass are high

If absolute numbers of the population or subpopulation in question are high, this could override the decline criteria. For example, some species may be so abundant that even 5% of the unexploited level represents a large number of individuals or a large biomass. However, by themselves, large numbers do not necessarily automatically imply low risk of extinction. If all individuals are contained in a

small number of aggregations (e.g., fish schools) that can be eliminated by a small number of fishing sets or large predators, the risk of extinction will be high. The working group was unable to come up with a single number that could apply across all taxonomic groupings and recommends that experts in different taxonomic groupings derive specific numbers appropriate to that taxonomic group.

► Existence of natural refugia

There are some species that by virtue of their life histories utilize habitats that are remote or inaccessible during a portion of their life cycle. Also, some species may be inaccessible through parts of their lifetimes through the establishment of managed protected areas. This protection could reduce the impact of removals on a species. For example, if the most vulnerable stage of a species is the most desired life history stage but for a significant portion of the species, this stage is protected through natural refugia, then removals of other stages may have a lesser impact than would be the case if the refugia did not exist. However, it is important to keep in mind that natural refugia often cease to be effective mechanisms for protection once appropriate technology to exploit them is developed (e.g., "rock hopper" trawls that can be used to fish for groundfish on previously untrawlable fishing grounds).

► Adaptations to small population size

Small population size, by itself, is a risk factor because of increased probability of extinction resulting from demographic stochasticity, and increased probability of loss of genetic variability. However, the risk associated with small population size should be considered greater in populations that have experienced declines than in populations that have been historically small. At the very least, historically small populations have demonstrated that they can persist at low abundance for extended periods of time. By virtue of their demonstrated persistence they should be considered at lower risk of extinction than populations that have not demonstrated this ability. Adaptations to small population size may include specialized mating strategies, or other physiological or behavioral responses that reduce variability in abundance.

► Selectivity of removals

Some types of selective removal of animals may act as mitigating factors with respect to the extent or rate of decline. For example, if a har-

vesting program were to focus mainly on post-reproductive individuals, it should have relatively little effect on the long-term viability of the population for most species. Similarly, in non-monogamous species, selective removals of males may act as a mitigating factor, up to a point.

APPENDIX VII. Relationship of CITES to the U.S. Endangered Species Act and the Marine Mammal Protection Act

The U.S. Endangered Species Act (ESA) serves both as the legislative authority for protection of endangered and threatened species in the United States and as the implementing authority for CITES. Thus, through the ESA, citizens of the United States are legally responsible for complying with both regimes. The ESA prohibits the import and export of endangered species.

Although many species are listed in both regimes, the Appendices of CITES and the lists of threatened and endangered species of the ESA are not identical. Inclusion in the regulatory framework of one does not necessitate inclusion in the other. Determination of species to be included in the ESA is made by the U.S. Fish and Wildlife Service and the National Marine Fisheries Service through a strict legal process known as a "rulemaking" (regulatory) procedure. The process, which can include several rounds of public hearings and comment periods in order to encourage the participation of all interested parties, including the general public, the scientific community, other government agencies, and foreign governments, can take over a year. All CITES listing decisions are begun through proposals by Party governments and must carry a two-thirds majority at Conferences of the Parties, which are convened about every 2½ years.

Listing in the Appendices of CITES and ESA are confused by many people. Although both regulate species according to their conservation status, this status is determined by different criteria. The basis for listing in the ESA is far more flexible than the CITES criteria. Under the ESA, the

following factors determine whether or not a species should be listed as endangered or threatened:

- the present or threatened destruction, modification, or curtailment of the species' habitat or range;
- overutilization for commercial, recreational, scientific, or educational purposes;
- disease or predation;
- the inadequacy of existing regulatory mechanisms; and
- other natural or manmade factors affecting the species' continued existence.

CITES and the Marine Mammal Protection Act

The U.S. Marine Mammal Protection Act (MMPA) established a moratorium, with certain exceptions, on the taking of marine mammals in U.S. waters and by U.S. citizens on the high seas, and on the importing of marine mammals and marine mammal products into the United States. According to the MMPA, "marine mammal" is defined as "any mammal which (A) is morphologically adapted to the marine environment (including sea otters and members of the orders Sirenia, Pinnipedia and Cetacea), or (B) primarily inhabits the marine environment (such as the polar bear); and, for the purposes of this Act, includes any part of any such marine mammal, including its raw, dressed, or dyed fur or skin." Thus, with very few exceptions, U.S. citizens are prohibited from engaging in international trade in marine mammals or their parts or products, regardless of their status in CITES.