CONVENTION ON INTERNATIONAL TRADE IN ENDANGERED SPECIES
OF WILD FAUNA AND FLORA

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IUCN/TRAFFIC ANALYSES OF THE PROPOSALS TO AMEND THE CITES APPENDICES AT THE 17TH MEETING OF THE CONFERENCE OF THE PARTIES

The attached document has been submitted by the Secretariat at the request of IUCN/TRAFFIC*, in relation to agenda item 88 on *Proposals to amend Appendices I and II*.

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INTRODUCTION

CITES (the Convention on International Trade in Endangered Species of Wild Fauna and Flora) was opened for signature in Washington DC on 3rd March 1973, and to date has 182[1] Parties from across the world. If CITES is to remain a credible instrument for conserving species affected by trade, the decisions of the Parties must be based on the best available scientific and technical information. Recognizing this, IUCN and TRAFFIC have undertaken technical reviews of the proposals to amend the CITES Appendices.

The Analyses - as these technical reviews are known - aim to provide as objective an assessment as possible of each amendment proposal against the requirements of the Convention as laid out in the listing criteria elaborated in *Resolution Conf. 9.24 (Rev. CoP16)* and other relevant Resolutions and Decisions. Unfortunately, due to a challenging fundraising environment and the high number of proposals, the funds necessary to conduct *the Analyses* and present them in the same manner as for previous Conference of the Parties (CoPs), were not found. Therefore, instead of producing the highly-detailed summary section and accompanying table for each proposal as we have done in previous years, for CoP17 we have instead produced a summary section for every proposal. The time available to research and consult experts has been reduced compared with previous Analyses, and is therefore not as exhaustive. In addition, to ensure *the Analyses* were completed in time to assist Parties with their decision making, the summary documents were made available online 10 weeks from the deadline for the submission of proposals as opposed to 12 weeks as in past years, in response to requests by Parties. To ensure *the Analyses* are as accessible as possible to Parties, we have sought to improve online dissemination through the creation of a bespoke webpage where *the Analyses* can be downloaded individually or in full (see http://citesanalyses.iucn.org/).

The Summary section presents a synthesis of available information taken from the Supporting Statement and other sources and, in a separate paragraph, a specific analysis of whether or not the proposal might be considered to meet the pertinent criteria in *Resolution Conf. 9.24 (Rev. CoP16)* or other relevant CITES Resolutions.

Information on the status and biology of species has been collected from IUCN's Species Survival Commission Specialist Group network and the broader scientific community, and used to evaluate the proposals and the information proponents provided against the CITES listing criteria. TRAFFIC has drawn on its own information sources on trade and expert networks. The resulting document brings together a broad range of expertise, which we are confident will be of assistance in the discussions of the proposals.

Following the deadline for Parties' submission of amendment proposals (27th April 2016), the Analyses team compiled available information to prepare a first draft of the summary section. Information compiled, together with a series of additional questions and clarifications were then sent to a variety of reviewers for comment if needed, particularly on the accuracy and reliability of information presented. **Reviewers were not sent the "Analysis" paragraphs, for which the IUCN/TRAFFIC Analyses Team takes full responsibly and which are only finalized at the end of the project.** Reviewers do not see, or have responsibility for, the analysis against the relevant criteria. Those named as "Reviewers of summary information" (previously listed as Reviewers) have only reviewed the information compiled by the Analyses team in the summary section for accuracy and gaps. The final text expressed in this publication does not necessarily reflect those of IUCN or TRAFFIC, nor the reviewers as a body.

To satisfy the needs of the Parties for information well before the CoP, *the Analyses* were completed and made available on the web on 6th July 2016. The Summary and Analysis paragraphs will be translated into French and Spanish and made available online as soon as possible. Printed versions of the Summary and Analyses paragraphs in all three languages will be made available at CoP17 in Johannesburg, South Africa.

[1] As of 20th June 2016 https://cites.org/eng/disc/parties/chronolo.php

These analyses aim to highlight relevant information on which the Parties can base their decisions, and are not to be considered exhaustive. There may be omissions and differences of interpretation in a document compiled on a wide range of species, particularly with such high number of proposals to consider within the allotted timeframe and under a tight budget. We have nevertheless tried to ensure that the document is factual and objective and consistent in how we have interpreted and applied the criteria across the range of taxa and proposals

A summary of the CITES listing criteria and the IUCN Red List Categories and Criteria is provided as an annex to the document. It should be emphasized that the numerical guidelines for the listing criteria in *Resolution Conf 9.24 (Rev. CoP16)*, Annex 5 are not thresholds and may not be appropriate for all species. Consequently, applying the criteria requires a certain amount of interpretation of them. Decision 15.29 (2010) invited IUCN and TRAFFIC to summarize their experience in applying criterion Annex 2 a B and the introductory text to Annex 2 a of Resolution Conf. 9.24 (Rev. CoP15) to commercially exploited aquatic species. The resulting paper can be found in AC25 Doc 10 Annex 3. TRAFFIC has also been commissioned by Environment Canada to prepare a paper on reviewing CITES proposals that involve projections of species trends in the future, for which there is limited guidance for interpreting the Listing Criteria, which will be submitted as an Information Document to CoP17.

References to source material are provided. In some cases, these sources have been consulted directly while in others, they have been cited by reviewers to support their statements. Where information is not referenced in the Summary paragraph, this is because it is presented in the Supporting Statement. The term ‘CITES trade data’ refers to data from CITES Annual Reports as provided by the Parties and managed by UNEP-WCMC on behalf of the CITES Secretariat. Where information has been provided from a particular country’s official trade statistics, this has been specified.

ACKNOWLEDGEMENTS AND CREDITS

Many individuals and institutions contributed to the review of the CITES amendment proposals and compilation of the present Analyses. Those to whom we would first like to extend our thanks are the reviewers of these proposals, many of them members of the IUCN Species Survival Commission (SSC) Specialist Groups, as well as the many other scientists and experts from other institutions who, although not formally linked with SSC, have volunteered their time and expertise to this process.

Dena Cator led on the fundraising, without which we would not have been able to undertake this work. We again acknowledge the generous support of all our funders in these economically difficult times.

The staff members of TRAFFIC, who assisted in the review of proposals, deserve recognition for the contribution they have made to this document.

We would also like to thank the French translation team of Daniele and Richard Devitre, and the Spanish translation team made up of Wendy Byrnes, Ana Bennett and Carlos Ibero.

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Contents

Delete Wood Bison *Bison bison athabascae* from Appendix II

Proponent: Canada

Summary: The Wood Bison *Bison bison athabascae* and the Plains Bison *B. b. bison* are the two recognised subspecies of the North American Bison *Bison bison. B. b. athabascae* is native to Canada and the USA where it has recently been reintroduced, having become extinct there. It has been introduced to the Russian Federation. Female bison typically give birth to one calf for the first time at three or four years; in some populations they reproduce annually. Generation time is estimated to be eight years.

Bison bison athabascae was listed in Appendix I in 1975 and transferred to Appendix II in 1997. *B. b. bison* is not listed in the Appendices.

Historically *B. b. athabascae* occurred over an estimated 2.5 million km^2, of which 80% was in Canada and the remainder in Alaska, USA. The current occupied area has been calculated to be around 121,000km^2 [2]. The geographic range is not considered to be reducing at present, but *B. b. athabascae* are generally prevented from expanding into areas where there may be increased risk of disease transmission, hybridization with *B. b. bison*, and conflict with other land uses (such as agricultural use).

Over-harvest by European settlers drove numbers down from an estimated 168,000 in northwest Canada in 1800 to one remnant herd of about 250 individuals by the early twentieth century. Intensive protection and active recovery efforts have resulted in an increase with recent estimates of a total population of ca. 9000 animals, of which ca. 5200 to 7200 were mature individuals, this representing a 47% increase in population since 2000[2]. This population is separated into nine (or possibly 10[1]) isolated herds in Canada. Six of the nine herds have fewer than 500 individuals: populations are considered viable in the long term if they exceed 1000 individuals[1]. In recent years one population has suffered severe mortality from disease and another from starvation during a severe winter[2].

Further increases to the total population size in Canada, or the addition of new wild subpopulations there, are not likely[2]. *B. b. athabascae* was recently re-introduced to the wild in the USA (Alaska) where the population numbered around 130 individuals in 2015. It has been also been introduced to Russia from Canada. In 2014 there were ca. 135,000 farmed *B. bison* in Canada of which 3000 were *B. b. athabascae* and approximately 51,000 were *B. b. athabascae* x *B. b. bison* hybrids.

Bison bison athabascae is currently listed on Schedule 1 of Canada's Species at Risk Act (SARA) as a Threatened species based on an assessment in 2000. However following a 2013 re-assessment, the subspecies may be downgraded to Special Concern. In Canada, *B. b. athabascae* is protected, meaning that hunting and other activities such as capture or harassment are controlled. Management and protection is independent of the listing in CITES. Individuals can be culled to limit the spread of disease, to prohibit hybridization or to manage human-bison conflict. In the USA, *B. b. athabascae* is listed as threatened throughout its range under the Endangered Species Act[3]. A federal rule from 2014 designates *B. b. athabascae* in Alaska as a nonessential experimental population. Hunting is not currently permitted but will likely occur in the future.

Some international trade in this subspecies is recorded in the CITES Trade Database. Canada reported exporting a number of commodities from wild *B. b. athabascae* between 2000 and 2014, including ca. 1540kg of meat, 11 bodies and 40 skins: the vast majority of which were exported as hunting trophies or for personal purposes. In the same period, less than 100 live *B. b. athabascae* were reported as exported to Russian Federation for the purposes of introduction into the wild, and three to the USA for commercial purposes. During the period, Canada reported exporting a small number of *B. b. athabascae* or hybrid bison that were declared as captive-bred, captive-born or ranched, including 51 live animals. Canada has no record of illegal export of wild *B. b. athabascae* in the past 15 years.

Bison bison athabascae has not been assessed by IUCN as a subspecies, but *B. bison* was classified as being Near Threatened in 2008.

Analysis: *Bison bison athabascae* was listed in Appendix I in 1975 and transferred to Appendix II in 1997. The population (currently ca. 9000) has increased in recent years although further increase is unlikely owing to constraints on available habitat. This population does not have a restricted range. Trade reported in the CITES Trade Database from 2000 to 2014 is at a very low level and there have been no reports of illegal trade. It would appear that harvest for international trade has a negligible impact on the subspecies, which would therefore not appear to meet the criteria for inclusion in Appendix II. It has not been subject to a recommendation under the provisions of the Review of Significant Trade within the last two intervals between meetings of the Conference of the Parties so that the precautionary measures in Annex 4 to *Res. Conf. 9.24 (Rev. COP16)* Annex 4 are met. Furthermore the current listing of *B. b. athabascae* in Appendix II while *B. b. bison* is outside the Appendices is inconsistent with recommendations for split-listing.

Reviewers of summary information only: R. Kramer.

References:

Information not referenced in the Summary section is from the Supporting Statement.

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[1] Gates, C. & Aune, K. (2008) *Bison bison*. The IUCN Red List of Threatened Species. Viewed on 17th June 2016.

[2] COSEWIC (2013) COSEWIC assessment and status report on the Plains Bison, *Bison bison bison*, and the Wood Bison, *Bison bison athabascae*, in Canada. Committee on the Status of Endangered Species in Canada. Ottawa. Xv – 109 pp.

[3] USFWS (2012) Endangered and Threatened Wildlife and Plants: Reclassifying the Wood Bison under the Endangered Species Act as Threatened throughout Its Range https://www.regulations.gov/document?D=FWS-R9-IA-2008-0123-0035. Viewed on 27th June 2016.

Inclusion of Western Tur *Capra caucasica* in Appendix II, with a zero quota for wild-taken *Capra caucasica caucasica* exported for commercial purposes or as hunting trophies

Proponents: Georgia and European Union

Summary: The Western Tur *Capra caucasica* is a wild goat endemic to the Caucasus Mountains of Azerbaijan, Georgia and the Russian Federation. The CITES standard nomenclature recognises three subspecies (*C. c. caucasica*, *C. c. cylindricornis*, and *C. c. severtzovi*). However, *C. c. severtzovi* is generally recognised as a synonym of *C. c. caucasica*. This treatment is followed here.

Capra caucasica mostly inhabits subalpine and alpine zones between altitudes of 800 and 4000m. Females mature at three to four years old and typically give birth to one kid annually, although some older females do not reproduce every year.

The population is estimated at 43,000 to 46,000 animals, most (possibly ca. 32,000) are in the Russian Federation, with 7000 to 8000 animals in Azerbaijan and 3000 to 5000 in Georgia. Most of the population (recognised as *C. c. cylindricornis*) is in the Eastern Caucasus where it is reported to have increased from the 1940s to the 1960s, reaching 25,000 to 30,000 animals before declining subsequently to an estimated 18,000 to 20,000 animals by the late 1980s and then increasing again.

The population in the Western Caucasus (recognised as *C. c. caucasica*) spans the border of Georgia and the Russian Federation, with a range likely to exceed 10,000km^2. In the late 1980s this population was estimated at 12,000 but declined, ascribed chiefly to illegal hunting, to perhaps 6000 to 10,000 in 2001, and 5000 to 6000 in 2004. In 2008 it was suspected that the number of mature individuals numbered 4000 to 6000 (and be decreasing), comprising around 1000 in Georgia and the remainder in the Russian Federation. However, another estimate indicates that the population in Georgia may be much smaller, perhaps 100 to150 individuals[1]. There is a zone of hybridisation between *C. c. caucasica* and *C. c. cylindricornis* in the central Caucasus.

Poaching has been identified as the most significant cause of decline of the population in the Western Caucasus, although livestock grazing and severe winters have also contributed[2].

Hunting regulations generally distinguish between subspecies. No hunting is permitted of *C. c. caucasica* in either range State. Hunting for *C. c. cylindricornis* is permitted in the Russian Federation under permit; ca. 300 to 320 permits are reportedly issued annually though only about half are used. Trophy hunting in the Russian Federation is reported to be undertaken mainly by foreign visitors[3]. Hunting for *C. c. cylindricornis* is permitted under licence in Azerbaijan; no information on annual quotas or the numbers of animals hunted there is currently available.

Hunting, including trophy hunting, of all *C. caucasica* is prohibited in Georgia, although it said to take place in northern Georgia, where it is part of the cultural heritage. Horns, meat and skin of harvested animals are either consumed or sold. The horns are typically made into drinking vessels for which there was reportedly high demand in Georgia and elsewhere in the 1960s to 1990s; there is apparently much less demand today[3]. Mounted heads and horns were also reportedly valued abroad in the 1990s[3].

There appears to be negligible international trade in *C. caucasica*[3] and no evidence that such trade has an impact on the species[3, 4].

The species may be affected by loss and degradation of habitat, severe winters, livestock grazing and disturbance from tourism.

Capra caucasica is found in several protected areas in its range States. *C. caucasica* was classified by IUCN as Endangered in 2008.

Analysis: *Capra caucasica* has an extensive range, and a relatively large and increasing population. There appears to be negligible international trade in the species. It would not appear to meet the criteria for inclusion in Appendix II.

Regarding the proposal for a zero quota for wild-taken *C. c. caucasica* exported for commercial purposes or as hunting trophies: there are no guidelines or criteria in *Res. Conf. 9.24 (Rev. CoP16)* for assessing such a proposal but as its impact would be similar to that of an Appendix-I listing it may be appropriate to assess it against the criteria for such a listing. The population does not have a restricted range but it does have a relatively small population which is reported to be declining and may meet the biological criteria for inclusion in Appendix I. No harvest of *C. c. caucasica* is permitted in either range State and there is no evidence that it is affected by trade. Furthermore, *Res. Conf. 9.24 (Rev. CoP16)* indicates that split-listing should be avoided in general and that when it does occur, this should generally be on the basis of national or regional populations, rather than subspecies. This same recommendation could be taken to apply to this case, where different annotations with different effects are proposed for the same species. This proposal does not adhere to that recommendation.

Reviewers of summary information only: S. Lovari, D. Mallon, P. Weinberg and K. Kecse-Nagy.

References:

Information not referenced in Summary section is from the Supporting Statement

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[1] Gurielidze, Z. (2016) *In litt*. to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
[2] Weinberg, P. (2008) *Capra caucasica*. The IUCN Red List of Threatened Species 2008.
[3] Weinberg, P. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
[4] Mallon, D. (2016) *In litt.* to IUCN/TRAFFIC Analyses Team, Cambridge, UK.

Amend the Appendix-II listing for *Vicugna vicugna*

For the exclusive purpose of allowing international trade in wool sheared from live vicuñas and in items made thereof, the following provisions shall apply:

> **- In addition to obtaining the CITES permit, any person or entity making products from vicuña wool must have a licence to use the country of origin mark. There are two marks:**
>
> **- For international trade in garments and cloth made from vicuña wool sheared from live animals, whether made inside or outside the country of origin, the "VICUÑA [country of origin]" mark must be used:**

VICUÑA[PAÍS DE ORIGEN]

For cloth, the selvages must bear the words "VICUÑA [country of origin]" or products made outside the country of origin, the name of the country where the product was processed or the garment was made must also be indicated.

> **- For international trade in handicrafts (artisanal processing) made in the country of origin from wool sheared from live vicuñas, the "VICUÑA [country of origin] – ARTESANÍA" mark must be used.**

VICUÑA[PAÍS DE ORIGEN]- ARTESANÍA

If processing takes place outside the country of origin, the name of the country where the product was processed or the garment was made must also be indicated.

> **- If articles are made from vicuña wool from several countries of origin, the countries from which the wool was obtained must be indicated, along with the percentage of wool from each country contained in the product.**
>
> **- All other specimens shall be deemed to be specimens of species listed in Appendix I and the trade in them shall be regulated accordingly.**

Proponent: Peru

Summary: The current Appendix-II listing for Vicuña is annotated as follows:

> *"Vicugna vicugna [Only the populations of Argentina[1] (the populations of the Provinces of Jujuy and Catamarca and the semi-captive populations of the Provinces of Jujuy, Salta, Catamarca, La Rioja and San Juan), Chile[2] (population of the Primera Región), Ecuador[3] (the whole population), Peru[4] (the whole population) and the Plurinational State of Bolivia[5] (the whole population); all other populations are included in Appendix I]"*

Each population has its own annotation, indicated by the numbered superscripts above, specifying the parts, purposes and labelling to be adhered to for that population. Current annotations vary marginally, allowing trade in wool sheared from live Vicuñas and items thereof under the condition that any cloth is marked with logo and indication of country of origin and that other items are labelled as artisanal products from the country of origin.

It appears that the proposed amendment is intended to replace all the current annotations 1-5 with a single annotation that would result in the following:

- For international trade in cloth or garments [from live Vicuña] processed within or outside the country of origin, the product must be marked with the logo and the indication of country of origin on the selvadge (edge of woven fabric finished so as to prevent unravelling, often in a narrow tape effect, different from the body of the fabric).

- Where processing has taken place outside the country of origin, a label must indicate the country where the transformation took place or the garment was made.

- For artisanal products, where these are produced outside the country of origin, as well as the label specifying “Vicuña [country of origin] – artisanal”, a label must indicate the country where the transformation took place or the garment was made.

- Where the product is made of wool from more than one country, it should indicate all those countries and the percentage of wool from each.

Currently, exported wool does not have to be marked; once processed outside countries of origin there is no labelling requirement for cloth or garments produced. It also appears that garments made from labelled cloth do not necessarily have to be labelled with the logo and the country of origin.

Analysis: Under the proposal, all items processed outside the country of origin would be expected to carry the above labels. This presumably applies both to those sold in the processing country and those re-exported. It is not clear that it is possible to enforce under the Convention a requirement that products for a domestic market be labelled in a particular way. It may in theory be possible to apply a labelling restriction to re-exports, essentially as a mechanism for ensuring that wool used was legally obtained in the first place (Under Article IV of the Convention (specifically paragraph 5 a) re-export of any product from a species included in Appendix II requires a re-export certificate which the Management Authority of the re-exporting State should only grant if they are satisfied that the specimen was imported into that State in accordance with the provisions of the Convention). There is an analogy with labelling of crocodilian skins, (*Res. Conf. 11.12 (Rev. CoP15): Universal tagging system for the identification of crocodilian skins*), although the latter contains recommendations rather than prescriptions for allowing trade.

Replacing the five separate annotations would remove any differences between the current annotations.

Transfer of African populations of African Lion *Panthera leo* from Appendix II to Appendix I

Proponent: Chad, Côte d'Ivoire, Gabon, Guinea, Mali, Mauritania, Niger, Nigeria and Togo

Summary: The African Lion *Panthera leo* is the second-largest cat species, found in sub-Saharan Africa and India; it was formerly also present in North Africa and the Middle East. In Africa it is known to be extant in 25 (possibly 26) range States and has been recently reintroduced to a 27th (Rwanda). It is possibly extinct in five others. Current area of occupancy is estimated at around 1.7 million km^2, this representing a small portion of its presumed historic range[1].

Panthera leo are absent from tropical moist forest and hyper-arid desert but otherwise have wide habitat tolerance; their optimum habitat is open woodland and thick bush, scrub and grass complexes. They are social, living in prides with an average size of four to six adults[1]. Population density is closely linked to seasonal prey availability and varies widely, from 1.5 adults per 100 km^2 in semi-desert (in South Africa) to 55 adults per 100 km^2 in prey-rich savannah (the Serengeti in East Africa)[1]. Generation time is estimated at seven years. Average litter size is 2.5, with an interbirth interval of around 20 months if some of the previous litter survive to maturity, four to five months if not[2]. The species is present in a large number of protected areas, both fenced and unfenced, operating under a range of management regimes.

The major factors adversely affecting *P. leo* populations are killing (often pre-emptive) in defence of human life and livestock, habitat loss, and declines in the prey base. Where not appropriately managed, trophy hunting may have an adverse effect on *P. leo* populations[1].

In 2013 a population of around 32,000 *P. leo* in Africa was suggested, based on a compilation of available data sources[3]. The status of *P. leo* was comprehensively reviewed for the IUCN Red List in 2015. The authors of the 2015 IUCN Red List Assessment (RLA) considered the 2013 figure to be an overestimate, as it did not take into account recent changes (mainly declines) that were believed to have taken place in some populations, and thought the number likely to be closer to 20,000 than over 30,000. IUCN categorised the species as Vulnerable in 2016.

The RLA carried out a time trend analysis of census data for the period 1993 to 2014 (three *P. leo* generations) for 47 relatively well monitored *P. leo* subpopulations that together comprise a substantial portion of the total species population[1]. Because of significant differences in observed regional trends, the sample populations were grouped into three regions for analysis: eastern Africa; southern Africa; and western and central Africa. The assessment used a decline from 1118 to 0 for Katavi in Tanzania, although acknowledged that the data were imprecise, and that *P. leo* were still present there. It excluded as an outlier one large population (estimated at ca. 1300 in 2014) in Mozambique (Niassa) that was recorded as having increased in size by over 250% since 1993; the circumstances surrounding this increase were considered unusual, and unlikely to be sustained in the future[1].

Overall, a reduction of 43% in the *P. leo* population over the period 1993 to 2014 was inferred in the RLA, resulting in a classification of the species as Vulnerable. In four southern African countries (Botswana, Namibia, South Africa and Zimbabwe) the population was assessed as having grown overall by 8% in the period; elsewhere in Africa the population was assessed as having declined by just over 60%. However, there were a number of stable or increasing *P. leo* populations in Africa outside southern Africa, and one large population in southern Africa (Okavango in Botswana) that was declining. On the basis of the RLA, the population in the four southern African countries was estimated to comprise around half the total African population of *P. leo* in 2014, compared with around one-quarter in 1993.

Re-analysis of the survey data without the figures for Katavi reduces the inferred overall decline in *P. leo* in Africa between 1993 and 2014 to around 33%; inclusion of the Niassa population would reduce it still further.

The authors of the RLA considered their estimate of the decline might be conservative because they believed that less well monitored populations for which data were not available would be more likely to be declining than well monitored populations. A 2015 paper noted that no reliable data were available on *P. leo* populations or population trends in Angola, Central African Republic, Ethiopia, Somalia and South Sudan and that systematic surveys were absent from large areas of potential *P. leo* habitat in other countries, such as Zambia and Tanzania[4].

Panthera leo has been included in CITES Appendix II since 1975 under the general listing of the family Felidae. The Indian population has been included (as *Panthera leo persica*) in Appendix I since that time. *P. leo* products are in trade. The CITES Trade Database indicates that South Africa is by far the largest exporter; a significant portion of the trade is in trophies from captive-breeding operations. Very little trade from any other range State has been reported. Illegal trade has been reported but is believed currently to be at a relatively low level.

Analysis: The African population of *Panthera leo* does not have a restricted range, nor does it have a small population. The population overall has been declining. Estimates of the rate of decline vary, from around 34% to 43% in the past 21 years (three *P. leo* generations). This is less than the guideline figure given in *Res. Conf. 9.24 (Rev. CoP16*) for a marked recent rate of decline. Moreover, the rate of decline has been slowing because stable or increasing *P. leo* populations, mainly in southern Africa, make up an increasing proportion of the overall population. The African population of *P. leo* would not therefore appear to meet the biological criteria for inclusion in Appendix I.

Reviewers of summary information only: C. Breitenmoser-Würsten.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] Bauer, H., Packer, C., Funston, P.F., Henschel, P. & Nowell, K. (2015) *Panthera leo.* The IUCN Red List of Threatened Species 2015.
[2] Nowell, K and Jackson, P. (Compilers and editors) (1996) *Wild Cats: Status survey and conservation action plan.* IUCN, Cambridge, UK and Gland, Switzerland.
[3] Riggio, J., Jacobson, A., Dollar, L., Bauer, H., Becker, M., Dickman, A., Funston, P., Groom, R., Henschel, P., de Longh, H., Lichtenfeld, L. & Pimm, S. (2013) The size of savannah Africa: A lion's (Panthera leo) view. *Biodiversity and Conservation* 22: 17-35.
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Transfer of North American endemic subspecies of cougar, *Puma concolor coryi* and *Puma concolor couguar*, from Appendix I to Appendix II

Proponent: Canada

Summary: The Cougar *Puma concolor* is a very widely distributed member of the family Felidae, occurring in 23 range States in the Americas. It is included in Appendix II under the general listing of the family Felidae. A number of subspecies have been recognised although the validity of many of these has been questioned. Three subspecies, *P. c. coryi, P. c. costaricensis* and *P. c. couguar*, are currently included in Appendix I. Two of these, both from North America, are proposed for transfer from Appendix I to Appendix II. The current standard taxonomic reference for the great majority of CITES-listed mammals (Wilson and Reeder, 2005)[1] does not recognise these as separate subspecies, considering all North American cougars to belong to a single subspecies *P. c. couguar*. For this reason, the current CITES standard reference for *Puma concolor* is the 1993 edition of Wilson and Reeder.

Puma concolor coryi
The Florida Panther *Puma concolor coryi* exists in a small remnant population in the State of Florida, USA where it occupies less than 5% of its former range. It was reduced to 12-20 individuals in the early 1970s, but now numbers 100-160 following recovery actions. The population is closely monitored and managed in accordance with a recovery plan which aims to achieve long term viability. However there are indications that the population might be at or approaching carrying capacity due to limited availability of suitable habitat in the area. It is affected by habitat loss, degradation, and fragmentation, while lack of human tolerance negatively affects its recovery, and mortality due to collisions with vehicles affects population expansion.

Puma concolor coryi is listed as endangered under the US Endangered Species Act (ESA) of 1973 and is on the State endangered lists for Florida, Georgia, Louisiana, and Mississippi. This prohibits (among other things) import, export, and shipment in foreign commerce without a permit.

There is no national use of *P. c. coryi.* One instance of export to Germany in 2009 of two museum specimens (bone, wild sourced) for scientific research is recorded in the CITES Trade Database. Other records in the CITES Trade Database are either confirmed or likely reporting errors.

Puma concolor couguar
The Eastern Cougar, *Puma concolor couguar* is considered to have been extinct in eastern North America since the late nineteenth century. Sightings of cougars within its former range are thought to be misidentifications, escaped or released exotic pets, or cougars which have migrated in from other areas. *P.c. couguar* is federally listed as endangered under the ESA, which prohibits hunting or trade. The U.S. Fish and Wildlife Service's most recent review (2011) of the subspecies recommends de-listing it based on extinction. In Canada, it is classified as "data deficient" based on doubts as to whether it ever merited recognition as a separate subspecies.

The very few records in the CITES Trade Database for this subspecies are either of old specimens for scientific research or are reporting errors.

The species *Puma concolor* is believed to number at least 30,000 in the USA and 7000-10,000 in Canada. It thrives in a wide range of habitats and is a generalist predator. Although extirpated from its former range in Midwestern and eastern North America, it is attempting to recolonize this region[2]. In the USA its management is under the jurisdiction of individual state and wildlife agencies; most of the western States with viable populations allow strictly regulated sport hunting. It is not popular in the fur trade and there is no significant commercial market. Most international trade is as hunting trophies (skins and trophies), with an average of 120 trophies and 215 skins per year recorded in the CITES Trade Database for 2005-2014. In Canada the species is managed under the Provincial or Territorial Wildlife Acts.

This proposal is based on the outcome of the Periodic Review of the Appendices for Felidae. The transfer would place the two subspecies *P. c. coryi* and *P. c. couguar* in Appendix II under the listing of Felidae spp. If this proposal is adopted, it is suggested that CoP17 adopt Wilson and Reeder 2005 as the taxonomic reference for *Puma concolor*: Regardless of reclassification under CITES, the two subspecies will continue to be recognised and regulated by the US Endangered Species Act of 1973, as amended, as well as by regulations of the States of Florida, Georgia, Louisiana, and Mississippi.

Analysis: It would appear that the subspecies *Puma concolor coryi* still meets the biological criteria for inclusion in Appendix I, having a small and fragmented population and a restricted area of distribution. It is subject to intensive recovery actions that have led to increases in its population. There appears to be no demand in international trade for this subspecies, or reason to expect its transfer would stimulate such demand. Transfer of this subspecies to Appendix II would result in this taxon being included as *Puma concolor* under the general listing of the family Felidae in Appendix II. Trade in *P. concolor* is predominantly in trophies. The subspecies would remain federally protected with strict domestic trade restrictions; hunting and trade would remain unlawful. Assuming *P. c. couguar* were also transferred to Appendix II, the only remaining cougar subspecies in Appendix I would be the Costa Rican Cougar *P. c. costaricensis*, which is geographically isolated from *P. c. coryi*. There is no reason to expect that transfer of *P. c. coryi* to Appendix II would stimulate trade in, or cause enforcement problems for any other species in Appendix I.

There is no risk to *P. c. couguar* from trade as this subspecies is considered extinct and has been since the late nineteenth century. In the unlikely event that the subspecies were it to be re-discovered, currently the subspecies would be protected from hunting and trade by the ESA. However, de-listing the subspecies from the ESA based upon extinction has been proposed.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] Wilson, D. E. & Reeder, D. M. (ed.) (2005) *Mammal Species of the World. A Taxonomic and Geographic Reference.* Third edition, Vol. 1-2, xxxv + 2142 pp. Baltimore (John Hopkins University Press).
[2] Nielsen, C., Thompson, D., Kelly, M. & Lopez-Gonzalez, C.A. (2015) *Puma concolor*. The IUCN Red List of Threatened Species 2015.

Transfer of the Cape Mountain Zebra *Equus zebra zebra* from Appendix I to Appendix II

Proponent: South Africa

Summary: The Cape Mountain Zebra *Equus zebra zebra* is one of two subspecies of the Mountain Zebra *Equus zebra.* It is endemic to South Africa where it is found in the Western, Eastern and Northern Cape provinces. It has been in Appendix I since 1975. The second subspecies, Hartmann's Mountain Zebra *Equus zebra hartmannae*, occurs in Namibia and South Africa. It was included in Appendix II in 1979.

Hunting of the Cape Mountain Zebra and habitat loss resulted in the population being reduced to around 80 individuals in the 1950s. Conservation measures since then involving reintroductions, almost all originating from Mountain Zebra National Park, have led to an increase in numbers and distribution. As of August 2015 the population was estimated to be at least 4791 individuals in at least 75 subpopulations well distributed over the historical range, which comprised around 180,000km^2. It is estimated that 55-70% of the population was mature[1]. Many of the subpopulations are small (37% have 20 or fewer animals), only 11% have over 100 individuals. The population has increased steadily at 8-9% per year since the early 1990s and there are no records of any significant population declines since the 1950s. The taxon has a low reproductive rate and individuals are long-lived.

The major concern regarding the Cape Mountain Zebra at present is the loss of genetic diversity because active meta-population management is not currently practised. However, the low genetic variation within individual populations is offset by moderate variation in the national population. There have been reports of hybridization with other zebras[2].

Approximately 70% of the population occurs in secure state-owned protected areas, the remainder being privately-owned. The movement of the Cape Mountain Zebra is restricted by fences and it is dependent on translocation (e.g. by game farmers) for dispersal. The future growth potential of formally protected source populations is constrained by the availability of state-owned land, which will likely reach its carrying capacity by 2020. To maintain current rates of population increase will either require extending the available land or founding new source populations in areas where suitable land is available[3].

The utilization of the Cape Mountain Zebra is controlled under national and provincial legislation. This includes a permit system regulated by the National Environmental Management: Biodiversity Act (NEMBA), and the Threatened or Protected Species (TOPS) Regulations. Permit holders are required to give annual feedback to the Issuing Authority on compliance with permit conditions, which provides a means of monitoring effectiveness.

Illegal translocations and poaching of the Cape Mountain Zebra occur on a limited scale but there is reportedly no illegal offtake at present from any of the national parks where it occurs[4,5]. Cases of the Cape Mountain Zebra being hunted, sold or exported as Hartmann's Mountain Zebra have been reported[4]. There is currently limited reported (assumed legal) international trade. Trade reported by South Africa in 2000 to 2014 included nine trophies and seven skins.

Conditional to the transfer of Cape Mountain Zebra from Appendix I to Appendix II, South Africa proposes to implement a combination of active adaptive harvest management and management strategy evaluation to set a hunting quota for the Cape Mountain Zebra. It is argued that introduction of a hunting quota will have a beneficial effect by providing incentives for private owners to invest in the Cape Mountain Zebras, increasing the possibility that new subpopulations will be established. Initial responses from the private sector indicate that this is the case.

The quota will be determined through a population viability analysis that considers genetic diversity within the population. The implementation of the quota will be monitored through a research project. As safeguards, a national Biodiversity Management Plan (BMP) for the species will be adopted and feedback will be required from permit holders in terms of TOPS. The BMP was being finalized[6] at the time of writing with plans to make it available as a CITES CoP17 Information Document.

An individual-based simulation tool has been developed to evaluate the impacts of life-stage and sex-specific hunting quotas and translocation strategies for populations over several years. An initial trial use of a population simulation model was applied using the available count data for eight protected populations[7]. The

simulation model will further be used to assess the viability of each hunting quota proposed by private sector owners of the Cape Mountain Zebra who had expressed interest in making use of a hunting quota.

Some concerns have been expressed regarding the efficacy of TOPS reporting as a management tool. The Scientific Authority (SA) of South Africa noted in 2015[4] that the effects of harvest, which included both translocation and hunting, were not monitored and there was often a lack of knowledge of what happens on the ground. Furthermore, budgetary and human resource capacity gaps may limit the efficacy of the harvest management and permitting system. It is also unclear whether the simulation tool intended to be used in setting quotas integrates the Cape Mountain Zebra population viability assessment data, important for management in the context of the potential loss of genetic diversity.

There is reported international trade in Hartman's Mountain Zebra. According to the CITES Trade Database, between 2000 and 2014, direct exports included 22,334 skins (96% from Namibia) and 9755 trophies (91% from Namibia and 8% from South Africa).

The Cape Mountain Zebra is classified in the IUCN Red List as Vulnerable (2008). The Red List of Mammals of South Africa, Swaziland and Lesotho[1] assessed the Cape Mountain Zebra as Least Concern, and the update of the global assessment is underway[8].

Analysis: The Cape Mountain Zebra does not have a restricted distribution. Its population is still relatively small but is increasing and not regarded as under threat, although in the long term loss of genetic diversity may be a concern. The subspecies does not appear to meet the biological criteria for inclusion in Appendix I.

For a transfer from Appendix I to Appendix II the precautionary measures in Annex 4 of the Resolution should be met. These can be met in various ways, including the Parties being satisfied with the range State's implementation of the Convention, particularly Article IV, and with its enforcement controls and compliance with the Convention, or if an integral part of the amendment proposal is an export quota or other special measure approved by the CoP, based on management measures described in the Supporting Statement, provided that effective enforcement controls are in place.

In this case the use of a system to set hunting quotas may be considered as a special measure. The Supporting Statement describes the approach that would be used and indicates that a Biodiversity Management Plan for the species will be adopted. It is not clear to what extent the plan addresses the long-term issue of potential loss of genetic diversity. At present 70% of the population is in protected areas where no hunting takes place. This would not change in the event of a transfer to Appendix II.

The current inclusion of the Cape Mountain Zebra in Appendix I is inconsistent with recommendations for split-listing set out in Annex 3 of *Res. Conf. 9.24. (Rev. CoP16)*, which advise that split-listings of a species in more than one Appendix should be avoided and that when split-listings occur they should be on the basis of national or regional populations rather than subspecies. Were it to be transferred to Appendix II, the entire species *Equus zebra* would be in Appendix II, consistent with the terms of this Resolution.

References:

Information not referenced in the Summary section is from the Supporting Statement.

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[1] Hrabar, H., Birss, C., Peinke, D., King, S., Novellie, P., Kerley, G. and Child, M. (2015) *A Conservation Assessment of Equus zebra ssp. zebra.* In: M.F. Child, E. Do Linh San, D. Raimondo, H. Davies-Mostert and L. Roxburgh (eds) *The Red List of Mammals of South Africa, Swaziland and Lesotho.* South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa.

[2] Winker, H. (2016a) *Time-series analysis of long-term established Mountain Zebras within protected areas (1985-2015) with implications for IUCN Red Listing*. SANBI Technical Report SANBI/BAM/STATS/2016/MZ/H1, 7th of March 2016, Kirstenbosch, South Africa.

[3] Winker, H. (2016b) *Incorporating carrying capacity limits into forward projection of source populations of Cape Mountain Zebra.* SANBI Technical Report SANBI/BAM/STATS /2016/MZ/H1S2, 16th of March 2016, Kirstenbosch, South Africa.

[4] Scientific Authority of South Africa. (2015) *Non-detriment finding for Equus zebra zebra (Cape Mountain Zebra).* Issued by the CITES Scientific Authority, South Africa. May 2015.

[5] CITES Trade Database http://trade.cites.org/.

[6] Pfab, M. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

[7] Winker, H. (2016c) *Development of a population simulation model for Cape Mountain Zebra towards formal evaluation of management strategies.* SANBI Technical Report SANBI/BAM/STATS/2016/MZ/H2, 9th of March 2016, Kirstenbosch, South Africa.

[8] King, S. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

To alter the existing annotation on the Appendix II listing of Swaziland's White Rhino, adopted at the 13th Conference of Parties in 2004, so as to permit a limited and regulated trade in White Rhino horn which has been collected in the past from natural deaths, or recovered from poached Swazi rhino, as well as horn to be harvested in a non-lethal way from a limited number of White Rhino in the future in Swaziland

Proponent: Swaziland

Summary: The Southern White Rhino *Ceratotherium simum simum* is one of two subspecies of White Rhino, the other being the Northern White Rhino *C. s. cottoni,* now believed extinct in the wild.

The Southern White Rhino currently numbers just over 20,000 wild individuals, having increased from 20 to 50 in 1895. Over 90% of the population is in South Africa. There are reintroduced populations in Botswana, Kenya, Mozambique, Namibia, Swaziland, Uganda and Zimbabwe and an introduced population in Zambia. There are ca. 700 individuals in captivity around the world. The subspecies was classified in the IUCN Red List as Near Threatened (2011). Until recently the population was growing (averaging 7% growth per year for 1992 to 2010)[1]. However due to escalating poaching since 2008 the global population has levelled off. At a continental level reported poaching declined slightly in 2015[1].

Having become extinct in Swaziland in the mid-20th century, the White Rhino was reintroduced from South Africa in 1965. The population in Swaziland reached a peak of around 120 in the late 1980s but was reduced by poaching to 24 in 1992. Improved protection led to population recovery. In 2015 the population stood at 90 individuals, representing an average of 6% annual growth since 1992. Drought-induced mortality has recently reduced this number to 73 (as of April 2016). Losses due to poaching have been negligible to date, comprising two individuals in 2011 and one in 2014. The population is confined to secure areas totalling 10,000ha in two protected areas (Hlane and Mkhaya Game Reserves) in the eastern part of the country.

The entire rhinoceros family, the Rhinocerotidae, was included in Appendix I of CITES in 1977. The South African population of Southern White Rhino was transferred to Appendix II in 1994 under the following annotation: "For the exclusive purpose of allowing international trade in live animals to appropriate and acceptable destinations and hunting trophies. All other specimens shall be deemed to be specimens of species included in Appendix I and the trade in them shall be regulated accordingly." In 2004, Swaziland's population was transferred to Appendix II under the same annotation. At that time the population there numbered 60 individuals.

There has been very limited trade of Southern White Rhinos between Swaziland and South Africa since 2005, almost all in live individuals with some scientific specimens. As well as exporting live individuals, Swaziland has imported some to enhance the genetic diversity of its rhino population.

This proposal is to alter the existing annotation as it applies to Swaziland's population, so as to permit a limited and regulated trade in White Rhino horn which has been collected in the past from natural deaths, or recovered from poached Swazi rhino, as well as horn to be harvested in a non-lethal way from a limited number of individuals in the future in Swaziland.

According to the Supporting Statement, Swaziland wishes to sell existing stocks of some 330kg to a small number of licensed retailers and to sell harvested horn, sourced from sustainable, non-lethal harvesting, at the rate of 20kg per year, to those retailers. Funds raised would be used to contribute to conservation of the White Rhino in Swaziland and maintenance of the protected areas where the species occurs there. Big Game Parks, the CITES Management Authority for Swaziland, would be the sole seller and buyers would be licensed and approved by CITES. The proponent states that the DNA profiles of all horn offered for sale would be recorded in a national register and made available to TRAFFIC. The trading operation would be open to inspection by the CITES Secretariat. It would be stopped if it were judged to be having a negative impact on the population of the species in Swaziland.

Analysis: The Swaziland population of Southern White Rhino was transferred from Appendix I to Appendix II in 2004 under an annotation that can be interpreted as satisfying the precautionary safeguard set out in sub-paragraph A 2 a) iii) of Annex 4 of *Res. Conf. 9.24 (Rev. CoP16)*, in that it is a special measure approved by the CoP, based on management measures described in the Supporting Statement. This sub-paragraph also states that effective enforcement controls should be in place. In approving the transfer, CoP13 agreed that this was the case.

The current proposal is to maintain the population in Appendix II under a different annotation which may also be considered a special measure under the same part of Annex 4 of *Res. Conf. 9.24 (Rev. CoP16).*

It appears that the management measures currently in place (i.e. since the transfer of the population to Appendix II in 2004) are satisfactory as the population has increased overall in that time, despite a recent drought-induced decline; the very small amount of poaching recorded indicates that effective enforcement controls are in place as regards protection of living animals.

Export of stockpiled horn would have no direct impact on the living population of Southern White Rhinoceros in Swaziland. Other than indicating that it will be non-lethal, the Supporting Statement does not provide details as to how horn would be collected from living individuals. Routine temporary immobilisation for dehorning is a standardised procedure in some South African White Rhino populations with no clear adverse effects on the population or their breeding performance[1].

Swaziland gives estimated average weights of horn per individual as 7.02kg, this presumably being derived from weights of horns in the current stockpile. This figure is slightly higher than an average of 5.8kg from a large sample. Swaziland proposes to harvest 20kg of horn per year, indicating an average annual offtake of 1kg per animal. This has been demonstrated to be sustainable production for male White Rhinos but a figure closer to 0.6kg is more realistic for females. Given existing numbers and expected age and sex structure of the population, sustainable harvest of 20kg per year should be achievable. Research of stress levels currently shows little impact of regular dehorning every 18 months in a South African White Rhino population that is breeding well[1].

Few details are provided as to how the proposed trade will be carried out and controlled; for example there is no indication of how appropriate purchasers would be selected and how and by whom these would be licensed. It is not clear exactly what role either TRAFFIC or the CITES Secretariat would be expected to play in oversight of any trade. Much of the detail needed to assess these aspects of the precautionary measures implied by the proposal is therefore not provided.

Reviewers of summary information only: R. Emslie.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] Emslie, R. (2016) *In litt*. to IUCN/TRAFFIC Analyses Team. Cambridge, UK.

Pangolins and CITES: An Overview

Pangolins, or scaly anteaters, are mammals covered in overlapping, horny scales. There are eight species, all in the genus *Manis* following the CITES standard nomenclature for these species. Four are collectively distributed in South, East and Southeast Asia: Chinese Pangolin *Manis pentadactyla*, Sunda Pangolin *M. javanica,* Indian Pangolin *M. crassicaudata*, and Philippine Pangolin *M. culionensis* (only recently split from *M. javanica*). Four are native to sub-Saharan Africa: Black-bellied Pangolin *M. tetradactyla*, White-bellied Pangolin *M. tricuspis*, Giant Pangolin *M. gigantea*, and Temminck's Ground Pangolin *M. temminckii*. Pangolins occur in a wide variety of habitats from tropical, sub-tropical, moist lowland and swamp forests to savannah woodland, scrub and floodplain grasslands. They feed exclusively on ants and termites.

Throughout their range in both Africa and Asia, pangolins have been exploited historically as a source of protein and traditional medicine[1]. In Asia, pangolin meat has been consumed historically at the local level in virtually every range State, as well as in urban centres. Scales have been used for a wide variety of applications, particularly as an ingredient in traditional medicines in China and Vietnam[2]. It is estimated that up to 160,000 pangolins were harvested annually in China between the 1960s and 1980s for domestic use[3].

Pangolin body parts, primarily meat, scales and skins, have also been traded internationally. Trade in commercial volumes of scales took place from Indonesia to China and Hong Kong as early as the 1920s; and an estimated 60,000 pangolins were harvested annually in Southeast Asia from the 1950s to 1970s for the Taiwanese leather industry[2].

Pangolins have a long history in CITES. In 1975 *M. pentadactyla*, *M. javanica* and *M. crassicaudata* were listed in Appendix II and *M. temminckii* in Appendix I. In 1994 *M. temminckii* was transferred from Appendix I to Appendix II, and all remaining species were included in Appendix II.

In 2000 *M. pentadactyla*, *M. javanica* and *M. crassicaudata* were subject to a proposal to transfer them to Appendix I. However, the proposal was not accepted because the species were at that time still under the Review of Significant Trade process (see below); instead the Parties adopted zero export quotas for wild-caught Asian pangolins traded for primarily commercial purposes. This has remained in place since. African pangolins do not have a zero quota but the species are protected in many of their range States and little legal trade has been reported.

The majority of trade reported in the CITES Trade Database from 1978 to 2000 involved the Asian species, almost all *M. javanica* (which at that time included *M. culionensis*) and was predominantly exports of skins for leather production; this trade involved around 10,000 skins per year. Reported volumes of scales in trade were much lower (less than 20,000kg in total for 1978-2000)[4]. Evidence from the Review of Significant Trade process (see below) suggests the volume of illegal trade, predominantly in scales and live animals, taking place at the time probably equalled, if not exceeded, reported trade volumes[4].

Since 2000, comparatively little trade in Asian (for which there is a zero quota) or African pangolins has been reported to CITES. However, large volumes of illicit trade have taken place, involving a minimum estimate of some 17,000 pangolins globally each year[5]. This has included large consignments of pangolins, their meat and scales within Asia, and an increasing number of confiscations of African pangolin derivatives, primarily scales, in Africa, Europe and Asia, with most seizures ultimately bound for Asia.

Concern over sustainability of trade reported to CITES, particularly in skins, led to the inclusion of Asian pangolins in various phases of the Review of Significant Trade process in 1988 (preliminary Phase), 1992 (Phase I) and 1999 (Phase IV). Recommended actions were made for various range States to control trade in Phases I and IV. In 1999 the Standing Committee recommended to all Parties that no export or re-export certificates be issued, or accepted, for specimens of Asian pangolins until, to the satisfaction of the Secretariat, exporters had implemented a series of measures that demonstrated compliance with Article IV of the Convention. Noting that zero export quotas were established for these species at CoP11, the Standing Committee, at SC45 (2001), agreed that if zero quotas were removed any range State wishing to trade in these species should satisfy the Secretariat that the 1999 recommendations had been implemented before any export took place.

The African species *M. tetradactyla, M. tricuspis, M. gigantea* and *M. temminckii* were included in Phase IV of the RST but were subsequently eliminated from the process on the basis that trade levels were not of concern. *M. gigantea* and *M. tricuspis* were again selected for the RST as species of priority concern in 2013. At AC28 (September 2015), the Animals Committee decided to retain in the RST all range States for these species that do not fully protect them through national legislation (with the exception of the United Republic of Tanzania which was the only range States to provide a response to the Secretariat). It is intended to review further information on these range States at AC29.

At SC66, the Standing Committee established an inter-sessional working group on pangolins which drafted a resolution on *Conservation of and trade in pangolins* to be submitted to CoP17. The CoP will also consider a draft decision on pangolins, directing the Secretariat, subject to external funding, to prepare a report on the conservation status of pangolins including relevant enforcement actions and developments regarding demand management measures.

If Proposals 8 - 12 are accepted at CoP17, all pangolin species will be in Appendix I.

References:

[1] Boakye, M.K. Pietersen, D.W., Kotze, A., Dalton, D.L. & Jansen, R. (2014) Ethnomedicinal use of African pangolins by traditional medical practitioners in Sierra Leone. *Journal of Ethnobiology and Ethnomedicine* 10:76.
[2] Anon. (1999) Review of Significant Trade in Animal Species included in CITES Appendix II: Detailed Reviews of 37 Species, *Manis pentadactyla*. Draft Report to the CITES Animals Committee. World Conservation Monitoring Centre, IUCN Species Survival Commission and TRAFFIC Network.
[3] Zhang, Y. (2008) Conservation and Trade Control of Pangolins in China. 66-74 *In* Pantel, S. & Chin, S-Y. (2008). Proceedings of the Workshop on Trade and Conservation of Pangolin Native to South and Southeast Asia. 30 June – 2 July 2008, TRAFFIC Southeast Asia, Petaling Jaya, Selangor, Malaysia.
[4] Challender, D.W.S., Harrop, S.R. & MacMillan, D.C. (2015) Understanding markets to conserve trade threatened species in CITES. *Biological Conservation* 187: 249-259.
[5] IUCN SSC Pangolin Specialist Group (2016) The conservation status, illegal trade and use of pangolins (*Manis* spp.). CITES SC66 Inf. 23. Prepared by the IUCN SSC Pangolin Specialist Group.

Transfer of Indian Pangolin *Manis crassicaudata* from Appendix II to Appendix I

Proposal 8 Proponent: Bangladesh

Proposal 9 Proponents: India, Nepal, Sri Lanka and United States of America

Note: *Proposals 8 and 9 are identical in intent. One analysis is presented for the two.*

Summary: The Indian Pangolin *Manis crassicaudata* occurs in South Asia from northeast and southeast Pakistan south throughout the Indian sub-continent, including Sri Lanka, and east to southern Nepal. It was found throughout Bangladesh historically but there are no recent records and the species may be extinct there. There are historical records of this species from southwest China (Yunnan Province) where its presence is uncertain, and dubious records from Myanmar. *Manis crassicaudata* occurs in various types of tropical forest as well as grassland, open land and degraded habitat. Like other species of pangolin it is solitary and is considered to have low fecundity, giving birth to one young typically (though there are observations of twins), after a gestation period of approximately six months. Females typically give birth annually.

There is a lack of quantitative population data for this species. Its status in India, which comprises by far the largest part of its range (very approximately three million $km^{2)}$, is not well known, though it was reported in the early 1980s that populations were greatly reduced by hunting, and it is currently listed as vulnerable nationally. This species is protected in India meaning hunting and trade are prohibited but seizures of pangolin derivatives have been made across India between 2009 and 2014 indicating some level of exploitation. It is reportedly of variable abundance in Sri Lanka, but nowhere common.

Manis crassicaudata is also protected in Pakistan where hunting is prohibited, but there is evidence of population declines, driven by harvesting for illegal trade. In the Potohar Plateau region of northeast Pakistan, which forms a large part of this species' range in Pakistan (ca. 22,000km^2), it is estimated that the average population density of *M. crassicaudata* underwent an 80% decline between 2010 and 2012, from around one individual per km^2 to one every 5 or so km^2. Data from the last three to four years reveals the killing of around 400 pangolins here, although this is likely to be an underestimate[1]. Additional data from Pakistan indicate that targeted exploitation of this species also occurs in Azad Jammu and Kashmir, with an estimated illegal offtake of ca. 500 between 2012 and 2016[1], again likely to be an underestimate.

Since 2000, seizure data indicate that illicit, international trade in at least 8000 *M. crassicaudata* may have taken place[2]. International demand for this and other species of *Manis*, especially for scales, is believed to be increasing as a result of significant declines in populations of *M. pentadactyla* and *M. javanica.* This species is also said to be heavily hunted for local consumption.

This species is classified in the IUCN Red List as Endangered (2014).

Analysis: *Manis crassicaudata* is affected by trade. There are few data on the population status of this species. It is believed to have been extirpated from some of its range, in Bangladesh and populations appear to have declined markedly due to poaching in parts of Pakistan. Very little is known on the population in India which is the majority of the species' range, although it is believed to have been reduced. There is therefore insufficient information to determine whether the species meets the criteria for inclusion in Appendix I.

Reviewers of summary information only: D. Pietersen.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] Mahmood, T. (2016) *In litt*. to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
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Transfer of Philippine Pangolin *Manis culionensis* from Appendix II to Appendix I

Proponents: Philippines and United States of America

Summary: The Philippine Pangolin *Manis culionensis* is endemic to six islands in the Philippines: mainland Palawan and the much smaller adjacent islands of Coron, Culion, Balabac, Busuanga and Dumaran Island. It has also been introduced to Apulit Island adjacent to Palawan. Pangolin populations in the Philippines were previously considered part of the Sunda Pangolin *Manis javanica*, but were split from it in 2005. The species occurs in lowland primary and secondary forests, grasslands/secondary growth mosaics, mixed mosaics of agricultural lands and scrubland adjacent to secondary forests. It is solitary and typically gives birth annually to one young after a gestation period of approximately six months[1]. It is thought that breeding occurs in August and September. Generation time is taken as seven years.

There is a lack of population data, mainly because the species is elusive, solitary and nocturnal. In 2004 it was described by local people as fairly common, though subject to moderately heavy hunting pressure[2]. There are relatively recent (2012) estimates of densities of 0.05 individuals per km^2 in primary forest and 0.01 per km^2 in mixed forest/brush land[3]. Higher estimates made in 2014 of 2.5 adult pangolins per km^2 on Palawan and Dumaran Island are considered unreliable[4]. The species is thought still to be considerably more abundant in northern and central Palawan than in the south; it is reportedly abundant on Dumaran Island (435km^2). Local hunters on Palawan report that populations are declining as a result of hunting. One study on Palawan in 2012 reported that increased effort is now needed to catch pangolins, potentially as a consequence of declining populations.

The species is believed to be affected by habitat loss and degradation caused by shifting cultivation and conversion of forest to permanent agricultural crops and industrial tree plantations, particularly palm oil. Palawan, with an area of 15,000km^2 and estimated tree cover in 2000 of 10,000km^2, lost an estimated 770km^2 of tree cover between 2001 and 2014[5]. As noted above, observed densities in secondary habitats are much lower than those in primary forests.

The CITES Trade Database records export of around 1200 pangolins per year from the Philippines between 1982 and 1989 (predominantly skins and live) reported as *Manis javanica* (before *M. culionensis* was split from it). Reported trade dropped to almost zero after 1989. The Philippines made illegal the export of all wild-caught fauna in 1995. A review of trade in Asiatic pangolins suggested that, based on seizures, around 70 per year were illegally traded between 2000 and 2013. Since 2010, there have been 17 seizures involving the *M. culionensis* in Palawan province believed to be destined for international trade[6].

This species is classified by IUCN as Endangered (2014). Since 2015, this species has been listed as critically endangered in the Philippines under Palawan Council for Sustainable Development (PCSD) Resolution No. 15-521.

Analysis: Information on the status of the *M. culionensis* is scarce. The species does not have a restricted range. If recent estimates of population density (of ca. one per 20km^2 in dense forest and one per 100km^2 in forest/scrub mosaic) are reliable, then the global population may be small. There are no baseline data on which to base population trends although there is anecdotal information that the species is scarcer than it was, in at least part of its range. If historic records of legal trade and recent estimates of illegal trade are at all reliable, there has been a marked decline in trade in the past 20-30 years, from ca. 1200 per year in the 1980s to around 70 per year in 2000-2013. Assuming this is not due to reduced hunting effort (unlikely) or improved enforcement efforts, it might be indicative of a corresponding marked decline in the wild population. If that were the case (or the estimate for a small and probably declining wild population were reliable), the species would appear to meet the criteria for inclusion in Appendix I in *Res. Conf. 9.24 (Rev. CoP16).*

Reviewers of summary information only: C. Waterman**.**

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Transfer of Sunda Pangolin *Manis javanica* and Chinese Pangolin *M. pentadactyla* from Appendix II to Appendix I

Proponent: Viet Nam, Bhutan and United States of America

Note: *This document should be read in conjunction with the introduction to the pangolin proposals.*

Summary: The Sunda Pangolin *Manis javanica* is widely distributed geographically across mainland and island Southeast Asia; the Chinese Pangolin *M. pentadactyla* is found from the Himalayan foothills into southern China. Both species occur in tropical and subtropical forests, as well as cultivated landscapes including plantations. Both are solitary and typically give birth to one young after a gestation period of approximately six months, possibly on an annual basis. Research suggests that *M. javanica* may breed all year round[1], but *M. pentadactyla* has a discrete breeding season[2]. Generation time is taken as seven years. There is generally a lack of information on population recruitment rates for these species.

Manis javanica

Manis javanica is native to Brunei Darussalam, Cambodia, China (based on a number of museum records)[3], Indonesia, People's Democratic Republic of Lao (Lao PDR), Malaysia, Myanmar, Singapore, Thailand and Viet Nam. It is considered to now be extremely rare in the northern part of its range. Populations are generally considered to be declining, much of which is attributed to harvesting. The species occurs in both cultivated and uncultivated habitats; there is little information on relative population densities in different habitat types.

Information on the status of populations of *M. javanica* is scarce. Where declines have been reported they have almost invariably been ascribed to hunting, chiefly for international trade. Little is known of status in Brunei Darussalam although confiscations in the last few years indicate that illegal trade does occur there[4]. In Cambodia, *M. javanica* is present but based on interviews with hunters is understood to be declining. In Indonesia there is very little information on status, but seizures in recent years, sometimes of several thousand animals, indicates that there is intense hunting pressure in the country. In Lao PDR, there have reportedly been huge declines in recent decades. Local communities in the late 1990s reported populations had declined, in some areas to as little is 1% of the level in the 1960s. Interviews with hunters in Peninsular Malaysia indicate the species is declining[5]. Populations appear to be stable in Singapore based on the frequency of sightings[6]. One study in 2005 and 2006 on Palau Tekong, a 25km^2 island immediately adjacent to mainland Singapore, found pangolins reasonably common there, and estimated an average home range of some 45ha, with some overlap, based on telemetry of three individuals. The species is reported to be increasingly rare in Thailand, but has been detected in a number of national parks in the last decade[7]. In Khlong Nakha Wildlife Sanctuary in Thailand *M. javanica* has been camera-trapped several times in the past 12 months[8]. In interviews conducted with 99 households around this sanctuary, 80% of respondents reported a decrease in the population[8]. In Viet Nam, this species is reported to have declined severely, especially since the opening of export markets in the 1990s.

Manis javanica has historically been exploited for consumptive use of its derivatives across its range, predominantly its meat and scales as a source of protein and for traditional medicine applications respectively. While domestic use continues, in many places this has been substituted for international trade. This species has been subject to significant levels of trade, both legal and illegal.

The majority of trade in pangolins reported in the CITES Trade Database from 1978 to 2000 was reported as *M. javanica* (which at that time included *M. culionensis*) and was predominantly exports of skins for leather production; this trade involved around 11,000 skins per year, of which just under 9000 was of *M. javanica* in its current sense, rather than *M. culionensis*. Reported volumes of scales in trade were much lower (less than 20,000kg in total for 1978 to 2000). It is thought that the volume of illegal trade, predominantly in scales and live animals, taking place at the time probably equalled, if not exceeded, reported trade volumes.

Since 2000, there has been virtually no legal trade in *M. javanica* reported, as there is a zero export quota for wild specimens of this and all other Asian species for commercial purposes. However, large volumes of illicit trade have taken place, involving a minimum estimate of some 17,000 pangolins globally each year[9]. It is believed that a large proportion of this trade involves *M. javanica.*

Manis javanica was classified as Critically Endangered by IUCN (2014).

Manis pentadactyla

Manis pentadactyla is native to Bhutan, China, India, Lao PDR, Myanmar, Nepal, Thailand and Viet Nam. In China the population was estimated at 50,000 to 100,000 animals in 2003. Populations in China are estimated to have declined by 88 to 94% since the 1960s to 2004. It is now very rare in Guangxi and Yunnan Provinces and considered to face a high risk of extinction in Hainan. It is considered rare in Hong Kong (Special Administrative Region). In Taiwan (Province of China) the species has reportedly recovered in some places from historical reductions, with estimated densities in some areas of 12 to 13 adult pangolins per km^2 [10]. There is virtually no information on wild status in India; confiscations suggest the species is under heavy pressure there. Field sightings in Lao PDR are also now extremely rare. The population in Nepal was estimated at approximately 5000 individuals in 2011 and is believed to be in decline. In Viet Nam hunters report that it has declined severely in the past two decades; it is now regarded as extremely rare.

Manis pentadactyla has historically been exploited for consumptive use of its derivatives across its range, predominantly its meat, as a protein source, and its scales for use in traditional medicines. In China, it is estimated that 160,000 animals were harvested annually for these reasons between the 1960s and 1980s[11].

Reported volumes of international trade are considerably lower than those for *M. javanica*. Before 2000, when the zero export quota for all Asian pangolin species was established, on average fewer than 1000 individuals were reported in trade each year, mostly skins imported by the USA and Mexico. As with *M. javanica* there is thought to have been a high volume of illegal trade at the time. Since 2000 seizures and records of trade (e.g., from court cases) indicate that a substantial illicit trade has taken place, potentially involving over 4000 individuals per year[12].

This species is classified as Critically Endangered by IUCN (2014).

Although there is generally a lack of quantitative population data for these species, historical declines have been documented in places, and in others available evidence indicates that populations are in serious decline.

Analysis: Both *Manis javanica* and *M. pentadactyla* are widespread species. Information on population status is scarce, but neither is likely to have a small global population. There are reports (some anecdotal) of very severe declines in the past two or three decades in a number of range States of both species, invariably ascribed to exploitation, and one quantitative estimate, for China, of a reduction in the population of *M. pentadactyla* of ca. 90% between 1960s and the early 2000s. China comprises the greater part of the range of *M. pentadactyla*. If this estimate is robust, it would indicate a severe historical decline in the global population of this species. Direct information on the status of *M. javanica* is lacking for large parts of its range, most notably Indonesia. The species is known to be harvested extensively there. Given its low productivity and likely relatively low population density (based on estimates for the closely related *M. culionensis* (see Analyses for Prop. 10) and indications of fairly extensive home range of *M. javanica*) it is possible that this harvest has led to a decline in population within the guidelines for inclusion in Appendix I, that is of 50% of more within three generations (21 years in this case). There are not known to be any major unexploited populations. Both species are affected by trade.

It is possible therefore that both species meet the criteria for inclusion in Appendix I.

Reviewers of summary information only: C. Waterman.

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Transfer of African pangolin species *Manis tetradactyla, M. tricuspis*, *M. gigantea* and *M. temminckii* from Appendix II to Appendix I

Proponents: Angola, Botswana, Chad, Côte d'Ivoire, Gabon, Guinea, Kenya, Liberia, Nigeria, Senegal, South Africa, Togo and United States of America

Note: *This document should be read in conjunction with the introduction to the pangolin proposals.*

Summary: There are four species of pangolin in Africa. Three, *Manis gigantea* (Giant Pangolin), *M. tricuspis* (White-bellied Pangolin) and *M. tetradactyla* (Black-bellied Pangolin), occur in moist forests and associated habitats in West and Central Africa. The fourth, *M. temminckii* (Temminck's Ground Pangolin), is more widespread and occurs in drier habitats principally in eastern and southern Africa. All are solitary and give birth to a single young. Gestation period is taken as five months except for *M. temminckii*, where it may be three to four months. This and *M. gigantea* may only breed every second year; in the other two species breeding is believed to be aseasonal and more or less continuous. Generation time is taken as seven years for *M. tetradactyla* and *M. tricuspis,* and nine years for *M. gigantea* and *M. temminckii*.

Manis gigantea
Manis gigantea is a terrestrial species discontinuously distributed in 18 range States in West and Central Africa from Senegal eastwards to South Sudan[1], Uganda and Tanzania. It was previously considered extinct in Rwanda but recent evidence suggests it still persists there[2]. The CITES Management Authority of Uganda reported a national population estimate, based on camera trapping, of just over 2000 individuals, with densities of up to six individuals per km^2. Based on observed population densities for *M. temminckii*, also a terrestrial species, this seems high. The species reportedly generally avoids areas of high human impact, but has been found in forest mosaics. It is classified in the IUCN Red List as Vulnerable (2014).

Manis tricuspis
Manis tricuspis is a semi-terrestrial species occurring in 22 range States from Guinea-Bissau through much of West and Central Africa to southwest Kenya and northwest Tanzania, south to northwestern Zambia and northern Angola. The species is known to be present in modified habitats, including secondary growth in oil palm groves, teak plantations and secondary rainforest as well as agricultural areas of former lowland rainforest. It can reportedly be found at relatively high densities in suitable habitat. Research in Benin has suggested an average density of 0.84 individuals per km^2 during the dry season in both plantations and natural forest. This species is classified in the IUCN Red List as Vulnerable (2014).

Manis tetradactyla
Manis tetradactyla has been recorded in 11 range States from Sierra Leone eastward through West Africa to the Congo Basin. It may also occur in Angola and Uganda. It is the most arboreal of African pangolins and may therefore be expected to be the most forest-dependent, although has also been recorded in altered forests and farmland[3]. It is classified in the IUCN Red List as Vulnerable (2014).

Manis temminckii
Manis temminckii is recorded in 14 range States and is possibly extinct in a 15^{th} (Swaziland). It is the most widespread African pangolin species occurring from southeast Chad and extreme northeast Central African Republic through much of East and southern Africa as far south as South Africa. It mainly inhabits savannah woodland with dense scrub as well as floodplain grasslands and sandveld, and occurs on well-managed livestock farms. Limited information is available on population densities. Research in the Northern Cape Province of South Africa showed average densities of 0.24-0.3 individuals per km^2. In the Gokwe district of Zimbabwe they have been estimated at 0.1 individuals per km^2. *Manis temminckii* is classified in the IUCN Red List as Vulnerable (2014).

The main factors believed to be affecting African pangolins are exploitation and, in the case of the three forest-dwelling species, habitat loss and degradation. African pangolins continue to be hunted locally for their meat and use of their body parts and derivatives in traditional medicines. As well as local use there is evidence of (growing) international trade most of which is illicit, and involves pangolin derivatives, mainly scales, to Asian markets. Recent research suggests that the proportion of pangolins hunted has increased significantly compared to other vertebrates in sub-Saharan Africa, with a 9-fold increase from 2005 to 2014. Prices for pangolins in Nigeria (where all three forest-dwelling species occur) have reportedly increased 10-fold over the last five years. In Zimbabwe (where only *M. temminckii* occurs) the number of poaching cases has been shown to be increasing rapidly, and as of 2015, poachers and traffickers from neighbouring

countries have reportedly become involved in pangolin poaching in Zimbabwe[4]. Seizures of African pangolin derivatives in trade have been made with increasing frequency in recent years. These have mainly involved scales destined for Asian markets; some shipments have been detected in Europe en route to Asia[5]. According to the Supporting Statement, almost 15,000kg of scales from African pangolins were seized between 2013 and 2015, representing between 4,000 and 25,000 animals, depending on the species involved.

The apparently growing international trade is believed to have two causes: declining availability of Asiatic pangolin derivatives, and increasing trade globally between African nations and East Asia, which facilitates trafficking of these species[6].

The scale or proportion of the illegal trade in the different species is difficult to determine from seizure data given the difficultly in differentiating between pangolin species on the basis of the products in trade (chiefly scales).

Conversion of forest is believed to have an impact on *M. gigantea, M. tricuspis* and *M. tetradactyla*. Overall loss of tree cover in West and Central Africa, where these species occur, has been estimated at just under 4% in the period 2000 to 2014, or roughly 0.25% p.a.[7]. Rates of loss in Central Africa, where roughly 80% of the total forest cover in this area is found, are lower (ca. 0.2%) than they are in West Africa. All three species are found in modified habitats, but there is little information on their ability to persist in areas of entirely modified habitat. As well as affecting habitat for the species, the opening up of areas for activities such as logging also improves access for hunting.

Manis temminckii is likely to be less affected by loss of tree cover but may be affected by other land-use changes. There is mortality from electrocution on electric fences, especially in South Africa. Mortality rates of one individual per 11km of electrified fence per year have been estimated, with a bias towards male mortalities, probably because males range more widely than females.

All four species have been included in Appendix II since1994 (*M. temminckii* was listed in Appendix I between 1975 and 1994). Trade since then reported in the CITES Trade Database has been very limited. Most has been in *M. tricuspis*, with an annual average of 50 live animals, 20 skins and 40kg of scales exported across all range States in the period 1994 to 2014. Annual averages of round 150kg of scales and 12 skins per year of *M. gigantea* were also recorded. Recorded trade in the other two species has been negligible.

A number of range States have prohibited hunting and trade in native pangolin species.

Analysis: Information on population status of all four African species of pangolins is scarce. None of them have a restricted range and, although they may occur at low densities (less than one individual per square kilometre in the case of the only two, *Manis tricuspis* and *M. temminckii,* for which reliable information is available), none is likely to have a small population. There is no population trend information. Changes in population have been inferred from the presumed impacts of habitat alteration and hunting. Rates of conversion of habitat for the three forest-dwelling species (*M. gigantea, M. tetradactyla* and *M. tricuspis*) are relatively low (roughly 0.25% p.a.) and would not in themselves lead to reductions in line with the guidelines in Annex 1 of *Res. Conf. 9.24 (Rev. CoP16)*, particularly as all species are known to occur in modified habitats. Given their low productivity, hunting is very likely to have an impact on populations of all species. There is evidence that hunting intensity for pangolins in general in Africa has increased markedly in recent years. However, there is insufficient information to determine whether this has led to declines in line with the guidelines in the Resolution. There is therefore insufficient information to determine whether any of the species meets the criteria for inclusion in Appendix I.

Reviewers: D. Pietersen, C. Waterman and C. Shepherd.

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Transfer of Barbary Macaque *Macaca sylvanus* from Appendix II to Appendix I

Proponents: European Union and Morocco

Summary: The Barbary Macaque *Macaca sylvanus* is a medium-sized monkey that occurs in northern Algeria and Morocco in North Africa and also as a relatively small (ca. 200) semi-wild population on Gibraltar (United Kingdom) presumed to have been introduced there. Its distribution is discontinuous. The largest population is found in the Middle Atlas of Morocco, with smaller populations in the High Atlas and Rif in Morocco and at a number of scattered sites in the Grand Kabylie and Petite Kabylie, and the Chiffa gorges in Algeria. The species occurs in a variety of wooded habitats but is now largely confined to montane forests and inaccessible scrub-clad rocky areas and gorges; altitudinal range is from sea level to 3500m. Females mature at between 3.5 and four years of age and give birth generally to a single young with an average interbirth interval of 1.3 years. Generation time is calculated as eight years[1]. The species occurs in a number of protected areas.

Habitat loss, alteration and fragmentation are believed to be the principal factors affecting the species. Illegal collection of live young is considered to have a significant impact on populations in some areas, particularly where animals are habituated to the presence of humans. There is no evidence that these are entering international trade.

It is widely agreed that the global population has declined. Recent population estimates, based in part on surveys carried out in the Middle Atlas and the main parts of the range in Algeria, are of a global population (excluding Gibraltar) of between ca. 8000 and 11,500, of which 6500 to 8000 are in Morocco and the remainder in Algeria. In the early 1990s the total population was estimated at between 10,000 and 16,000. A 1975 study estimated a global population of between 14,500 and 22,500 at that time, with between 9000 and 17,000 in Morocco and up to 5500 in Algeria[2].

Very little trade (including confiscated specimens) is recorded in the CITES Trade Database: 31 live specimens in total between 2005 and 2014; no trade from a range State has been reported since 2010 and very little before then. Spain reported the import of 15 individuals as “confiscated” in the period 2005 to 2010; eight from Morocco, one from Algeria and six of unknown origin.

Barbary Macaque was categorised as Endangered by IUCN in 2008, and has been included in the Order listing for Primates in Appendix II since 1977. The species is legally protected in Algeria and Morocco.

Analysis: The Barbary Macaque has a reasonably extensive range. Its estimated population (8000 to 11,200) is larger than the guideline figure given for a small population in *Res. Conf. 9.24 (Rev. CoP16*). The population is agreed to have been declining; the best available information indicates that the decline has been of the order of 30% in the past three generations (24 years), which is below the guideline figure for a marked decline in the Resolution. It would appear therefore that the species does not meet the biological criteria for inclusion in Appendix I.

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Background to the African Elephant proposals

The African Elephant *Loxodonta africana* occurs in 38 range States in Africa. It was included in Appendix II in 1977 and transferred to Appendix I in 1989. The populations of Botswana, Namibia and Zimbabwe were transferred to Appendix II in 1997, and the population of South Africa in 2000. These transfers were subject to detailed conditions that were further modified during subsequent meetings of the Conference of the Parties and are at present expressed in annotation 6, agreed at CoP14. The annotation allows for trade in various non-ivory African Elephant specimens and products under a range of conditions, somewhat different for each of the four range States in question. With regard to trade in ivory, it allows for trade in individually marked and certified ekipas incorporated in finished jewellery for non-commercial purposes for Namibia and ivory carvings for non-commercial purposes for Zimbabwe. It also allowed for these four range States to dispose of agreed quantities of stockpiled raw ivory in a one-off sale, under a series of restrictions. One of these is that no further proposals to allow trade in elephant ivory from populations already in Appendix II should be submitted until at least nine years after the date of the single sale of ivory. It also specifies that such further proposals should be dealt with in accordance with Decisions 14.77 and 14.78 (currently as revised at the Sixteenth Conference of the Parties (CoP16)). The sale of ivory in question took place in November 2008; nine years from that time is November 2017.

Decision 14.77 instructed the Standing Committee, assisted by the Secretariat, to propose for approval at the latest at CoP16 a decision-making mechanism for a process of trade in ivory under the auspices of the Conference of the Parties.

Decision 14.77 was not implemented, in that no decision-making mechanism for a process of trade in ivory was submitted by the Standing Committee to CoP16 for approval. This Decision was deleted at CoP16. Instead, the CoP agreed Decision 16.55 which directs the Standing Committee, with the assistance of the Secretariat, to propose for approval at the latest at the 17th meeting of the Conference of the Parties (CoP17) a decision-making mechanism for a process of trade in ivory under the auspices of the Conference of the Parties.

The draft summary record of the 66th meeting of the Standing Committee (11-15 January 2016) indicates that Decision 16.55 has also not been implemented, as no decision-making mechanism for a process of trade in ivory will be submitted to CoP17. The Standing Committee intends instead to seek further guidance from the CoP as to how to proceed in this matter. Annotation 6 still contains reference to Decision 14.77.

The original Decision 14.78 instructed the Standing Committee to conduct ongoing comprehensive reviews of the status of the elephant, trade in its specimens and the impact of the legal trade, based on data from Monitoring the Illegal Killing of Elephants (MIKE), the Elephant Trade Information System (ETIS) and the implementation of the Action plan for the control of trade in elephant ivory (formerly Decision 13.26, which has now been integrated into *Res. Conf. 10.10 (Rev. CoP16)* and the African Elephant action plan, developed as directed in Decision 14.75 and adopted by the African Elephant range States in 2010.

Decision 14.78 was substantively revised at CoP15 and CoP16, the revisions shifting responsibility for action from the Standing Committee to other actors, principally the Secretariat. Under the current Decision 14.78 (Rev. CoP16), in preparation for the 65th and 66th meetings of the Standing Committee the Secretariat is instructed, pending the necessary external funding, to:

> produce an updated analysis of MIKE data, pending the availability of adequate new MIKE data;
> invite TRAFFIC to submit an updated analysis of ETIS data and UNEP-WCMC to provide an overview of the latest elephant trade data;
> invite the IUCN/SSC African and Asian Elephant Specialist Groups to submit any new and relevant information on the conservation status of elephants, and on pertinent conservation actions and management strategies;
> invite the African elephant range States to provide information on progress made in the implementation of the *African elephant action plan;*
> on the basis of the information specified above, recommend actions for consideration by the Standing Committee.

Three proposals concerning the African Elephant have been proposed for consideration at CoP17. Proposal 14, submitted by Namibia, is to remove any reference to Namibia in annotation 6 so that the African Elephant population of Namibia would be included in Appendix II with no annotation. Proposal 15, submitted by Namibia and Zimbabwe, would have the same effect on Zimbabwe's African Elephant population. Proposal

16, submitted by 13 Parties, is to transfer from Appendix II to Appendix I the African Elephant populations of Botswana, Namibia, South Africa and Zimbabwe.

To delete the Annotation to the listing of the Namibian African Elephant *Loxodonta africana* population in Appendix II by deleting any reference to Namibia in that Annotation

Proponent: Namibia

Note: *See Background to the African Elephant proposals for a history of the African Elephant under CITES.*

Summary: The Namibian population of the African Elephant was transferred from Appendix I to Appendix II in 1997; it is currently covered by annotation 6. The proponent (Namibia) wishes to delete reference to Namibia in that annotation so that the African Elephant population of Namibia will be in Appendix II with no annotation. Namibia aims to establish a regular form of controlled trade in all elephant specimens from Namibia, including ivory, in support of elephant conservation, including community-based conservation and the maintenance of elephant habitat. The Supporting Statement indicates that revenue from regulated trade will be managed through a trust fund and used exclusively for elephant conservation and community conservation and development programmes within the elephant range.

The most comprehensive and reliable information on African Elephant distribution and population is contained in the African Elephant Database (AED), maintained by the IUCN/SSC African Elephant Specialist Group[1]. The 2013 version gives the range in Namibia as just under 150,000km^2 (the Supporting Statement indicates that African Elephants have a dispersed, wet season range in Namibia of over 100,000km^2)[1]. The most recent population data available in the AED are from the end of 2013. These along with figures from 2002 and 2006 are presented here divided into 'definite', 'probable', 'possible'. The database is being updated with the most recent data available, and a full version will be available at CoP17. Permission has been granted to use in the IUCN/TRAFFIC Analyses of Proposals the most recent figures from Namibia. Because of differences in survey techniques and extent of coverage, figures for different years are not strictly comparable. The most recent data are presented in the form in which they have been submitted to the AED and therefore a total estimate is not given. Data for Namibia's population of elephants from the African Elephant Database[1, 2] are:

2002 – 7769 definite, 1872 probable and 1872 possible;
2006 – 12,531 definite, 3276 probable and 3296 possible;
2013 – 13,684 definite, 2871 probable and 2891 possible;

More recent data are:

Survey Area	Year	Estimate (with 95% CL)	Source
Etosha National Park	2015	2,911 +/- 697	Kilian, 2015
Khaudum National Park & Neighbouring Conservancies	2015	6,413 +/- 2,566	Gibson & Craig, 2015a
Kunene	2011	314 +/- 154	MET, 2012
Mangetti Game Reserve	2014	67	F. Weiss, pers. comm., 2014
Zambezi Region	2015	13,136 +/- 3435	Gibson & Craig, 2015b

The Supporting Statement provides an estimated total of 22,711, all based on 2015 estimates other than that for the Kunene Region (352, based on a 2009 estimate).

The Supporting Statement outlines management measures for African Elephants in Namibia and enforcement controls and compliance with CITES. It indicates that no elephants have been, or will be, killed specifically to obtain ivory or other products for commercial trade. Ivory is recovered from all recorded natural mortalities as well as elephants destroyed as problem animals, and strict national legislation makes it obligatory for the public to hand in any ivory found. It indicates that the level of sport hunting is largely determined by a guideline of 0.5% of the standing population. A national export of 90 trophy hunted specimens per year has been established, to allow for the possibility that the tusks of elephants hunted in one year may be exported the following year. On average 49 trophies (98 tusks) have been exported each year in 2000 to 2015.

The Supporting Statement notes that a computer database of all specimens in storage is maintained with source documentation, and all specimens are marked so as to make them individually recognisable. The annex to CoP17 Doc. 57.6 (Report on the Elephant Trade Information System (ETIS)) notes that Namibia is one of only five CITES Parties (along with Chad, Congo, Thailand and Zimbabwe) to have submitted ivory stock reports with inventory figures in 2015.

Information on the number of elephants recorded as illegally killed in Namibia, and on ivory seizures, is included as annexes to the Supporting Statement.

CoP17 Doc. 57.5 (Report on Monitoring the Illegal Killing of Elephants (MIKE)) contains the most up-to-date synthesised information on illegal killing of elephants, based on information from 2003 until the end of 2015. It reports on the proportion of illegally killed elephants (PIKE) at 58 sites in 30 countries in Africa and 27 sites in 13 countries in Asia. A PIKE level of 0.5 or lower is generally considered sustainable. In its analysis of sub-regional trends in Africa, the report observes (para 14) that the southern African sub-region (Angola, Botswana, Malawi, Mozambique, Namibia, South Africa, Swaziland, Zambia, Zimbabwe) is the only one of the four African sub-regions where the estimated PIKE has not exceeded the 0.5 level in the period 2003 to 2015. It is difficult to estimate poaching impact at the site level, especially in sites that do not report sufficiently large numbers of carcasses, or where there may be indications of bias in reported PIKE levels[3].

The annex to CoP17 Doc, 57.6 presents an analysis of illegal ivory trade, based on data in ETIS. Part of this is a cluster analysis of 55 countries or territories divided into 13 groups with similar characteristics. Namibia forms part of group eight, along with Botswana and Zimbabwe. The report's analysis of this group is as follows:

> "As in the CoP16 analysis, three of the four African Elephant range States whose elephant populations were transferred to Appendix II in 1997 fall in the same group. These countries regularly report data to ETIS. In terms of all data which implicate these countries in an ivory seizure, this southern African grouping reflects middle range values in terms of mean number of seizures and the mean weight of ivory seized. The measure for assessing the presence of organised crime stands at zero which is indisputably a good sign. Governance indicators are mixed, however, with the rule of law score problematic and suggesting the presence of corruption, but the relatively high law enforcement ratio partially mitigates that concern. Indeed, as before, Zimbabwe is the country that pulls the rule of law score down, indicating far greater governance challenges exist in that country, but it is worth noting that Namibia's scores have also dropped too. The domestic ivory market score is low, reflecting the complete absence of a market in Botswana and a very low level of trade in Namibia. Again, Zimbabwe is the exception with the tenth largest ivory market of any country in this analysis."

The Supporting Statement draws attention to the failure to implement Decision 14.77, concerning a decision-making mechanism for a process of trade in ivory, which is an integral part of annotation 6 covering the Appendix II African Elephant populations. It states that if such a mechanism is not approved at CoP17, Namibia will regard current annotation 6 as invalid.

Analysis: The Namibian population of African Elephant was transferred from Appendix I to Appendix II, under a series of constraints set out in an annotation (6). Acceptance of the current proposal would delete this annotation as it refers to Namibia. There are no explicit guidelines in *Res. Conf. 9.24 (Rev. CoP16)* as to how to deal with a proposal to amend or delete an annotation for an Appendix-II listed species. However, these constraints can be interpreted as special measures under the terms of the precautionary measures in Annex 4 of *Res. Conf. 9.24 (Rev. CoP16)*. It would be appropriate to examine whether these precautionary measures are still met under the proposed change.

The Namibian African Elephant population does not have a restricted range, is not small, and is not undergoing a marked decline. It does not therefore appear to meet the biological criteria for inclusion in Appendix I set out in Annex 1 of *Res. Conf. 9.24 (Rev. CoP16).*

Regarding the further precautionary measures in Annex 4, the proposal should include a special measure (as envisaged in para A 2 a) iii)) set out in the Supporting Statement. The Supporting Statement indicates that no African Elephants will be harvested for commercial trade. This may be interpreted as a special measure. The Parties would also need to be satisfied that appropriate enforcement controls and compliance with the requirements of the Convention are in place. Details of enforcement controls are set out in the Supporting Statement. Information from ETIS and MIKE indicates that controls in place for the time periods analysed (up to 2014 for ETIS and 2015 for MIKE).

Sources for population estimates:

Gibson, D. S. C., & Craig, G. C. (2015a). Aerial Survey of Elephants & Other Wildlife in Khaudum National Park & Neighboring Conservancies: October 2015. Ministry of Environment & Tourism, Namibia.

Gibson, D. S. C., & Craig, G. C. (2015b). Aerial Survey of Elephants and Other Wildlife in Zambezi Region September/October 2015. WWF.

Kilian, J.W. (2015). Aerial Survey of Etosha National Park. Internal Report to the Ministry of Environment and Tourism: September 2015.

Ministry of Environment and Tourism. (2012). Countrywide survey of Elephants in Namibia. Namibia: Ministry of Environment and Tourism.

Weiss, F. (2014). Personal Communication: Information on the elephants of Mangetti. E-mail to C. Thouless, 11 August 2014.

Reviewers of summary information only: D. Skinner and T. Milliken.

References:
The information not referenced in the Summary section is from the Supporting Statement.

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[1] IUCN/SSC African Elephant Specialist Group (2013) Continental Totals Provisional African Elephant Population Estimates: update to 31 Dec 2013. http://www.elephantdatabase.org/preview_report/2013_africa_final/2013/Africa. Viewed on 5th July 2016.

[2] Blanc, J.J., Thouless, C.R., Hart, J.A., Dublin, H.T., Douglas-Hamilton, I., Craig, C.G. & Barnes, R.F.W. (2003) African Elephant Status Report 2002: An update from the African Elephant Database. IUCN/SSC African Elephant Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

[3] CoP17 Doc 57.5 Report on Monitoring The Illegal Killing of Elephants (MIKE) https://cites.org/sites/default/files/eng/cop/17/WorkingDocs/E-CoP17-57-05.pdf Viewed on 26th July 2016.

Amend the present Appendix-II listing of the population of Zimbabwe of African Elephant *Loxodonta africana* by removing the annotation in order to achieve an unqualified Appendix II listing

Proponents: Namibia and Zimbabwe

Note: *See Background to the African Elephant proposals for a history of the African Elephant under CITES.*

Summary: The Zimbabwean population of the African Elephant *Loxodonta africana* was transferred from Appendix I to Appendix II in 1997; it is currently covered by annotation 6. The proponents (Namibia and Zimbabwe) seek to achieve an unqualified Appendix-II listing of the Zimbabwean African Elephant population, arguing that effective and sustainable conservation of Zimbabwe's elephants is dependent on establishing regular open market sales of elephant ivory to fund management and enforcement actions.

The most comprehensive and reliable information on African Elephant distribution and population is contained in the African Elephant Database (AED), maintained by the IUCN/SSC African Elephant Specialist Group[1]. The 2013 version gives the range as just under 77,000km^{2} [1] (the Supporting Statement indicates that the four main elephant populations have a combined range of ca. 63,000km^{2}). The most recent population data available in the AED are from the end of 2013. These along with figures from 2002 and 2006 are presented here divided into 'definite', 'probable', 'possible' and 'speculative'. The database is being updated with the most recent data available, and a full version will be available at CoP17. Permission has been granted for use in the IUCN/TRAFFIC Analyses of Proposals the most recent figures from Zimbabwe. Because of differences in survey techniques and extent of coverage, figures for different years are not strictly comparable. The most recent data are presented in the form in which they have been submitted to the AED and therefore a total estimate is not given.

Data for Zimbabwe's population of elephants from the African Elephant Database[1] are:

2002 – 81,555 definite, 7039 probable, 7373 possible and 291 speculative;
2006 – 84,416 definite, 7033 probable, 7367 possible and 291 speculative;
2013 – 67,954 definite, 6974 probable, 6974 possible and 14,730 speculative (all provisional estimates);

More recent data are:

Survey Area	Year	Estimate (with 95% CL)	Source
Gonarezhou and SVC	2014	11,120 +/-2753	Dunham & van der Westhuizen, 2015
Greater Mapungubwe Trans- frontier Conservation Area	2014	212	Selier & Page, 2014
Northwest Matabeleland	2014	53,991 +/-7711	Dunham et al., 2015a
Sebungwe	2014	3407 +/- 1215	Dunham et al., 2015b
Various Areas	2014	2143	Dunham, 2015
Zambezi Valley	2014	11,657 +/-2259	Dunham et al., 2015c

The Supporting Statement provides a 2014 estimate of 80,507.

The Supporting Statement indicates that Zimbabwe adopts an experimental, rather than programmatic, adaptive management approach towards its elephants, involving a devolutionary policy that allows its primary stakeholders (those with wildlife on their land) to experiment with elephant management. It alludes to a method of quota setting for elephants based on monitoring of mean tusk weight of trophies which takes into account the long response time of elephant populations to any change in their management regime. The Supporting Statement includes an analysis of legal ivory trade from Zimbabwe for 1980 to 2014, as reported in the CITES Trade Database, noting discrepancies between import and export data and showing that recorded annual exports for 2012 and 2013 exceeded 20t, the first time this has occurred since 1990.

The Supporting Statement indicates that the average mortality due to illegal hunting is 4.5% of the total population noting that the populations in two of the main areas of elephant distribution in the country are increasing while in two others they are declining rapidly towards extinction.

CoP17 Doc. 57.5 (Report on Monitoring the Illegal Killing of Elephants (MIKE)) contains the most up-to-date synthesised information on illegal killing of elephants, based on information from 2003 until the end of 2015. It reports on the proportion of illegally killed elephants (PIKE) at 58 sites in 30 countries in Africa and 27 sites in 13 countries in Asia. A PIKE level of 0.5 or lower is generally considered sustainable. In its analysis of sub-regional trends in Africa, the report observes (para 14) that the southern African sub-region (Angola, Botswana, Malawi, Mozambique, Namibia, South Africa, Swaziland, Zambia, Zimbabwe) is the only one of the four African sub-regions where the estimated PIKE has not exceeded the 0.5 level in the period 2003 to 2015. It is difficult to estimate poaching impact at the site level, especially in sites that do not report sufficiently large numbers of carcasses, or where there may be indications of bias in reported PIKE levels[2].

The Supporting Statement cites an estimate of 439t of ivory illegally traded in the period 2002 to 2014 in Zimbabwe, compared with legal ivory production of 180t and trophy hunting of 74t in the same period.

The annex to CoP17 Doc. 57.6 presents an analysis of illegal ivory trade, based on data in ETIS. Part of this is a cluster analysis of 55 countries or territories divided into 13 groups with similar characteristics. Zimbabwe forms part of group eight, along with Botswana and Namibia. The report's analysis of this group is as follows:

> "As in the CoP16 analysis, three of the four African Elephant range States whose elephant populations were transferred to Appendix II in 1997 fall in the same group. These countries regularly report data to ETIS. In terms of all data which implicate these countries in an ivory seizure, this southern African grouping reflects middle range values in terms of mean number of seizures and the mean weight of ivory seized. The measure for assessing the presence of organised crime stands at zero which is indisputably a good sign. Governance indicators are mixed, however, with the rule of law score problematic and suggesting the presence of corruption, but the relatively high law enforcement ratio partially mitigates that concern. Indeed, as before, Zimbabwe is the country that pulls the rule of law score down, indicating far greater governance challenges exist in that country, but it is worth noting that Namibia's scores have also dropped too. The domestic ivory market score is low, reflecting the complete absence of a market in Botswana and a very low level of trade in Namibia. Again, Zimbabwe is the exception with the tenth largest ivory market of any country in this analysis."

The proponents argue that paragraph h) of annotation 6, which states that no proposals to allow trade in elephant ivory from populations already in Appendix II shall be submitted before nine years after the sale of ivory (which took place in November 2008) cannot override the right of Parties under the Convention to submit an amendment proposal at any time.

Analysis: The Zimbabwean population of African Elephant was transferred from Appendix I to Appendix II, under a series of constraints set out in an annotation (6). Acceptance of the current proposal would delete this annotation as it refers to Zimbabwe. There are no explicit guidelines in *Res. Conf. 9.24 (Rev. CoP16)* as to how to deal with a proposal to amend or delete an annotation for an Appendix-II listed species. However, these constraints can be interpreted as special measures under the terms of the precautionary measures in Annex 4 of *Res. Conf. 9.24 (Rev. CoP16)*. It would be appropriate to examine whether these precautionary measures are still met under the proposed change.

The Zimbabwean African Elephant population does not have a restricted range, is not small, and is not undergoing a marked decline. It does not therefore appear to meet the biological criteria for inclusion in Appendix I set out in Annex 1 of *Res. Conf. 9.24 (Rev. CoP16).*

Although reference is made to a possible quota-setting method, no specific export quota or other special measure is proposed in the Supporting Statement. Under the precautionary measures set out in Annex 4 of *Res. Conf. 9.24 (Rev. CoP16),* Parties would therefore need to be satisfied that Zimbabwe is implementing the requirements of the Convention, particularly Article IV, and that appropriate enforcement controls and compliance with the requirements of the Convention are in place.

The Supporting Statement indicates that Zimbabwe adopts an experimental, adaptive approach to management of its African Elephants. It is not possible to determine *a priori* if such an approach would be effective in implementing Article IV if this proposal were accepted. Regarding enforcement controls and compliance, the Supporting statement itself, as well as analysis from ETIS in the annex to CoP17 Doc. 57.6 indicates that this may be problematic in some areas. It is likely that in this case the precautionary measures may not be met.

Sources for population estimates:

Dunham, K. C. (2015). National Summary of Aerial Survey Results for Elephant in Zimbabwe: 2014. Harare, Zimbabwe: Parks and Wild Life Management Authority.

Dunham, K. M., Mackie, C. S., Nyaguse, G., & Zhuwau, C. (2015a). Aerial Survey of Elephants and other Large Herbivores in north-west Matabeleland (Zimbabwe): 2014. Harare, Zimbabwe: Parks and Wild Life Management Authority.

Dunham, K. C., Mackie, C. S., Nyaguse, G., & Zhuwau, C. (2015b). Aerial Survey of Elephants and other Large Herbivores in the Sebungwe (Zimbabwe): 2014. Harare, Zimbabwe: Parks and Wild Life Management Authority.

Dunham, K. M., Mackie, C. S., & Nyaguse, G. (2015c). Aerial Survey of Elephants and other Large Herbivores in the Zambezi Valley (Zimbabwe): 2014. Harare, Zimbabwe.

Dunham, K. M., & van der Westhuizen, H. F. (2015). Aerial Survey of Elephants and other Large Herbivores in Gonarezhou National Park and Save Valley Conservancy (Zimbabwe): 2014. Frankfurt Zoological Society.

Selier, J., & Page, B. (2014). Dry season fixed-wing aerial survey of large mammals in the Northern Tuli Game Reserve and Mapungubwe National Park and of elephants in the Greater Mapungubwe Transfrontier Conservation Area, Botswana, South Africa and Zimbabwe, August 2014. Central Limpopo River Valley Elephant Research Project and the University of KwaZulu-Natal.

Reviewers of summary information only: D. Skinner and T. Milliken.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] IUCN/SSC African Elephant Specialist Group (2013) Continental Totals Provisional African Elephant Population Estimates: update to 31 Dec 2013. http://www.elephantdatabase.org/preview_report/2013_africa_final/2013/Africa. Viewed on 5th July 2016.
[2] CoP17 Doc 57.5 Report on Monitoring the Illegal Killing of Elephants (MIKE) https://cites.org/sites/default/files/eng/cop/17/WorkingDocs/E-CoP17-57-05.pdf Viewed on 26th July 2016.

Include all populations of African Elephant *Loxodonta africana* in Appendix I through the transfer from Appendix II to Appendix I of the populations of Botswana, Namibia, South Africa and Zimbabwe

Proponents: Benin, Burkina Faso, Central African Republic, Chad, Ethiopia, Kenya, Liberia, Mali, Niger, Nigeria, Senegal, Sri Lanka and Uganda

Note: *See Background to the African Elephant proposals for a history of the African Elephant under CITES.*

Summary: This proposal applies only to the African Elephant population of four contiguous southern African countries: Botswana, Namibia, South Africa and Zimbabwe. The most comprehensive and reliable information on African Elephant distribution and population is contained in the African Elephant Database (AED), maintained by the IUCN/SSC African Elephant Specialist Group[1]. This gives the combined area of distribution of the species in the four countries considered here as ca.350,000km^2 [1] The most recent population data available in the AED are from the end of 2013. These along with figures from 2002 and 2006 are presented here divided into 'definite', 'probable', 'possible' and 'speculative' (not all categories are used in all range States). The database is being updated with the most recent data available, and a full version will be available at CoP17. Permission has been granted to use in the IUCN/TRAFFIC Analyses of Proposals the most recent figures from Botswana, Namibia, South Africa and Zimbabwe (more recent figures for Namibia and Zimbabwe are also presented in the Supporting Statements for Proposals 14 and 15 respectively). Because of differences in survey techniques and extent of coverage, figures for different years are not strictly comparable. The most recent data are presented in the form in which they have been submitted to the AED.

Botswana:
2002 – 100,629 definite; 21,237 probable and 21,237 possible;
2006 – 133,829 definite, 20,829 probable and 20,829 possible;
2013 – 133,453 definite, 20,818 probable and 20,818 possible (all provisional estimates);

Namibia:
2002 – 7769 definite, 1872 probable and 1872 possible;
2006 – 12,531 definite, 3276 probable and 3296 possible;
2013 – 13,684 definite, 2871 probable and 2891 possible (all provisional estimates);

South Africa:
2002 – 14,071 definite and 855 possible;
2006 – 17,847 definite, 638 possible and 22 speculative;
2013 – 20,260 definite and 4767 possible (provisional estimates);

Zimbabwe:
2002 – 81,555 definite, 7039 probable, 7373 possible and 291 speculative;
2006 – 84,416 definite, 7033 probable, 7367 possible and 291 speculative;
2013 – 67,954 definite, 6974 probable, 6974 possible and 14,730 speculative (all provisional estimates).

More recent information is as follows:

Survey Area	Year	Estimate (with 95% CL)	Source
Botswana			
Northern Botswana	2016	129,939 +/-12,514	Chase *et al.*, 2015
Greater Mapungubwe Trans- frontier Conservation Area	2014	890	Selier & Page, 2014
Namibia			
Etosha National Park	2015	2,911 +/- 697	Kilian, 2015
Khaudum National Park & Neighbouring Conservancies	2015	6,413 +/- 2,566	Gibson & Craig, 2015a
Kunene	2011	314 +/- 154	MET, 2012
Mangetti Game Reserve	2014	67	F. Weiss, pers. comm., 2014
Zambezi Region	2015	13,136 +/- 3435	Gibson & Craig, 2015b
South Africa			
Hluhluwe Imfolozi Game Reserve*	2015	700	EKZNW, 2016
Ithala Game Reserve*	2015	162	EKZNW, 2016

Survey Area	Year	Estimate (with 95% CL)	Source
St. Lucia Reserves*	2015	110	EKZNW, 2016
Tembe Elephant Park*	2015	220-230	EKZNW, 2016
uMkhuze Game Reserve*	2015	90	EKZNW, 2016
Marakele National Park	2012	171	Ferreira *et al.*, 2012
Addo Elephant National Park	2012	595	Ferreira *et al.*, 2012
Kruger National Park	2015	17,086	Ferreira *et al.*, 2015
Great Fish River Provincial Reserve*	2015	2	J. Selier, pers. comm., 2016
Kariega Private Game Reserve*	2015	41	J. Selier, pers. comm., 2016
Knysna Forest*	2015	2	J. Selier, pers. comm., 2016
Kwandwe Private Game Reserve*	2015	57	J. Selier, pers. comm., 2016
Atherstone Provincial Nature Reserve*	2015	105	M. Garai, pers. comm., 2016
Madikwe Provincial Reserve*	2015	1006	M. Garai, pers. comm., 2016
Pilanesberg Provincial Reserve*	2015	240	M. Garai, pers. comm., 2016
Balule, Timbavati Umbabat and Klaserie*	2015	2772	M. Garai, pers. comm., 2016
Letaba Provincial Nature Reserve*	2015	621	M. Garai, pers. comm., 2016
Makuya National Park*	2015	9	M. Garai, pers. comm., 2016
Maremani*	2015	64	M. Garai, pers. comm., 2016
Mthetomusha Provincial Reserve*	2015	57	M. Garai, pers. comm., 2016
Songimvelo Game Reserve*	2015	105	M. Garai, pers. comm., 2016
Other Private Reserves*	2015	2482	M. Garai, pers. comm., 2016
Manyeleti Game Reserve	2009	222	SANParks, 2009
Mapungubwe Ecosystem	2014	347	Selier & Page, 2014
Zimbabwe			
Gonarezhou and SVC	2014	11,120 +/-2,753	Dunham & van der Westhuizen, 2015
Greater Mapungubwe Trans- frontier Conservation Area	2014	212	Selier & Page, 2014
Northwest Matabeleland	2014	53,991 +/-7,711	Dunham *et al.*, 2015a
Sebungwe	2014	3,407 +/- 1,215	Dunham *et al.*, 2015b
Various Areas	2014	2,143	Dunham, 2015
Zambezi Valley	2014	11,657 +/-2,259	Dunham *et al.*, 2015c

**Due to the absence of a detailed report (with methodology), these estimates have been entered as informed guesses, which carry less weight than data from systematic survey efforts*

CoP17 Doc. 57.5 (Report on Monitoring the Illegal Killing of Elephants (MIKE)) contains the most up-to-date synthesised information on illegal killing of elephants, based on information from 2003 until the end of 2015. It reports on the proportion of illegally killed elephants (PIKE) at 58 sites in 30 countries in Africa and 27 sites in 13 countries in Asia. A PIKE level of 0.5 or lower is generally considered sustainable. In its analysis of sub-regional trends in Africa, the report observes (para 14) that the southern African sub-region (Angola, Botswana, Malawi, Mozambique, Namibia, South Africa, Swaziland, Zambia, Zimbabwe) is the only one of the four African subregions where the estimated PIKE has not exceeded the 0.5 level in the period 2003 to 2015. It is difficult to estimate poaching impact at the site level, especially in sites that do not report sufficiently large numbers of carcasses, or where there may be indications of bias in reported PIKE levels[2].

The Supporting Statement of the proposal deals extensively with the wider African Elephant population, which is not the subject of the amendment proposal. It draws attention to the high levels of illegal killing of elephants that have been recorded (chiefly through the MIKE programme) in many parts of the range since 2006 (see Doc. CoP17 57.5), associated with elevated levels of illegal trade in ivory recorded from 2008 onwards, as indicated by seizure data contained in the Elephant Trade Information System (ETIS) (see CoP17 Doc. 57.6). The proponents argue that transferring the Appendix-II African Elephant population to Appendix I will indicate that the CITES Parties do not intend to allow commercial trade in ivory in the future, and that this will serve as a disincentive for the illegal killing of elephants, thereby enhancing the conservation status of this species in its range as a whole, and also benefitting the Appendix-I listed Asian Elephant *Elephas maximus*.

Analysis: Regarding the impact of this proposal on elephant populations elsewhere, there is no provision to address this question in any guidelines or criteria under the Convention and it will therefore not be considered further here.

The African Elephant population of Botswana, Namibia, South Africa and Zimbabwe does not appear to meet the biological criteria for inclusion in Appendix I set out in *Res. Conf. 9.24 (Rev. CoP16)*. It does not have a restricted range, nor is its population small or undergoing a marked decline.

Annex 3 of *Res. Conf. 9.24 (Rev. CoP16)* states that listing of a species in more than one Appendix should be avoided in general in view of the enforcement problems it creates. It adds that if split-listing does occur, this should generally be on the basis of national or regional populations, rather than subspecies.

Sources for population estimates:

Botswana

Chase, M., Schlossberg, S., Landen, K., Sutcliffe, R., Seonyatseng, E., Keitsile, A., & Flyman, M. (2015). Dry season aerial survey of elephants and wildlife in northern Botswana: July – October 2014. Elephants Without Borders, the Department of Wildlife and National Parks (Botswana), Great Elephant Census.

Selier, J., & Page, B. (2014). Dry season fixed-wing aerial survey of large mammals in the Northern Tuli Game Reserve and Mapungubwe National Park and of elephants in the Greater Mapungubwe Transfrontier Conservation Area, Botswana, South Africa and Zimbabwe, August 2014. Central Limpopo River Valley Elephant Research Project and the University of KwaZulu-Natal.

Namibia

Gibson, D. S. C., & Craig, G. C. (2015a). Aerial Survey of Elephants & Other Wildlife in Khaudum National Park & Neighboring Conservancies: October 2015. Ministry of Environment & Tourism, Namibia.

Gibson, D. S. C., & Craig, G. C. (2015b). Aerial Survey of Elephants and Other Wildlife in Zambezi Region September/October 2015. WWF.

Kilian, J.W. (2015). Aerial Survey of Etosha National Park. Internal Report to the Ministry of Environment and Tourism: September 2015.

Ministry of Environment and Tourism. (2012). Countrywide survey of Elephants in Namibia. Namibia: Ministry of Environment and Tourism.

Weiss, F. (2014). Personal Communication: Information on the elephants of Mangetti. E-mail to C. Thouless, 11 August 2014.

South Africa

EKZNW. (2016). Personal communication from Pete Ruinard. E-mail to H. Dublin, 28 April 2016.

Ferreira, S., Greaver, C., & Simms, C. (2015). Elephant Management Update (02/2015): Elephant survey of the Kruger National Park. South African National Parks.

Ferreira, S., Pienaar, D., Freitag-Ronaldson, S. and Magome, H. (2012). An update on managing the effects of elephants in National Parks. Skukuza, South Africa: South Africa National Parks.

Garai, M. (2016). Personal Communication: ESAG DATABASE Update 2015/2016: Compiled by Marion E. Garai. E-mail to T. Daniel, 17 May 2016.

SANParks (2009). Elephant estimates in Addo Elephant, Kruger, Marakele, and Mapungubwe National Parks, 2005-2009. Pretoria, South Africa: SANParks.

Selier, J. (2016). Personal communication: Information on elephant populations in South Africa. Email to T. Daniel. 25 May 2016.

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Reviewers of summary information only: D. Skinner and T. Milliken.

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Transfer of Peregrine Falcon *Falco peregrinus* from Appendix I to Appendix II

Proponent: Canada

Summary: The Peregrine Falcon *Falco peregrinus* is a falcon in the order Falconiformes with a global distribution encompassing over 200 range States across the Americas, Africa, Asia, Australasia and Europe. It was included in Appendix I in 1977. Currently all Falconiformes other than those in Appendix I are in Appendix II.

The extent of occurrence is now estimated to be nearly 40 million km^2, and a very preliminary estimate of the global population size is 230,000 to 440,000 mature individuals, although further validation of this estimate is needed, and previous estimates placed it closer to 90,000 mature individuals[1]. The population overall is said to be stable, having undergone an increase in North America and increasing in Europe. There are some regional exceptions to the general global trend of stable or increasing populations. In Turkey, populations decreased over the decade from 1990 to 2000 while in central Europe some small sub-populations have not recovered from earlier declines. The global population was classified by BirdLife for the IUCN Red List as being of Least Concern (2015).

Falco peregrinus underwent severe declines in the mid-20th century owing to the widespread use at that time of pesticides containing dichloro-diphenyl-trichloroethane (DDT) which reduced the reproductive success, leading to a significant reduction in its distribution and extirpations of some populations[2]. Re-introduction programmes and restrictions on pesticide use in some areas of its range have allowed the species to recover[2], although significant further efforts are needed to fully restore the species across its former range[1]. Current factors impacting *F. peregrinus* populations are likely to include environmental toxins, habitat destruction and alteration, illegal killing and take from the wild.

The species is traded internationally for falconry, and also for re-introduction purposes. According to the CITES Trade Database, between 2000 and 2014 a total of 4674 live *F. peregrinus* were traded, the majority declared as captive-bred (source code C/D = 3667 birds). During the same time period, 665 wild birds were exported, mainly for the purpose of re-introduction or introduction into the wild, with a smaller number for personal use. Illegal trade of wild *F. peregrinus* does occur, but is not significant in relation to the population size[3].

The commercial captive-breeding of raptors in general is of growing economic importance. Much of this growth has come from producers in the United Kingdom, Germany, and other European countries supplying Middle Eastern consumers who are driving both the demand and prices worldwide; some Middle Eastern countries are becoming major producers in addition to being large centres of demand[4]. Currently the global commercial demand for *F. peregrinus* is very largely met by captive-bred birds although some are reportedly taken from the wild for domestic use in consumer countries including Qatar, Saudi Arabia and United Arab Emirates[4]. Such take is often illegal.

A survey of 21 current key trading countries found that all (with the possible exception of Mongolia) had controls on wild-take of *F. peregrinus* and falconry either through specific regulation or more general wildlife regulations. The Supporting Statement and Information Document submitted by Canada contain more detailed information on national regulation. At least 13 of the countries do not currently permit wild-harvest. Most of the key trading countries indicated that national-level controls would not change as a result of a transfer of *F. peregrinus* to Appendix II[5]. However, responses were not received from some countries which are large exporters/importers (e.g. Kazakhstan, Japan).

There may be some demand in international trade for wild birds following a transfer to Appendix II, as breeders look for new bloodlines and falconers become interested in obtaining them because they were previously unavailable in trade[3,6]. However, all major consuming countries are also range States where it appears in general that domestic regulations concerning take from the wild would be unlikely to change following a transfer to Appendix II.

Analysis: The available information indicates that *Falco peregrinus* does not meet the biological criteria for inclusion in Appendix I: it has an extremely wide distribution and a large and stable population.

Regarding the precautionary measures outlined in Annex 4 of *Res. Conf. 9.24 (Rev. CoP16),* it is likely that a transfer will stimulate trade in wild *F. peregrinus***.** However, the impact on the wild population as a whole will likely be minimal, as there is already a well-established captive-breeding trade which is able to largely satisfy current market demands.

Given the species is present in over 200 countries it is hard to determine whether management and appropriate enforcement and compliance controls in each is such that Parties can be satisfied with implementation by the range States of Article IV. However, the majority of current key trading countries indicated that national-level controls would not change as a result of a transfer of *F. peregrinus* to Appendix II. The species is unlikely to enter commercial trade from the great majority of range States. It seems likely, therefore, that precautionary measures are met in the greater part of the range and would be proportionate to the anticipated risks to the species.

Reviewers of summary information only: R. Watson.

References:

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Transfer of the Helmeted Honeyeater *Lichenostomus melanops cassidix* from Appendix I to II

Proponent: Australia

Summary: The Helmeted Honeyeater *Lichenostomus melanops cassidix* is the largest and most brightly coloured subspecies of the Yellow-tufted Honeyeater, a bird endemic to Australia. It was previously distributed in an area of 2000-3000 km^2 in south-central Victoria but it is now limited to a small section of creek in an area of less than 5km^2. Because of conservation measures its population has been growing since 2011 but still remains at under 100 mature individuals. The small population size and limited range make the subspecies vulnerable to natural events and disease. Other adverse factors include poor habitat, predation and harassment by Bell Miner *Manorina melanophrys*, which reduces breeding success and competes for food.

The subspecies is listed as critically endangered under Australia's Environment Protection and Biodiversity Act 1999, which regulates trade in CITES-listed and Australian native wildlife and their products. Export of a live Australian native mammal, bird, reptile or amphibian is strictly prohibited for commercial purposes but they may be exported for specific non-commercial purposes (e.g. for research, education, exhibition or a pet bird). Permits are required for import and export. The taxon is one of the most intensively managed in Victoria, Australia. The long term objective of management is to increase the population to a minimum of 1000 individuals.

The subspecies was included in CITES Appendix I in 1975. No other *Lichenostomus* is included in the CITES Appendices. Very limited trade in specimens for scientific purposes is recorded in the CITES trade database, the most recent of which (of non-viable eggs) was for research to enhance the conservation prospects of the taxon. There have been no reports of illegal trade. The species was included in the Periodic Review.

Analysis: The Helmeted Honeyeater *Lichenostomus melanops cassidix* has a restricted range and a population, which although increasing is very small. On this basis it would appear still to meet the biological criteria for inclusion in Appendix I in Annex 1 of *Res. Conf 9.24 (Rev. CoP16).* However, the only reported trade has been in specimens for scientific purposes and there are no indications of illegal trade of any commercial demand. It is highly unlikely that its transfer to Appendix II would stimulate trade in it or any Appendix-I listed species. In the event of a transfer to Appendix II, no commercial trade would be permitted under Australian legislation. The anticipated risks to the taxon of such a transfer would appear to be negligible.

Transfer of African Grey Parrot *Psittacus erithacus* from Appendix II to Appendix I

Proponents: Angola, Chad, European Union, Gabon, Guinea, Nigeria, Senegal, Togo and United States of America

Summary: The African Grey Parrot *Psittacus erithacus* is a medium-sized frugivorous parrot from forested parts of Western and Central Africa. It occurs in 22 or 23 range States and has a range estimated at around three million km^2, of which nearly 90% is in Central Africa (from eastern Nigeria and Cameroon eastwards), around half of this in Democratic Republic of the Congo. It has been included in Appendix II under the general listing of Psittaciformes since 1981.

Typically inhabiting dense, moist lowland forest, it may also occur in or at forest edges, clearings, gallery forest, mangroves, wooded savannah, cultivated areas and gardens. The species often forms large communal roosts of hundreds, sometimes thousands, of birds and may also congregate in large numbers at mineral licks[1]. Breeding is dispersed or loosely colonial[2]. The nest is in a tree cavity usually between 10 and 30m above the ground. In captivity birds have a mean lifespan of around 45 years and first breed at about five years of age; from this generation time is estimated at just over 15 years. It is estimated that in the wild 15 to 30% of the population breeds in any one year. Clutches comprise three to five eggs; wild productivity has been estimated at around 0.4 chicks/nest per year[1], or one to 1.8 fledglings per year.

Population density is very variable: estimates in different areas and different habitats range from 0.15 birds per km^2 to two breeding pairs per km^2. Combining these figures with estimates of habitat extent, a very rough estimation of between ca. 700,000 and 13 million birds in total was derived in 2008, with 160,000 to 360,000 in West Africa and the remainder in Central Africa[2].

Information on changes in population is patchy, not well quantified and often anecdotal. There are indications of local declines, some of them marked, over the past two to three decades in countries including Angola, Burundi, Democratic Republic of the Congo, Gabon, Guinea-Bissau, Kenya, Nigeria, Republic of the Congo and Rwanda, and more widespread marked declines in Ghana[3] and Guinea[1, 4]. A recent country-wide estimate in Cameroon of around 200,000 is lower than one made in the mid-1990s (300,000 to 500,000); however the basis of both these estimates has been questioned and the two are not comparable[4]. The 2013 BirdLife assessment for the IUCN Red List noted that the rate of decline was hard to quantify, but that a rate of 30 to 49% over three generations might be a conservative estimate. The species was classified in the IUCN Red List as Vulnerable (2013) on this basis.

There is essentially no information on population status or trends for a very large proportion of the range in Central Africa. Population declines here have been inferred from habitat loss and harvest for international trade. Loss and fragmentation of forest cover is generally agreed to have affected African Grey Parrot populations although quantitative data linking the two are lacking. FAO figures indicate that, as a very rough estimation, some 8% of forest cover has been lost in countries within the range of the species between 1990 and 2010. However, forest loss has been considerably lower in the Central African basin, where the bulk of the population is believed to occur, with 4% loss from 1990 to 2010, or roughly 0.2% per year in Democratic Republic of the Congo.

The African Grey Parrot is a popular pet. Wild-caught birds to supply the demand have featured prominently in international trade. Records from importers in the CITES Trade Database indicate fluctuating levels of trade since the early 1980s, averaging around 35,000 birds per year from 1982 to 2006; fluctuations were in part due to changing trade patterns based on introduction of stricter domestic measures in importing countries and regions, notably bans on imports of wild birds into the USA in 1992 and into the European Union in 2005. Declared trade in wild-caught birds since then has been lower, averaging around 11,000 birds per year according to importers' records (about half this according to exporters). There are numerous reports of unauthorised or illegal capture and trade, including from Central African parts of the range, but these are not well-quantified. Estimates of post-capture pre-export mortality of wild birds vary, but average 30 to 40%[1].

The CITES Trade Database shows that in recent years South Africa has been reporting the export of large and rapidly increasing numbers of captive-bred African Grey Parrots as captive-bred, rising from some 8000 in 2007 to ca. 29,000 in 2010 to ca. 76,000 in 2014. A recent assessment indicated that there were over 1600 separate breeding facilities for the species in South Africa with, collectively, around 50,000 breeding pairs[5].

The species is vulnerable to trapping at roosts and mineral licks where it tends to congregate, and there are

reports of population declines at such sites where these have been targeted[5]. However there is very little information on the intensity of trapping or its impact in large parts of the range.

Legal status varies across the range. In some countries it is completely protected, in others partially. The species has been included in the Review of Significant Trade three times (in the 1980s, in 2004 and 2011) resulting in recommendations for various exporting range States. Currently Cameroon and Democratic Republic of the Congo have published annual export quotas (3000 and 5000 respectively)[5]. In 2015 the CITES Standing Committee recommended that all Parties suspend imports of African Grey Parrots from Democratic Republic of the Congo, the major exporter in recent years, because of persistent irregularities in the trade (Notification 2016/021).

Analysis: The African Grey Parrot has a very extensive range. Total population is unknown, but it is clearly not small, and may be very large (several million). There is evidence of severe widespread declines in two range States in West Africa and declines have been observed elsewhere, particularly in areas where the species is known to be collected. Population trends are unknown in a very large proportion of its range although declines have been inferred from loss of habitat and over-collection. Given the relatively low rate of forest conversion in major parts of the range (notably Central Africa), and the ability of the species to survive in some modified habitats, habitat loss alone is highly unlikely to have led to a decline in line with the guidelines in *Res. Conf. 9.24 (Rev. CoP16)*, in this case a reduction of 50% in 45 years (three generations). Given the relatively low productivity of the species, and taking into account estimates of post-capture mortality, it is likely that much collection has led to population declines in areas where it takes place or has taken place. Overall, however, there is insufficient information to determine whether these declines have been widespread and severe enough for the entire population to have undergone a marked decline in the sense of the Resolution (the Red List Assessment note that the rate of decline is uncertain and may be between 30% and 49%). It is unclear, therefore whether the species meets the biological criteria for inclusion in Appendix I or not. Much reported trade is now in captive-bred birds originating outside of range States.

References:

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Transfer of the Southern Boobook (Norfolk Island) *Ninox novaeseelandiae undulata* from Appendix I to Appendix II

Proponent: Australia

Summary: The Norfolk Island Southern Boobook *Ninox novaeseelandiae undulata*, is a subspecies of owl that was once found on Norfolk Island and probably on the adjacent Philip Island, external territories of Australia. Extensive conversion of the native forest for agriculture has made the habitat unsuitable for the owl, leading to precipitous population decline. The last known genetically pure female of the subspecies was recorded in 1996. Surveys in 2005 found no birds of the subspecies on either Norfolk Island or Philip Island[1].

The parent species, *Ninox novaeseelandiae* (as recognised under CITES taxonomy) occurs in Australia, Indonesia, New Zealand, Papua New Guinea and Timor-Leste. Individuals of *Ninox novaeseelandiae novaeseelandiae* were introduced to Norfolk Island in 1987. Cross-breeding with the remaining female *N. n. undulata* resulted in a small hybrid population which is managed and subject to intensive monitoring.

Ninox novaeseelandiae undulata was included in Appendix I in 1977. In 1979 all owls (order Strigiformes) other than those included in Appendix I were included in Appendix II. No trade in the subspecies has been recorded in the CITES Trade Database. Extremely limited trade in *Ninox novaeseelandiae* has been recorded since the listing of Strigiformes; since 2002, Australia has exported 18 scientific specimens according to the CITES Trade Database.

Ninox novaeseelandiae undulata is listed as endangered in the Environment Protection and Biodiversity Conservation Act (1999), which regulates trade in CITES-listed and Australian native wildlife and their products. Export of a live Australian native mammal, bird, reptile or amphibian is strictly prohibited for commercial purposes but they may be exported for specific non-commercial purposes (e.g. for research, education, exhibition or a pet bird).

Analysis: The genetically pure subspecies *Ninox novaeseelandiae undulata* is evidently extinct. It was never recorded in trade; trade in its parent species *N. novaeseelandiae* has been extremely limited, with no commercial trade reported from a range State. The remaining hybrid population (*N. n. novaeseelandiae* x *N. n. undulata*) is managed and intensively monitored. In the unlikely event of *N. n. undulata* being rediscovered, Australian national legislation would prohibit its export for commercial purposes. All trade in *N. novaeseelandiae* (and its hybrids) will come under the provisions of CITES Appendix II and Australian national legislation, which prohibits export of live specimens for commercial purposes. It would appear that the precautionary measures in Annex 4 of *Res. Conf. 9.24 (Rev. CoP 16*) have been met.

References:
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Transfer of the American Crocodile *Crocodylus acutus* population in the Bahia Cispata, Tinajones, La Balsa and Sectores Alendanos in the District of Cordoba, Colombia from Appendix I to Appendix II for the purposes of ranching

Proponent: Colombia

Summary: The American Crocodile *Crocodylus acutus* is a widely distributed species, occurring in 17 range States in southern North America, Central America, the Caribbean and northern South America. The population of Cuba is in Appendix II; all other populations are in Appendix I. The species is classified as Vulnerable in the IUCN Red List (2012).

In Colombia, the species is found in a number of mangrove swamps and river deltas, including Cispata Bay, Tinajones, La Balsa and surrounding area of the department on Department of Cordoba. The mangroves extend over a total area of almost 115km^2 [1]. In 2006, these and the surrounding area were declared an Integrated Management District (DMI-BC): hereafter referred to as Cispata Bay or DMI-BC. A proposal for the same area was submitted to CoP16.

Since 2003 the species has been the subject of an active management programme in Cispata Bay, involving the construction of artificial nesting areas and head-starting based on release of juveniles hatched from eggs taken from the wild. Around 2500 juveniles were released in total between 2004 and 2014. Fertile eggs collected from nests and eggs artificially incubated to the point of hatching have also been "released"; with a hatching rate of almost 70% from those artificially incubated.

The current population is believed to be between 800 and 2356 individuals based on surveys covering 14km^2, or 80% of its habitat, the remainder being inaccessible. The population structure is considered to reflect a recovering or stable population. It is possible that the population has reached carrying capacity, and individuals are thought likely to be migrating out of the area.

Management measures in line with *Res. Conf. 11.16 (Rev. CoP15)* on Ranching are detailed in the proposal and a management plan is said to be under development (but not supplied). Relevant parts of this Resolution and responses to them in the supporting statement are:

b) **i) The programme must be primarily beneficial to the conservation of the local population**
Benefits to the local community will provide incentives to continue protecting this population. The area can act as a replicable model.

ii) all products (including live specimens) must be adequately identified
Details on marking of eggs through to hatchlings and skins are provided. Skin marking will follow the CITES universal marking system and will include a label "ACUTUS CISPATA COLOMBIA", which will distinguish skins from those originating in the 7 registered captive breeding facilities for the species in Colombia.

iii) the programme must have in place appropriate inventories, harvest controls and population monitoring mechanisms
The Cispata Bay population, including nests, has been monitored using standardized methods since 2003, led by the local competent environmental authority (CVS).No detail is provided on harvest controls except that only authorized ASOCAIMAN members will be permitted to harvest eggs. A management plan under development will include guidelines for egg collection from a sampled area (this management plan is not currently available). Trade quotas will be established and adaptively managed based on monitoring, as will the populations of individuals to be reintroduced to the wild.

iv) adequate numbers of animals must be returned to the wild
Progeny from 10% of all eggs collected will be returned to the wild after reaching approximately 100cm size in controlled conditions. This figure will be revised based on population monitoring.

c) **i) details of the marking system must be submitted**
Details of the marking system have been provided.

ii) a list of products must be provided
It appears that skins are the only products that will exported.

iii) a description of marking methods to be provided
Details are provided.

iv) an inventory of current stocks
Current stocks are 857 juveniles and sub adults. The intention is for these to be exported with an experimental quota of 200 skins per year until the stocks are exhausted.

d) **i) off-take should not be detrimental**
Details of the proposed offtake have not been provided however, the local community has been collecting eggs (with replacement) for over 10 years and the population appears to be stable or increasing. Management will be adapted on the basis of monitoring.

ii) the likely of biological and economic success of the operation
The DMI-BC has demonstrated the potential biological success over the past 10 years with the recovery and stabilisation of the population. Sustainable use will allow costs to be internalised for long tem economic success. Income will also be generated through ecotourism, research and education.

iii) there should be an assurance of no cruelty
The proponent states that humane methods will be used that guarantee no cruelty and comply with national legislation.

iv) there should be documentary evidence that the programme is beneficial to the wild population
The community conservation project that has been running for over 10 years has reintroduced animals and also built artificial nests, many of which are being used by female crocodiles on the a regular basis. The community group ASCOCAIMAN is predominantly composed of ex-hunters, who are now conserving instead of hunting the crocodiles. It is believed the area may serve as a replicable model.

v) there should be assurance that the above conditions will continue to be met
The programme has been running successfully over the past 10 years and a management plan is in development.

Overall, the IUCN/SSC Crocodile Specialist Group considers the proposed management measures to be sound[2].

Regarding implementation of *Res .Conf. 11.16 (Rev CoP15)*: recent problems with the management of *Caiman crocodilus fuscus* farming in Colombia have been identified, in particular the export of ranched and wild-harvested specimens declared illegally as captive-bred. At the 66th Meeting of the CITES Standing Committee Colombia and the EU made a joint declaration in which Colombia undertook to take action to ensure the legal origin of the traded specimens, to be implemented by 31 May 2016[3]. It is not clear to what extent these actions have been implemented.

Analysis: The *Crocodylus acutus* population of Cispata Bay, Colombia, remains small (<2500 individuals), with a restricted range; however the population appears to be increasing or stable, and possibly at carrying capacity and does not appear to be threatened at present. Most management conditions set out in *Res. Conf. 11.16 (Rev. CoP15*) appear to be in place, although some details on key elements such as harvesting controls and offtake are not available. It is possible that these will be included in the management plan under development.

References:
Information not referenced in the Summary section is from the Supporting Statement.

---

[1] CoP16 Prop. 23. (2013) Transfer of the population of the Bay of Cispata, municipality of San Antero, Department of Córdoba, Republic of Colombia, from Appendix I to Appendix II. https://cites.org/sites/default/files/eng/cop/16/prop/E-CoP16-Prop-23.pdf. Viewed on 24th June 2016.
[2] IUCN/SSC Crocodile Specialist Group (2016) *In litt.* to IUCN/TRAFFIC Analysis Team, Cambridge, UK.
[3] SC66 (2016) Summary record. https://cites.org/sites/default/files/eng/com/sc/66/ExSum/E-SC66-SR.pdf. Viewed on 24th June 2016.

Delete the zero quota for wild specimens traded for commercial purposes from the Appendix II listing of the population of Mexico of Morelet's Crocodile *Crocodylus moreletii*

Proponent: Mexico

Summary: Morelet's Crocodile *Crocodylus moreletii* is a small to medium-sized crocodilian that occurs in freshwater lagoons, swamps, streams and backwaters in forested areas or those with dense waterside vegetation in Belize, Guatemala and Mexico. It was included in Appendix I in 1975. In 2010 the populations of Mexico and Belize were transferred to Appendix II with a zero quota for wild specimens for commercial purposes. The population of Guatemala remains in Appendix I.

Surveys indicate that Mexico's population has continued to increase since its transfer to Appendix II, from an estimated 54,000 (of which almost 13,500 were adults) in 2010 to ca. 100,000 (of which some 19,000 were adults) in 2015. The population structure is pyramidal, generally accepted as reflecting a healthy population. The species is present throughout its natural range in Mexico of around 400,000km^2 of which 25,277km^2 is considered optimal habitat.

Management is based around Wildlife Management Units known as UMAs which must have seven basic elements: registration with the CITES Management Authority; an approved management plan; management and conservation of habitat; monitoring of the harvested wild populations; submission of regular reports and inventories; technical supervision visits; and marking or labelling (in line with the universal tagging system in *Res. Conf. 11.12 (Rev. CoP15)*).

The Supporting Statement indicates that Mexico's population of *C. moreletii* may have the potential to be harvested in all size classes. However, the current intention is that wild harvest will be restricted to eggs with the resulting hatchlings raised in UMAs. No quotas or harvest limits are mentioned in the proposal, although the Supporting Statement notes that many programmes in use around the world for different species of crocodilians suggest that 50-80% annual removal of eggs can be sustainable[1].

The Mexican Government is developing a pilot project on sustainability, production systems and traceability for *C. moreletii* skins in collaboration with RESP (Responsible Ecosystem Sourcing Platform). The programme aims to involve local communities in the conservation of the species and its habitat through ranching. The CITES Scientific Authority of Mexico (CONABIO) is funding the development of a ranching protocol in conjunction with experts to support the implementation of the pilot project. This protocol will include aspects of population monitoring and nests; monitoring and habitat management; estimation of sustainable harvest rates for ranching; management of nests, extraction and transfer of eggs; incubation (including details on the infrastructure, equipment and materials); and care of offspring from birth to sale. This will be available as an Information Document at CoP17.

Because the population of *C. moreletii* was not transferred to Appendix II for ranching in accordance with *Res. Conf. 11.16 (Rev. CoP15)* Mexico intends to export specimens with the source code "w" (wild) although initially eggs will be harvested from the wild and raised – that is effectively ranched.

Since 2011 Mexico has implemented a monitoring programme for the species over its entire range, which includes the monitoring of 73 permanent sites in Mexico. Monitoring within Mexico is considered to be sophisticated and sufficient to detect any adverse impacts of harvesting on the population[1].

Analysis: There are no explicit guidelines in *Res. Conf. 9.24 (Rev. CoP16*) for assessing removal of a zero quota for wild specimens from an Appendix-II listed species. However, such removal may be seen as analogous to a transfer from Appendix I to Appendix II. Mexico's population of *Crocodylus moreletii* evidently does not meet the criteria for inclusion in Appendix I: it is not small nor does it have a restricted range, and it has been increasing for at least the past 10 years.

The precautionary measures set out in Annex 4 of *Res. Conf. 9.24 (Rev. CoP16)* can be met in various ways, including the Parties being satisfied with the range State's implementation of the Convention, particularly Article IV, and with its enforcement controls and compliance with the Convention, or if an integral part of the amendment proposal is a special measure approved by the CoP, based on management measures described in the Supporting Statement, provided that effective enforcement controls are in place.

In this case, the intent only to harvest eggs from the wild population in the coming years could be taken as such a special measure. Management measures and enforcement controls described in the Supporting Statement appear to be sufficient to ensure that such harvest will not have an adverse impact on the population. Further details should be provided in the ranching protocol under development.

References:

Information not referenced in the Summary section is from the Supporting Statement.

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[1] IUCN SSC Crocodile Specialist Group (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

Maintenance of the Malagasy Population of Nile Crocodile *Crocodylus niloticus* in Appendix II, pursuant to *Res. Conf. 9.24 (Rev. CoP16)* Annex 2(a), paragraph B) rather than to *Res. Conf. 11.16 (Rev. CoP15),* subject to the following annotations:

1. No skins or products within the artisanal industry from wild *C. niloticus* less than 1m or greater than 2.5m total length will be permitted for national or international trade
2. An initial wild harvest ceiling of 3000 animals per year for the artisanal industry will be imposed for the first three years of operation (2017-2019)
3. No export of raw or processed skins harvested from the wild will be permitted for the first three years
4. Farm production shall be restricted to ranching and/or captive breeding, with national skin production quotas
5. Management, wild harvest ceiling and national skin production quotas will be audited and reviewed annually by international experts for the first three years to ensure sustainability

Proponent: Madagascar

Summary: The Nile Crocodile *Crocodylus niloticus* is a large crocodile with a wide distribution across sub-Saharan Africa and in Madagascar. The populations of 13 range States, including Madagascar, are included in Appendix II, some under particular restrictions; all other populations are in Appendix I. This proposal only affects the Madagascar population.

The Malagasy population of *C. niloticus* was originally transferred in 1985 from Appendix I to Appendix II under *Res. Conf. 5.21* on special criteria for the transfer of taxa from Appendix I to Appendix II (no longer in effect), and subject to an annual export quota to allow limited trade in wild skins from nuisance animals. At the Tenth Conference of the Parties (CoP10) in 1997 a proposal was accepted to include the species in Appendix II pursuant to *Res. Conf. 3.15* [now *Res. Conf. 11.16 (Rev. CoP15)*] on ranching rather than *Res. Conf. 5.21.* Between 1985 and 1997 the CoP agreed on varying export quotas for wild specimens, ranched species or wild nuisance specimens. From then on the CoP did not set quotas; instead, Madagascar authorized exports in accordance with its ranching programme[1].

Concerns regarding Madagascar's compliance with *Res. Conf. 11.16* were raised at the Animals Committee (AC) in 2006 and transmitted to the Standing Committee (SC). The Secretariat visited Madagascar in 2006 and confirmed that Madagascar was not fully complying with the Resolution and that controls of farming operations had been insufficient to prevent abuses. The SC developed a series of recommendations for Madagascar (Annex 1 to SC55 Doc. 13[2]).

At SC60 the SC recommended, in view of persistent concerns about the management of ranching operations in the country, that trade in this species with Madagascar be suspended; this recommendation came into effect in June 2010. At SC65 (July 2014) the SC decided, in view of the progress that Madagascar had made, that the suspension could be withdrawn, conditional on evidence that specific Decrees and Ministerial Orders, complying with the SC's recommendations, had entered into force. Parties were notified that the recommendation to suspend trade was withdrawn in December 2014[3].

At the same time the SC noted that Madagascar had agreed to the following actions[4]: submit a zero quota to the Secretariat for wild skins as it did not intend to export any wild skins in 2014 or in the future; adopt a zero quota for ranched specimens for 2014 and 2015; carry out an inventory of live captive-bred animals at the ranches and set annual export quotas for captive-bred skins and products, based on the results of the inventory and production potential on the farms; evaluate the skin stockpile at the ranches; and determine in consultation with the Secretariat how to dispose of the stockpiles on the international market in 2014 (and 2015 if necessary), taking into consideration whether they were legally acquired or not.

Crocodylus niloticus is widely distributed in Madagascar in a variety of freshwater habitats below 1500m. It is most abundant in northwest and western area and in the northeast. The wild population has been estimated at 30,000 to 40,000 non-hatchlings although the basis of this is questionable. In 2015, surveys indicated a high proportion of juvenile and sub-adult individuals in the population (36% of sightings); and based on various indices the adult population was estimated at 1500 to 2000 individuals. However, many wetlands remain unsurveyed, and extrapolating from available relative density estimates in these to an estimate of the total or absolute population size in Madagascar is problematic[5]. Survey results conducted since 2000 suggest that in many areas surveyed the population is increasing or stable. Hunters and local communities are also said to consider that crocodile abundance has increased over the last few years, attributed to a

reduction in wild harvest since 2010. The species is affected by habitat degradation, including loss or alteration of nesting areas. Expansion of the human population has resulted in the local extinction of *C. niloticus* due mainly to habitat loss and public safety concerns.

Crocodylus niloticus was categorised by IUCN as being globally of Lower Risk/least concern in 1996 (needs updating).

Past annual export quotas for wild skins have been 100 to 200 for 1992 to 1997, 500 to 750 wild skins for 1998 to 2007 and 200 wild skins for 2008 to 2011. In addition to the export quotas, a much larger (annual average of ca. 5000 for 1987 to 2009 and ca. 2500 for 2010 to 2015) wild harvest of skins has been permitted for the artisanal industry for the production of finished leather goods and taxidermy specimens (75% of which are said to be sold and used domestically). Export during 2010 and 2015 may have accounted for ca. 625 wild crocodiles per year. Artisanal products purchased by tourists and taken with them as personal effects (pursuant to *Res. Conf 13.7 (Rev. CoP16)* have continued to be exported from the country. Specimens are considered exempt and they are not accounted for in Madagascar's trade data.

Madagascar wishes to maintain its population of *C. niloticus* in Appendix II under *Res. Conf. 9.24 (Rev. CoP16)* Annex 2(a), paragraph B) rather than to *Res. Conf. 11.16 (Rev. CoP15).*

The Supporting Statement notes that the proposal reflects the rebuilding of a revised management paradigm for Madagascar's *C. niloticus*, established in 2014 and aimed primarily at sustaining and rebuilding the wild population, and consolidating and better regulating the wild harvest associated with the artisanal industry. The new approach entails moving towards export of products of wild harvested crocodiles that have been processed through the artisan industry. Details of the proposed system are supplied in the Supporting Statement. Some of these are included in the proposed annotation. These are as follows:

1) Harvest size limits (skins must be over1 m and less than 2.5m in total length)
All wild skins must pass through registered artisanal tanneries, of which there are currently 14. Each tannery is obliged to register information on each skin which is tagged on arrival; this information is submitted to the relevant Government department on a quarterly basis. Random inspections are carried out on tanneries by Government officials. Morphometric relationships predicting the size of crocodiles from which finished products were derived have been developed and are used to verify compliance with skin size limits. The Supporting Statement notes these limits may need to be adjusted (up or down) over time, as more information becomes available on the population size, structure and trends[5].

2) Wild harvest limited to 3000 animals per year for the first three years (2017-2019)
A wild harvest of ca. 5000 per year occurred between 1987 and 2009 and ca .2500 between 2010 and 2015. The harvest ceiling will be reviewed in 2019. Egg collection will continue to be the main source of stock for ranches in tandem to the wild harvest. Quotas are not specified. Improvements in the egg collection system (in place since 1980s) have been proposed and a "desire to implement these has been expressed". The proposed egg harvests are considered unlikely to impact detrimentally on the wild population.

3) No export of raw or processed skins harvested from the wild will be permitted for the first three years
As noted above, wild skins are tagged on arrival at artisanal tanneries, so that these are distinguishable from those produced through ranching and captive-breeding, which are tagged in accordance with *Res. Conf. 11.12 (Rev. CoP15)*. Products must also be tagged with a label provided by the Government department. The current paper label has been problematic and new options (e.g. plastic tags, embossing, etc.) are being examined. Very small products (e.g. teeth) have been exempted from the requirements of labelling for the time being. However, small products must still comply with skin/crocodile size limits.

4) Farm production shall be restricted to ranching and/or captive-breeding, with national skin production quotas
Skins produced through ranching and captive-breeding, and being exported, are tagged in accordance with *Res. Conf. 11.12 (Rev. CoP15).* The Supporting Statement indicates that skins that enter the domestic market will also be tagged, and monitored through the registers maintained by tanneries and manufacturers.

5) Management including harvest quotas and skin production quotas will be audited and reviewed annually
Further provisions not specified in the annotation include:

- A hunting season for wild crocodiles, currently specified as between January and September, but in reality due to the wet season they state that the effective hunting season is April to November.

- Permits to take problem animals must be approved by the relevant Provincial Forestry authority, and the skin must be delivered to the local forestry or local government authority if retrieved. Skins of problem crocodiles larger than 2.5m total length are currently not allowed to enter the domestic or international markets, and remain the property of the Government. However, options are being examined for the legal disposal of such skins.
- Population monitoring. Standard surveys will be undertaken inside and outside harvesting areas.
- Compliance with Article IV, particularly the non-detriment provisions, will be assessed annually based on indices from population surveys and the industry. Management and levels of harvest will be assessed annually, in collaboration with international experts in at least the first three years of the program (2017-2019), and harvest levels may be adjusted up or down after three years, on the basis of these independent assessments.

Analysis: The Malagasy population of the *Crocodylus niloticus* was originally in Appendix I and is now in Appendix II under the conditions of *Res. Conf. 11.16 (Rev. CoP16)*. The current proposal entails a detailed, substantive annotation that could be counted as a special measure under Annex 4 of *Res. Conf. 9.24 (Rev. CoP16)* (sub-para. A 2 a) iii)) to be approved by the CoP based on management measures described in the Supporting Statement, provided that effective enforcement controls are in place.

If successfully implemented, it appears that management measures specified in the annotation and in the Supporting Statement would ensure compliance with the Convention, particularly Article IV. As noted above, there have been problems in compliance with the Convention in Madagascar with respect to export of *C. niloticus*, resulting in a recommendation from the Standing Committee to suspend trade with Madagascar in the species in 2010. In 2014 the Standing Committee (SC) agreed that these problems had largely been resolved, but that some final steps were needed. These have taken place, and the suspension was withdrawn in December 2014.

Madagascar indicated to the SC in 2014 that it did not intend to export wild skins at any time in the future. The current annotation indicates that this restriction would only apply for three years from the date of its adoption.

The proposed annotation contains substantive management measures and is not in conformity with recommendations on the use of annotations in Appendices I and II in *Res. Conf. 11.21 (Rev. CoP16)*, which states that substantive annotations should be confined to designation of types of specimens or export quotas, or inclusion or exclusion of geographically separate populations. Any change to the substantive provisions in it would need an amendment proposal to be approved by the CoP.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] CITES (2014) Ranching Operations in Madagascar – Report of the Secretariat SC65 Doc. 25.2 https://cites.org/sites/default/files/eng/com/sc/65/E-SC65-25-02.pdf Viewed on 5th July 2016.
[2] CITES (2007) Species Trade and Conservation Issues. Ranching Operations. https://cites.org/sites/default/files/eng/com/sc/55/E55-13.pdf Viewed on 5th July 2016.
[3] CITES (2014) Notification No. 2014/064.
[4] CITES (2014) Compliance and Enforcement. Report of the Working Group on Ranching Operations in Madagascar. SC65 Com. 1. https://cites.org/sites/default/files/eng/com/sc/65/com/E-SC65-Com-01.pdf Viewed on 5th July 2016.
[5] Webb, G. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

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Transfer of the Saltwater crocodile *Crocodylus porosus* in Malaysia from Appendix I to Appendix II, with wild harvest restricted to the State of Sarawak and a zero quota for wild specimens for the other States of Malaysia (Sabah and Peninsular Malaysia), with no change in the zero quota unless approved by the Parties

Proponent: Malaysia

Summary: The Saltwater or Estuarine Crocodile *Crocodylus porosus* currently occurs in Australia, Bangladesh, Brunei, India, Indonesia, Malaysia, Myanmar, Palau, Papua New Guinea, Philippines, Singapore, Sri Lanka, Solomon Islands, Thailand (where is it virtually extinct) and Vanuatu. It is widely distributed in Malaysia's three states of Peninsular Malaysia, Sabah and Sarawak. The species is currently included in Appendix I, except for the populations of Australia, Indonesia and Papua New Guinea, which are included in Appendix II.

By the late 1980s Malaysia's population of *C. porosus* was seriously depleted owing to overexploitation, primarily for hides and meat[1]. Initiation of conservation programmes at that time has resulted in significant increase in the populations of Sarawak and Sabah. This proposal would entail harvest for export only of the population of Sarawak. Little is known about the size of the population in Peninsular Malaysia but it is thought to be small. In Sabah surveys in 2002 indicated that in some areas *C. porosus* numbers had increased by about 10 fold since the 1980s, with nearly four crocodiles per km of river bank in some rivers.

Sarawak covers an area of 12 million hectares with 22 major river basins; *C. porosus* is reported to occur in all these. Two separate recent surveys which covered just over 2000km in essentially the same wetlands came up with very similar estimates of ca. 12,000 individuals and 13,507 non-hatchlings. These estimates are considered conservative as heavily vegetated swamps were not surveyed. The population structure in Sarawak appears to be consistent with a population that has recovered and is both viable and healthy. Suitable habitat reportedly remains abundant. There has been increased incidence of human-crocodile conflict, including fatal and non-fatal attacks on humans.

The Supporting Statement states that the maximum sustainable yield for wild *C. porosus* populations is not known precisely, but notes that a 5% annual harvest rate for alligators did not interfere with continued population growth. It considers a 5% annual harvest rate for the non-hatchling population in the surveyable rivers of Sarawak to have a high probability of being sustainable.

Harvesting is proposed of 500 non-hatchlings and 2500 eggs or their equivalent based on average survival rates i.e. 750 hatchlings or 375 yearlings. The figure of 500 is derived from 5% of the higher of the two population estimates above reduced to be more precautionary (5% of 13,507 is 675). If all the additional harvest is of yearlings, the total non-hatchling offtake could be 875, which represents around 6.5% of the higher population estimate.

Harvest of 2500 eggs is equivalent to around 50 nests per year. Based on offtake of the species in Australia this harvest is thought unlikely to have any impact on the population because density-dependent factors will increase the survival rate of hatchlings in non-harvested nests[2].

A Master Plan for Wildlife in Sarawak has been put in place, providing recommendations and guidelines for wildlife and its habitats. A crocodile Management Plan has been drawn up to address the use of crocodiles in Sarawak. Funding has been provided for necessary monitoring. Based on population monitoring and assessment of the impact of harvesting on the non-hatchling wild population, the offtake will be adaptively managed, with harvesting in successive years reduced proportionately if the wild population is seen to be declining.

Movement within the state of Malaysia may require export and/or import license or permit to be issued by the Controller of Wild Life. Malaysia currently has seven registered captive-breeding facilities for *C. porosus*, two of which are in Sarawak, which primarily produce skins for export.

The species is in trade from other range States where populations are already in Appendix II (Australia, Indonesia and Papua New Guinea) as well as from captive-breeding facilities. Trade from different states in Malaysia would not be differentiated in the CITES Trade Database.

Analysis: Malaysia's population of the Saltwater or Estuarine Crocodile *C. porosus* is neither small nor does it have a restricted range. Conservation action over the past 30 years has resulted in a marked population increase in Sarawak and Sabah, two of the three Malaysian States. Sarawak's population is currently estimated at over 10,000 individuals. The population would appear to no longer meet the biological criteria for inclusion in Appendix I set out in Annex 1 of *Res. Conf. 9.24 (Rev. CoP16).*

For a transfer from Appendix I to II the precautionary measures in Annex 4 of *Res. Conf. 9.24 (Rev. CoP16)* apply. These can be met in various ways, including the Parties being satisfied with the range State's implementation of the Convention, particularly Article IV, and with its enforcement controls and compliance with the Convention, or if an integral part of the amendment proposal is a special measure approved by the CoP, based on management measures described in the Supporting Statement, provided that effective enforcement controls are in place.

In the case of Sarawak the intent is to harvest a limited number of non-hatchlings and eggs, or the equivalent of those eggs in hatchlings or non-hatchlings, with initial harvest level set on the basis of current population estimates and future harvest adjusted adaptively based on results from annual population monitoring. This could be interpreted as a special measure under the terms of Annex 4 of *Res. Conf. 9.24 (Rev. CoP16)*. Relatively few details are provided on management measures to control harvest and trade. If all initial harvest is in non-hatchlings, the proposed offtake may exceed the reference level of an annual sustainable harvest suggested in the Supporting Statement (ca. 6.5% vs 5% of the population). No mention is made of intention to comply with the universal tagging system for the identification of skins in *Res. Conf. 11.12 (Rev. CoP15)*. No details are given of how specimens would be differentiated from those from the captive-breeding facilities, particularly as the marking provisions in *Res. Conf. 10.16* (*Specimens of animal species bred in captivity*) would no longer be applicable if the population were transferred to Appendix II. The proposal includes zero quotas for [wild] specimens from Peninsular Malaysia and Sabah. It is not clear whether measures detailed would be adequate to ensure that specimens from Peninsular Malaysia and Sabah do not enter the trade chain through Sarawak. The Crocodile Management Plan that has been drawn up may provide further information to verify whether precautionary safeguards are met.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] Gani, M.I.Z.A. (2014) *Population density, human-crocodile conflict and genetic variation among saltwater crocodile, Crocodylus porosus in Sarawak*. Master's thesis, University Malaysia Sarawak, (UNIMAS). http://ir.unimas.my/9017/.
[2] IUCN SSC Crocodile Specialist Group (2016) *In litt*. to IUCN/TRAFFIC Analyses Team, Cambridge, UK.

A) Inclusion of the following species of the Genus *Abronia* into Appendix I: *Abronia anzuetoi, A. campbelli, A. fimbriata, A. frosti* and *A. meledona*

B) Inclusion of the following species of the Genus *Abronia* into Appendix II: *Abronia aurita, A. gaiophantasma, A. montecristoi, A. salvadorensis* and *A. vasconcelosii*

An annotation is also proposed:
a) for zero quota for wild specimens, and
b) zero quota for captive bred specimens from non-range Sates. This annotation would allow for captive-bred exports from range States

Proponent: Guatemala

Summary: The genus *Abronia*, known as alligator lizards or abronias, are medium-sized insectivorous arboreal lizards from Mexico (MX) and northern Central America (El Salvador (SV), Guatemala (GT), and Honduras (HN). They mainly inhabit montane cloud forests where they are associated with epiphytes in the canopy of tall mature oak or pine trees. They give birth to between one and twelve live young once a year.

This proposal considers ten species that are found in Guatemala, El Salvador and Honduras. A second proposal at CoP17 submitted by Mexico and the European Union proposes the inclusion of all species of *Abronia* in Appendix II. See analysis of CoP17 Proposal 26 for a discussion of the genus as a whole.

Inclusion in Appendix I

Abronia anzuetoi (GT): Only known from one patch of forest with an area of 24km^2 [1]. No information on population size or trends. No major threats known. Reported in pet trade in China and Switzerland (see CoP17 Proposal 26). Classified in the IUCN Red List as Vulnerable (2014).

Abronia campbelli (GT): Only known from one patch of forest with an area of 18km^2 and an estimated population of 500 individuals[2]. There is ongoing habitat loss and degradation from cattle ranching. In 2010, 47 individuals were confiscated from an illegal pet market in Mexico[3]. Known illegal trader has asked locals about this species within its native range[4]. Classified in the IUCN Red List as Critically Endangered (2013).

Abronia fimbriata (GT)***:*** Known from four locations with an extent of occurrence of around 1500km^2. No data on population status or trends. There is continuing decline in the extent and quality of its habitat due to conversion to agriculture, and collection of ornamental plants[5]. There is evidence online of international trade for the pet market. Classified in the IUCN Red List as Endangered (2014).

Abronia frosti (GT): Only known from a few specimens at one location which is a patch of forest 0.7km^2 in area. No information on population size or trends. It is reported to be subject to continuing decline in the extent and quality of its habitat due to logging for firewood. A trader has asked locals about this species within its native range[4]. Classified in the IUCN Red List as Critically Endangered (2013).

Abronia meledona (GT): Limited to one area with an extent of occurrence of less than 900km^2. No information on population size or trends. It is subject to continuing decline in the extent and quality of its habitat due to agricultural activities. A known illegal trader has asked locals about this species within its native range[4]. Classified in the IUCN Red List as Endangered (2013).

Inclusion in Appendix II with a zero quota for wild specimens and a zero quota for captive-bred exports from non-range States

Abronia aurita (GT): Only known from one locality with an extent of occurrence of approximately 400km^2. No information on population status or trends. The forest where it is found is reportedly heavily fragmented and degraded. Three specimens of this species were seized in 2009 hidden in video cassette in UK, on route from Guatemala to Czech Republic[6]. Classified in the IUCN Red List as Endangered (2013).

Abronia gaiophantasma (GT): Known from fewer than five locations with an extent of occurrence of approximately 750km^2. Described as uncommon, population trend unknown. Reported to be affected by

habitat loss. Evidence online of international trade of this species. Classified in the IUCN Red List as Endangered (2014).

Abronia montecristoi (SV, HN, GT): Known from two locations of intact forest, with an extent of occurrence of approximately 800km^2. Not recorded recently, despite the area where it occurs being well surveyed for reptiles[7]. There is ongoing destruction of old growth forest where it is found in Honduras; habitat in El Salvador is reportedly better preserved[8]. Classified in the IUCN Red List as Endangered (2013).

Abronia salvadorensis (HN**):** Only known from a few specimens. Recorded from two locations with an extent of occurrence of up to 200km^2. Likely to be affected by habitat loss and degradation. Classified in the IUCN Red List as Endangered (2013).

Abronia vasconcelosii (GT): Known from 10 localities with an extent of occurrence of about 2500km^2. Previously described as common, the population is thought to be in decline as much of the land has been converted to agriculture since the 1990s. Reported trade of this species in Czech Republic and United Kingdom and advertised for sale online. Classified in the IUCN Red List as Vulnerable (2013).

Abronia species are in trade for the exotic pet market. This trade is reviewed in the analysis of Proposal 26. The great majority of recorded trade is in the Mexican *A. graminea* and most of the remainder in unspecified *Abronia* spp[9]. Trade data records a small number of specimens imported with origin Guatemala, all for scientific purposes[9].

There is no authorised collection for trade or commercial export of *Abronia* species native to El Salvador, Honduras and Guatemala. However there is reported commercial trade or evidence online of the sale of *A. anzuetoi, A. campbelli, A. fimbriata, A. aurita, A. gaiophantasma* and *A. vasconcelosii*.

The range of all of these species overlaps with protected areas, although often only partially. There are ongoing monitoring programmes in Guatemala, plus local education and awareness programmes. A captive breeding program has begun for *A. campbelli, A. frosti* and *A. meledona* with some successful re-releases. In Mexico there is at present captive-breeding in government Wildlife Management Units (UMAs) of *A. campbelli* as well as a number of Mexican species; a private initiative in Mexico is also captive-breeding *A. vasconcelosii*.

Analysis:

Inclusion of *Abronia anzuetoi, A. campbelli, A. fimbriata, A. frosti* and *A. meledona* in Appendix I

Available information indicates that *Abronia anzuetoi, A. campbelli,* and *A. frosti* all have small or very small ranges in which there is said to be ongoing habitat degradation. These appear to meet the biological criteria for inclusion in Appendix I in *Res. Conf. 9.24 (Rev. CoP16)*.

Abronia fimbriata and *A. meledona* have more extensive distributions, although habitat in these is also believed to be declining in quality and extent. There is no information on population levels or trends, other than the inference that populations are likely to be declining. There is insufficient information to determine whether these species meet the biological criteria for inclusion in Appendix I.

There is international demand for *Abronia* species indicating that these species meet the trade criteria for inclusion in Appendix I.

Inclusion of *Abronia aurita, A. gaiophantasma, A. montecristoi, A. salvadorensis* and *A. vasconcelosii* in Appendix II

These species have known areas of occurrence ranging from 200km^2 to 2500km^2. There is no information on population levels or trends on any, other than an inference that population are likely to be declining owing to declines in quality and extent of habitat. There are indications of trade in three of them (*A. gaiophantasma, A. aurita* and *A. vasconcelosii).* However such trade (which is illegal in wild-caught specimens from range States) appears to be at a very low level and it seems unlikely that harvest for it will be reducing the species to a level at which they may qualify for inclusion in Appendix I in the near future, or at which its survival might be threatened by continued harvesting or other influences.

The proposal includes a zero quota for captive-bred specimens from non-range States. This is intended to reflect that no legal export for commercial purposes has been permitted for these species and therefore any founding stock of commercial captive breeding facilities is believed to have been imported illegally. There is no other example of such a restriction on trade in captive-bred specimens of Appendix-II listed species in the Appendices.

There is a great deal of variation within species, and it can be difficult to distinguish between some species. Given that at least three of the species proposed here for listing in Appendix I appear to meet the criteria, then the other species meet the criteria in Annex 2 b of *Res. Conf. 9.24 (Rev. CoP16)* (lookalike).

Reviewers of summary information only: D. Ariano-Sánchez, J. Campbell, W. Schmidt, J. Janssen and S. Chng.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] Ariano-Sánchez, D., Acevedo, M. & Johnson, J. (2014) *Abronia anzuetoi*. The IUCN Red List of Threatened Species 2014.
[2] Ariano-Sánchez, D., & Torres-Almazán, M. (2010) Rediscovery of *Abronia campbelli* (Sauria: Anguidae) from a Pine-Oak Forest in Southeastern Guatemala: Habitat Characterization, Natural History, and Conservation Status. *Herpetological Review*. 41: 290.
[3] Ariano-Sánchez, D., Johnson, J. & Acevedo, M. (2013) *Abronia campbelli*. The IUCN Red List of Threatened Species 2013.
[4] Ariano-Sánchez, D. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge.
[5] Acevedo, M., Ariano-Sánchez, D. & Johnson, J. (2014) *Abronia fimbriata*. The IUCN Red List of Threatened Species 2014.
[6] Daily Mail (2009) Real-life video nasty: Customs officials discover 3 rare lizards smuggled inside cassette box. http://www.dailymail.co.uk/news/article-1233257/Real-life-video-nasty-Customs-officials-discover-3-rare-lizards-smuggled-inside-cassette-box.html. Viewed on 29th June 2016.
[7] Campbell, J. A. & Frost, D.R. (1993) Anguid lizards of the genus *Abronia*: revisionary notes, descriptions of four new species, a phylogenetic analysis, and key. *Bulletin of the American Museum of Natural History.* 216.
[8] Townsend, J.H. & Köhler, G. (2013) *Abronia montecristoi.* The IUCN Red List of Threatened Species 2013.
[9] Analysis of US Fish & Wildlife Service Law Enforcement Management Information System (LEMIS) data, May 2016.

Inclusion of all species of Alligator Lizards in the genus *Abronia* in Appendix II

Proponents: Mexico and European Union

Summary: The genus *Abronia*, known as alligator lizards or abronias, are medium-sized insectivorous arboreal lizards from Mexico (MX) and northern Central America (El Salvador (SV), Guatemala (GT), and Honduras (HN)). Currently 29 species are recognised; most are endemic to Mexico. There may be as many as four as yet undescribed species[1]. They mainly inhabit montane cloud forests where they are associated with epiphytes in the canopy of tall mature oak or pine trees; four species (including an undescribed one) occur in lower altitude tropical forests. They give birth to between one and twelve live young once a year.

Ten of the species are the subject of a second proposal at CoP17 submitted by Guatemala (see Proposal 25).

Most species are only known from small areas, often occurring in single montane forests[1]. Seven Mexican species – *A. deppii*, *A. graminea*, *A. lythrochila* (also in Guatemala), *A. mixteca*, *A. oaxacae*, *A. smithi* and *A. taeniata* – have relatively extensive ranges of 400 to 3000km^2 although habitat is generally fragmented and actual area of occupancy of each species is likely to be considerably smaller than the overall range.

There is little population information for most of the species. *A. lythrochila* (GT, MX) and *A. oaxacae* (MX) have been described as common or moderately common within their ranges. Two studies of *A. graminea* (MX) (which has an extent of occurrence of about 3000km^2) at the same site in 2005 and 2015 produced estimates, based on capture of individuals, of approximately 30 and 45 per hectare respectively. The two were based on somewhat different methods and sampled relatively small areas but give an indication of minimum likely population densities in suitable habitat. In 2005 local people stated that the abundance of the species at the site was considerably lower than previously, although it is not clear how reliable this observation is. One study of *A. taeniata* (MX) found it to occur at much lower density than *A. graminea,* although it has also been described as common in suitable habitat.

Of the other species, *A. anzuetoi, A. campbelli* and *A. frosti* are known from small patches of forest (0.7 to 24km^2) in Guatemala. The population of *A. campbelli* was estimated at 500 adults in 2010, based on the number of mature trees within its range and an estimate of the average occupancy of each tree. *A. fimbriata* (GT), *A. gaiophantasma* (GT), *A. martindelcampoi* (MX), *A. meledona* (GT) and *A. vasconcelosii* (GT) are also reported to have limited distributions. The remainder are known from few specimens, sometimes only from single collections. These are: *A. aurita* (GT), *A. bogerti* (MX), *A. chiszari* (MX), *A. cuetzpali* (MX), *A. fuscolabialis* (MX), *A. leurolepis* (MX), *A. matudai* (GT, MX), *A. mitchelli* (MX), *A. montecristoi, A. ochoterenai* (GT, MX) (recently rediscovered[2]), *A. ornelasi* (MX), *A. ramirezi* (MX), *A. reidi* (MX) and *A. salvadorensis* (HN).

Abronia species are in trade for the exotic pet market. Trade has reportedly increased since the 1990s but recorded trade remains at a low level. Animals command high prices (several hundred USD or more per individual).

Available trade information comes from USA trade data[3] and Mexican export records. USA data report that the USA imported just over 230 *Abronia* in the period 2004 to 2013. Most of these (131) were *A. graminea* (MX), virtually all reported as captive-bred, the majority (110) from Mexico. Mexican records show legal export in the period 2005 to 2015 of just under 100 *A. graminea,* of which 55 were declared as captive-bred. The species is advertised for sale online in Europe and USA.

Very small numbers of *A. deppii*, *A. lythrochila, A. oaxacae* and *A. taeniata* were also recorded as imported by the USA, none from a range State. Remaining USA imports were not identified to species level; virtually all are of animals reported as captive-bred in non-range States. Mexican records show legal export in the period 2005 to 2015 of small quantities (fewer than ten each) of *A. taeniata, A. oaxacae* and *A. ornelasi,* all reported as of wild origin.

Internet searches and observations at trade fairs indicate that some 16 species in addition to *A. graminea* (MX) have been offered for sale, including some that do not have legal authorization for exploitation or export from their native country. Around 130 *Abronia* specimens were confiscated within Mexico in 2005 to 2015. There is evidence of demand online for *A. ochoterenai*, which has recently been re-discovered[4]. A known trader has been seen in the Guatemala asking locals about *A. campbelli*, *A. frosti* and *A. meledona*. This is

thought to be part of an established connection that is used for the illicit trade of reptiles out of Mesoamerica[5]. The greatest demand is reportedly from European countries and the USA.

Collection for the pet trade has been said to be a cause of concern for a number of species, including *A. campbelli*, *A. deppii*, *A. graminea*, *A. martindelcampoi*, *A. mixteca* and *A. taeniata*, but there is no information on the impact of collection for international trade on any of the species.

There is no authorised collection for trade or exportation of *Abronia* species native to El Salvador, Honduras and Guatemala. In Mexico, trade is regulated for most species, including *A. graminea*.

The range of several of the species overlaps with protected areas, although often only partially. There are ongoing monitoring programmes in Mexico and Guatemala, plus local education and awareness programmes. There are several captive breeding programmes both in Mexico and USA for *A. campbelli , A. graminea, A. taeniata* and *A. vasconcelosii* , and Guatemala has had some success with captive breeding and re-release of the native species *A. campbelli, A. frosti* and *A. meledona*. It appears that captive breeding is relatively straightforward for at least some of the species (*A. graminea* and *A. lythrochila*) and is done by private hobbyists in Europe and the USA.

Of the 29 species, 19 are classified as threatened on the IUCN Red List, two are classified as Least Concern (*A. lythrochila*) and *A. smithi*), and seven are classified as Data Deficient due to a lack of information on the population status and trends. *A frosti* is classified in the IUCN Red List as Critically Endangered (2013). *A. cuetzpali* was only described in 2016 and is yet to be assessed.

Analysis: There is little information on the wild population of most *Abronia* species although a number of them are believed to have very restricted ranges and probably small population sizes. *Abronia* species are sought after and may command high prices, although the specialist market for them – that is of collectors who seek out particular species – is almost certainly small or very small. The recorded legal trade in *Abronia* species is small. Most of it is in the Mexican species *Abronia graminea,* a large proportion of which are individuals reported as captive-bred. Available information indicates that this species is relatively widespread in the wild and can occur, at least locally, at moderately high population densities. It seems unlikely that the level of recorded trade in wild specimens is sufficient for this species to meet the criteria in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)*.

A number species other than *A. graminea* are reported in trade, including several for which no legal export from range States is permitted. Volumes in trade are unknown, but available information indicates they are likely to be small or very small. Some species (e.g. *A. campbelli*) may have such small wild populations that collection of a small number of individuals for export might be detrimental; specimens of this species have been confiscated in Mexico (a non-range State) and it is reported to be captive-bred there. It is possible, that some of the species meet the criteria for inclusion in Appendix I (see analysis for Proposal 25). Overall, however, there is insufficient information to determine whether any species of *Abronia* meets the criteria for inclusion in Appendix II in Annex 2 a of the Resolution.

There is a great deal of variation within species, and it can be difficult to distinguish between some species. If it is concluded that some of the species considered here meet the criteria for inclusion in the Appendices then the others would meet the criteria in Annex 2 b of *Res. Conf. 9.24 (Rev. CoP16)* (lookalike).

Reviewers of summary information only: D. Ariano-Sánchez, J. Campbell, W. Schmidt, J. Janssen and S. Chng.

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[3] Analysis of US Fish & Wildlife Service Law Enforcement Management Information System (LEMIS) data, May 2016.
[4] Janssen, J. (2016) *In litt*. to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
[5] Ariano-Sánchez, D. (2016) *In litt*. to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

Inclusion of all species of African Pygmy Chameleons in the genera *Rhampholeon* and *Rieppeleon* in Appendix II

Proposal 27 Proponents: Central African Republic, Chad, Gabon, Kenya, Nigeria and the United States of America

Proposal 28 Proponent: Kenya

Note: *Proposals 27 and 28 are identical in intent. One analysis is presented for the two.*

Summary: The proposals seek to include all species of *Rhampholeon* and *Rieppeleon* in Appendix II. *Rieppeleon* (three species) and *Rhampholeon* (22 species) are both genera of pygmy chameleons occurring in Africa. *Rieppeleon* spp. are distributed across Eastern Africa inhabiting lowland forests and non-forest habitats, including grassland, wet and dry savannah and woodland. *Rhampholeon* spp. occur in Western, Central and Eastern African and tend to be confined to montane forests. They appear to be at greater risk from habitat alteration and loss than *Rieppeleon* spp. as most species have more restricted ranges, more specialised habitat requirements and do not adapt as well to altered habitats.

Only two species of *Rieppeleon* and one of *Rhampholeon* have been reported in trade in any number:

Rieppeleon brevicaudatus is widespread across United Republic of Tanzania (Tanzania) and occurs in a small part of southeast Kenya. It has an estimated extent of occurrence of 163,800km^2, of which less than 10% is forest. It is reported to be locally abundant at lower elevations in forest and disturbed habitats, but less common in disturbed forests at higher elevation (ca. 800m)[1]. Much of its habitat is within protected areas[2]. Surveys conducted in 2009 in the Amani Nature Reserve in Tanzania (at ca. 900m elevation), encountered it very infrequently along edge habitats. The species is subject to harvesting for the international pet trade. US trade data reported an average of almost 5000 imported annually into the USA between 2004 and 2013, almost all from Tanzania[6]. The species is also found in offers for sale in Europe. Shipments of *Ri. brevicaudatus* are apparently frequently mislabelled as other species of *Rhampholeon* and *Rieppeleon*. Classified in the IUCN Red List as Least Concern (2014).

Rieppeleon kerstenii occurs widely in Ethiopia, Kenya, Somalia, and Tanzania. It is reported to be common, although there are no quantitative population data. US trade data indicate an average of ca. 8200 imported annually between 2004 and 2013 into the USA, all from Tanzania[6]. The species is also found in offers for sale in Europe. Some shipments labelled as *Ri. kerstenii* are reportedly in reality to comprise *Ri. brevicaudatus* and *Rh. temporalis*. Classified in the IUCN Red List as Least Concern (2014).

Rhampholeon spectrum occurs in Cameroon, Equatorial Guinea (including Bioko), Gabon and Nigeria. It has been reported as common in montane areas of Cameroon and in parts of southern Nigeria but is believed to be rarer in the lowlands and degraded habitats; in southern Nigeria suitable habitat for the species is believed only to represent 5% of its original extent. According to US trade data, it is the most commonly traded *Rhampholeon* spp. with ca. 555 specimens per year imported into the USA between 2004 and 2013, mainly from Equatorial Guinea and Cameroon[6]. *Rh. spectrum* is apparently widely available for sale in Europe. Classified in the IUCN Red List as Least Concern (2010).

US trade data also shows import of an average of ca. 350 live specimens of *Rhampholeon* spp. per year between 2004 and 2013, the majority of which were from Tanzania[6].

Information on the remaining species is as follows:

Rieppeleon brachyurus occurs widely in Tanzania, northern Mozambique and Malawi. The species is believed likely to be common although there are no population estimates. US trade data report a small number of imports (ca. 33 per year for 2004-2013) into the USA[6]. Classified in the IUCN Red List as Least Concern (2014).

Rhampholeon acuminatus is currently only known from a single locality in the Nguru South Catchment Forest Reserve in Tanzania, where there is an estimated 28km^2 of suitable habitat remaining[3]. The population is likely to be small, due to its limited range. Around 70 specimens were imported in total into the USA between 2004 and 2013 with the majority of imports in 2013; two additional shipments totalling 107 individuals were refused in 2010 and 2013[6]. The species is considered desirable in the pet trade. It is regularly found in offers for sale in Europe and the USA. The species is also believed to be particularly at risk from loss of habitat owing to its limited range. Classified in the IUCN Red List as Critically Endangered (2014).

Rhampholeon nchisiensis is mainly confined to Malawi, with peripheral occurrence in Tanzania and Zambia[4]. Its overall range extends over some 12,600km^2; only 10% of this is suitable forest habitat[5]. There is no quantitative information on abundance. It is reportedly imported into the pet trade in limited quantities every few years, and can be found in offers for sale in Europe and the USA, although five live imports to the USA have been recorded between 2004 and 2013[6]. Classified in the IUCN Red List as Least Concern (2014).

Rhampholeon temporalis is endemic to the Usambara Mountains of Tanzania where there is believed to be less than 300km^2 of suitable habitat remaining, some of which may be of low quality. In the East Usambara Mountains, average population density is reported to be just over 30 per ha$^-$ with lower densities towards forest edges. Not recorded in US import data, but the species can be found in offers for sale in Europe and the USA. Shipments of *Rh. temporalis* are apparently frequently mislabelled as other species of *Rhampholeon* and *Rieppeleo*n in the international trade. Classified in the IUCN Red List as Endangered (2014).

Rhampholeon uluguruensis is endemic to the Uluguru Mountains of Tanzania, where it occurs in ca. 280km^2 of suitable habitat. No quantitative data on population abundance, but populations are assumed to be stable. Ca. 350 specimens were imported into the USA between 2004 and 2013, almost all in 2012 and 2013. The species can also be found in offers for sale in Europe. Classified in the IUCN Red List as Least Concern (2014).

Rhampholeon viridis is endemic to the Pare Mountains in northern Tanzania, where there is an estimated 152km^2 of suitable habitat scattered over a much larger area. The population is believed to be decreasing due to severe loss of habitat. US trade data report that ca. 200 specimens were imported into the USA between 2004 and 2013, almost all in 2013[6]. The species can also be found in offers for sale in Europe. Classified in the IUCN Red List as Endangered (2014).

Rhampholeon moyeri is endemic to the eastern Udzungwa Mountains, Tanzania. It is occasionally available in European market. Classified in the IUCN Red List as Least Concern (2014).

Rhampholeon boulengeri is widespread in Burundi, Democratic Republic of Congo, Kenya, Rwanda, Tanzania and Uganda. 47 imported into the US from Burundi and Democratic Republic of the Congo between 2004 and 2013. Known to be offered for sale in Europe. Classified in the IUCN Red List as Least Concern (2014).

Of the remaining species most have limited and/or fragmented ranges: *Rh. bruessoworum (*IUCN Critically Endangered, 2014), *Rh. chapmanorum* (IUCN Critically Endangered, 2014) and *Rh. hattinghi* (Critically Endangered, 2015) have an area of occupancy ranging from ca. 1km^2 to 5km^2, while *Rh. beraducci* (IUCN Vulnerable, 2014*)*, *Rh. nebulauctor* (IUCN Vulnerable, 2014) and *Rh. tilburyi* (IUCN Critically Endangered, 2014)*,* are limited to areas of occupancy between ca. 12.5km^2 and 18km^2. *Rh. platyceps* (IUCN Endangered, 2014), occurs in forest fragments in Malawi totalling 61km^2, *Rh. maspictus* (IUCN Near Threatened, 2014) is limited to intact forest patch of 79km^2. *Rh. gorongosae* has an area of occupancy of around 100km^2 (IUCN Least Concern, 2014). *Rh. marshalli* (IUCN Vulnerable, 2014) inhabits a severely fragmented area of ca. 540km^2 which is subjected to ongoing forest transformation. None of these species are known to be in trade.

At least eight species of *Rhampholeon* and almost certainly all species of *Rieppeleon* occur in protected areas.

Available information indicates that national protection is limited. In Cameroon, capture of *Rhampholeon* spp. requires a permit although this rule is reportedly often disregarded. In Kenya, all chameleon species are protected. In May 2016, it was reported that Tanzania had banned the export of live reptiles until proper procedures to control trade were implemented[7].

One species in the genus *Bradypodion* (*B. spinosum*, endemic to Tanzania) which was included in Appendix II in 1977 is now widely regarded as a species of *Rhampholeon*, (*Rh. spinosus)* but is still recognised as *Bradypodion* under CITES taxonomy. Very little trade in this species is reported in the CITES

Trade Database: a total of 147 live specimens reported in trade between and 1993 and 2011, 93 of which were reported as born in captivity.

Rhampholeon and *Rieppeleon* spp. are reported to be subject to ongoing misidentification in trade, both between species within each genus and between genera, due to their similar morphological characteristics, particularly colouration and physical size. Shipments labelled "assorted pygmy chameleons" containing wild-caught *Rhampholeon* spp. have included the CITES-listed *B. spinosum* (*Rh. spinosus*).

Analysis: Of the three species in the proposal that are known to be in trade in any number, *Ri. kerstenii* and *Rh. spectrum* are both widespread species which are not known to be under threat. It is very likely that the populations of both are large. *Ri. kerstenii* is known to have been exported in some numbers from one (of four) range States, *Rh. spectrum* in considerably smaller numbers from two (of four) range States. It does not appear that either meets the criteria for inclusion in Appendix II in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)*. The third species, *Ri. brevicaudatus* is widely distributed and reportedly locally abundant in Tanzania, also occurring marginally in Kenya, and is currently considered not under threat. It has been exported from Tanzania in some numbers, but it seems unlikely that harvest for export is reducing the population to a level at which its survival might be threatened, or at which it might become eligible for inclusion in Appendix I in the near future.

Of the remaining species eight (*Ri. brachyurus***,** *Rh. acuminatus, Rh. boulengeri, Rh. moyeri, Rh. nchisiensis, Rh. temporalis, Rh. uluguruensis* and *Rh. viridis*) have been recorded in trade, the exact level of which is unknown but is likely to be small. Only *Rh. acuminatus, Rh. temporalis* and *Rh. viridis* are currently considered threatened. There is insufficient information to determine whether either of these three meet the criteria in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)*. It is unlikely that any of the others do.

Some of the remaining species are believed to have very restricted or fragmented ranges but are not known to be in trade.

Distinguishing between all species of *Rhampholeon* and *Rieppeleon* may be difficult and there are reports of mislabelling of species in trade. Shipments of unnamed *Rhampholeon* have reportedly included the Appendix-II listed *Bradypodion spinosum* and it might be argued on this basis that the other species meet the criteria for inclusion in Annex 2 b. However, it should be noted that with the exception of the geographically distant *Rh. spectrum* (which does not appear to meet the criteria in Annex 2 a of the Resolution), all reported trade in species in these genera (and in *B. spinosum*) originates from a single range State (Tanzania). It would appear that species with Tanzania as a range State would meet the criteria in Annex 2b A (lookalike) in that individuals resemble specimens of a species of *Bradypodion spinosum* so that enforcement officers who encounter specimens of CITES-listed species are unlikely to be able to distinguish between them. These include *Rh. acuminatus, Rh. beraduccii, Rh. boulengeri, Rh. moyeri, Rh nchisiensis, Rh. temporalis, Rh uluguruensis, Rh. viridis, Ri. Brahyurys, Ri brevicaudatus* and *Ri kerstenii.* Although other species may also resemble *B. spinosus,* it is unlikely that enforcement officers outside of Tanzania would need to distinguish specimens of them from *B. spinosum..* It is unclear whether any species elsewhere would meet these criteria.

Reviewers of summary information only: C. Anderson, K. Tolley, P. Shirk and S. Chng.

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Inclusion of Psychedelic Rock Gecko *Cnemaspis psychedelica* in Appendix I

Proponents: Viet Nam and European Union

Summary: The Psychedelic Rock Gecko *Cnemaspis psychedelica* is moderate-sized gecko mostly active during the day. It is known from the island of Hon Khoai off the southern tip of Viet Nam, which has a total area of $8km^2$, and has recently been reported as occurring on the smaller island of Hon Sao nearby to the southeast; its presence on other smaller neighbouring islands is unconfirmed. On Hon Khoai it is found on granite boulder outcrops in the shade of the surrounding dense vegetation. It has much brighter colouration than other species of *Cnemaspis* (of which there around 75), with both sexes having orange forelimbs and a blue-grey body. Females are reported to lay a single clutch of two eggs once a year, incubated in communal clutches on the underside of overhanging boulders.

The species has been described as very common and abundant in suitable habitat. It is unclear how much suitable habitat there is, although given the small size of the island it is unlikely to be extensive. A population assessment carried out on Hon Khoai in November 2015 and January 2016 estimated the total population to be up to 732 individuals, with an effective mature population of 507.

The island of Hon Khoai is an outpost of the Ca Mau border guard and therefore public access to the island should be prohibited, however several tourist websites offer trips to visit the island. A fishing port is under construction on the island, which will impact suitable habitat and increase the number of people living on the island. In addition, introduced Long-tail Macaques *Macaca fascicularis* have been observed eating the gecko and its eggs.

Cnemaspis psychedelica was first described in 2010, and live individuals have been offered for sale for the pet trade since 2013. The largest market appears to be in the EU and the Russian Federation but it is also advertised on the internet in the USA. The species commands a high price, likely because of its bold colouration and rarity on the market. Online it is generally advertised in breeding pairs, with suggestions that this species is easy for hobbyists to breed[1]. The species is reportedly easy to collect in the wild[2].

Currently there are no protection measures in place for this species or its habitat, although trapping and exportation of forest animals is only allowed by permit. A captive breeding programme has been started which has recently reported successful reproduction.

Analysis: *Cnemaspis psychedelica* has a very restricted area of distribution, occurring in only one or two known locations. It has a low reproductive rate. Indications are that it also has a very small wild population which may be vulnerable to extrinsic factors including predation by introduced species and collection for export. It would appear therefore to meet the biological criteria for inclusion in Appendix I. The species is or may be affected by trade.

Reviewers of summary information only: L. Grismer and S. Altherr.

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Inclusion of Turquoise Dwarf Gecko *Lygodactylus williamsi* in Appendix I

Proponents: United Republic of Tanzania and European Union

Summary: The Turquoise Dwarf Gecko *Lygodactylus williamsi* is a species endemic to eastern Tanzania where it is known to occur in four isolated tropical lowland forest patches (Kimboza, Ruvu, Mbagalala and Muhalama) in the Uluguru foothills in the Morogoro Region. Within these forests it exclusively occurs on a species of screwpine *Pandanus rabaiensis*. It has an estimated area of occurrence of 20km² and area of occupancy of 8km². The species exhibits distinct sexual dichromatism; males have a striking turquoise-blue back while females and immature males are a greenish-bronze. Reproduction is reported to occur throughout the year with a relatively high output of offspring. Generation time is not known.

The only available quantified information on population status is for that in the Kimboza Forest Reserve, estimated in 2009 at around 150,000 adults based on visual encounter surveys and mean number of specimens found per *P. rabaiensis.* Populations elsewhere have not been quantified; those in the Mbagalala and Muhalama forest patches are believed to be small due to the small number of *P. rabaiensis* trees.

The estimated population in Kimboza forest in 2009 was believed to be around one-third smaller than the carrying capacity, based on the number of *P. rabaiensis* trees. If this represents an actual decline, this may be a result of collection pressure for international trade, which has reportedly been high since 2004. Reports suggest that some 22,000 were collected in 2005 and some 8000 per year in 2006 and 2007.

Despite legal protection (see below), the forests in which the species occurs are also reported to be affected by logging, collection of firewood, conversion to agricultural land and mining of the limestone substrate on which *P. rabaiensis* grows. There have also been reports that *P. rabaiensis* trees are cut down to collect the geckos.

It has been offered for sale online in the recent past in the USA and in Europe, at prices of ca. USD 30-250 per individual. The species has been reported as relatively easy to breed in captivity and specimens reported as captive-bred are offered for sale on the internet[1,2].It has been reported that males can lose their striking coloration in captivity, which may result in continuing demand for wild-caught individuals.

Kimboza and Ruvu are both Forest Reserves, protected under the 2002 Forest Act and managed by the Tanzania Forest Service. Collection of wild specimens within these areas requires a license. According to officials from the Tanzania Wildlife Research Institute, collection and export of *L. williamsi* has never been licensed, indicating that all trade in it is illegal[1,2]. It has been reported that wild-caught specimens of *L. williamsi* are frequently deliberately mislabelled and exported as *Lygodactylus* spp. or as *L. capensis* to facilitate trade.

The species is classified in the IUCN Red List as Critically Endangered (2012).

Analysis: *Lygodactylus williamsi* has a very restricted area of distribution in which the quality of habitat is declining. The abundance of the species is also likely declining, due to habitat loss, and possible due to illegal collection for the international per trade. It would therefore appear to meet the criteria for inclusion in Appendix I in *Res. Conf. 9.24 (Rev. CoP16*).

Reviewers of summary information only: M. Bungard, S. Chng and S. Nash.

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All viewed on 23rd May 2016.

Inclusion of Masobe Gecko *Paroedura masobe* in Appendix II

Proponents: Madagascar and European Union

Summary: The Masobe Gecko *Paroedura masobe* is one of 15 species of Malagasy Ground Geckos in the genus *Paroedura*. It is endemic to Madagascar with a limited range in low elevation humid forest, typically between 300 and 600m above sea level in the east of the country in the province of Toamasina. In 2011 it was assessed as having an extent of occurrence of no more than 410km^2, with ca. 100km^2 of suitable habitat remaining. The species is believed unlikely to occur more widely than its currently known distribution[1]. Remaining habitat is fragmented and the population is suspected to be declining, although no quantitative information is available[1]. It has been reported that there is a relatively stable population in the Betampona Natural Reserve[2]. Surveys conducted over a period of seven months in the reserve in 2007 and 2010 found just two male specimens. However, in 2013 surveys in an area where the species was known to be present found 23 individuals in three weeks[2]. As well as Betampona Natural Reserve, the species is also known to occur in Zahamena National Park and the newly established protected area of Ambohidray.

Continuing declines in the quality and extent of habitat are believed to be the most important factors affecting the species. Concern has also been expressed about the possible impact of harvest for the international pet trade. The species is highly attractive, and local collectors have indicated that it is one of the most profitable reptile species to collect in the area. There are reports of illegal collection in Betampona Reserve and Zahamena National Park. There is no reported domestic use of the species.

A report (currently in preparation) from two regional administration centers of Analamanga and Antsiranana recorded exports of just under 2500 individuals between 2000 and 2005, before controls on exports were put in place in 2006 (see below). Import data from the USA recorded just under 300 wild-caught individuals imported from Madagascar between 2011 and 2015, most (ca. 250) in 2014 and 2015[3]. Additional imports of 53 captive-bred individuals from Canada, Germany and the United Arab Emirates were reported between 2011 and 2015. Online surveys between 2011 and 2016 also reveal the presence of the species in the international pet trade with traders in Europe and the USA offering specimens for ca. USD 380-950[4,5]. There is also evidence of it being kept as a pet in Japan[6]

Since 2006, *P. masobe* has been listed under Category I, Class I of the National Decree 2006-400, which strictly prohibits the hunting, capture, possession and commercial trade of the species except under license for scientific purposes, breeding or exhibitions[7]**.** It has been reported that there is an annual quota of ten individuals that can be legally exported[1]. Trade data above indicate that this figure has been routinely exceeded.

The species is classified in the IUCN Red List as Endangered (2011). The assessment noted that if further research into the distribution of *P. masobe* revealed that it has a true extent of occurrence is less than 100km^2, it would merit reclassification as Critically Endangered.

Analysis: *Paroedura masobe* has a restricted distribution in an area where habitat is fragmented and declining. It is sought after in the international pet trade with a few hundred specimens having been exported annually in recent years since 2006, exceeding the apparent quota. There is no information on population densities, overall population size or population trends, although the species has reportedly only infrequently been encountered in one of the protected areas within its range in recent years. The species may meet the criteria in Annex 2 aA of *Res. Conf. 9.24 (Rev. CoP16)* in that regulation of trade is necessary to avoid it becoming eligible for inclusion in Appendix I in the near future.

Reviewers of summary information only: S. Chng, J. Janssen and G. Rosa.

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Inclusion of Earless Monitor Lizard Lanthanotidae in Appendix I

Proponent: Malaysia

Summary: The family Lanthanotidae comprises a single species, the Earless Monitor Lizard *Lanthanotus borneensis*, endemic to the island of Borneo where it is known from Indonesia and Malaysia. It is burrowing, semi-aquatic and nocturnal, being generally recorded from damp soil along river banks. It is oviparous, producing clutches of ca. 2 to 6 eggs[1, 2, 3].

There are only a few occurrence records in the wild; many of these have come from incidental encounters posted on social media. Records are from the coastal lowlands of Sarawak, Malaysia, and from northwest Kalimantan, Indonesia in Landak and Sanggau Districts. There is also a single record of the species from East Kalimantan. Its presence has not been confirmed in Brunei Darussalam. It is generally regarded as rare, but is known by local residents, some of whom considered it to be common, at several sites in West Kalimantan[4]. It has been suggested that it may have a wider distribution than generally thought, with its scarcity in scientific collections due to its nocturnal and secretive life habits[4]. However, extensive state-wide herpetofauna surveys are reported to have failed to find it. All sites where it is known are from below ca. 300 m altitude[5].

The species is believed likely to be affected by habitat loss and alteration through widespread conversion of forests to agro-industrial and forestry plantations as well as forest fires and swidden agriculture. However, residents in West Kalimantan indicated that the species was most often encountered in immature forest, “tembawang” (cultivated forest planted with fruit trees) and along river edges. This indicates that the species can survive in at least partially modified habitats. An individual was found in 2008 by a survey team in West Kalimantan in a tembawang forest within a recently developed oil palm plantation[4] .

There has been a rapid emergence of illegal trade in this species since 2013. Intelligence reports suggest that more than 40 individuals were collected in spring 2014[6] and at least 95 individuals were offered for sale on the internet during a 17 month period[7]. There have been seizures of at least 35 *L. borneensis* from Indonesia between October 2015 and March 2016[8, 9, 10]. Specimens advertised as captive-bred are recorded in trade. This species is highly desirable, reported to be reaching prices of USD 7500 to 9000 on the illegal market.

This species has been fully protected throughout its range in Malaysia since 1971, Indonesia since 1980, and Brunei Darussalam since 1978, and it has never been legally exported. Therefore all specimens in trade appear to have been illegally obtained, or are the progeny of specimens illegally obtained.

Analysis: *Lanthanotus borneensis* is a rarely observed species known from relatively few locations, a number of which have only recently been discovered. It may be more widespread than current records indicate. There are no population estimates although there are indications that it may be at least locally not uncommon. It is believed likely to be declining owing to habitat loss and alteration, although there are indications that it can survive in modified habitats. Overall, there is insufficient information to determine whether it meets the criteria for listing in Appendix I. The species is in demand for, and potentially affected by, trade. Harvest and trade is illegal in all range States.

Reviewers of summary information only: D. Bennet, I. Das and V. Weijola.

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Transfer of Chinese Crocodile Lizard *Shinisaurus crocodilurus* from Appendix II to Appendix I

Proponents: China, Viet Nam and European Union

Summary: The Chinese Crocodile Lizard *Shinisaurus crocodilurus* is the only living member of the family Shinisauridae[1]. It is a semi-aquatic lizard ca. 40cm in length that occurs in undisturbed and densely vegetated rocky streams in tropical evergreen broadleaf forests in southern China and northern Viet Nam. Adults have a territory of around 10m^2. In the wild maturity is reached at two to four years of age. Pregnancy lasts between nine and 11 months and between two and 12 fully developed young are born into the water. The species was included in Appendix II in 1990.

The current population is estimated at 950 individuals in China and fewer than 100 in Viet Nam. In 1978 China's total population was estimated at about 6000 individuals decreasing to about 2500 individuals in 1990. Sub-populations are fragmented due to loss of habitat, with 19 known in China (of 10-350 individuals) in an area of ca. 460km^2 and three in Viet Nam (of 17-22 mature individuals) spread over an area of about 1500km^2. The Chinese sub-populations are the most studied, and show significant population declines over the last 30 years, ascribed to collection for the pet trade and local consumption and use in traditional medicine. Five sub-populations appear to have been extirpated completely, and the remaining 19 have recorded declines of up to 90%. At some streams in Viet Nam, this species was reported to have declined or disappeared between 2013-2014, due to increased accessibility to the area, and electrofishing[2].

Three of the eight Chinese sub-populations fall within protected areas, although only one of these is considered not under threat. The sites of the other Chinese populations are believed to be affected by agricultural conversion, logging and water pollution. All three Vietnamese sites fall within protected areas but are surrounded by cultivated lands, preventing migration between sites. At least one site is affected by coal mining, and ongoing developments such as new roads and tourist and religious sites have increased accessibility to the habitat.

Collection for the international pet trade and domestically for food and traditional medicine is regarded as the major cause of the recent population decline. Anecdotal evidence suggests locals caught as many as 50 per day to sell at markets, until the numbers became too few to collect.

There is demand for this species for the pet trade, and it is available for sale at local and international reptile markets; there is also apparently considerable trade on internet platforms.

The CITES Trade Database records an average of about 23 live individuals in trade each year for the period 2004-2014, except for 2005 when 400 live individuals were recorded as exported from Lebanon and originating in Kazakhstan (although no corresponding export has been reported from Kazakhstan).[3] Other than this the largest recorded exporter is Germany. The largest importers are Germany, Japan, Thailand and USA. Virtually all this trade has been recorded as in captive-bred animals. This is believed to be implausible, as the species has been reported to be challenging to keep and to suffer high levels of mortality in captivity[2].

Multiple dealers have reported that wild-caught specimens from China have been labelled as captive-bred. Several dealers in Viet Nam advertise specimens for sale from "farms" but there is no evidence of breeding facilities sufficient to support this, and one Vietnamese hobbyist reported that "farmed" specimens were in fact wild-caught[2].

Although captive-breeding techniques for this species are improving there is concern that the level of demand is much greater than captive production could supply.

Shinisaurus crocodilurus has been a Class I protected species in China since 1989, meaning no unauthorised collection or trade may take place. In Viet Nam the species is not explicit protected, although such protection is under consideration and it is an offence to hunt or trap an animal in a protected area. There are conservation programmes including breeding and monitoring for *Shinisaurus crocodilurus* in place in both China and Viet Nam.

Shinisaurus crocodilurus is classified in the IUCN Red List as Endangered (2014)[4].

Analysis: *Shinisaurus crocodilurus* appears to have a restricted and fragmented area of distribution, with a small population each of whose sub-populations is small. The population is also believed to have undergone a marked decline, ascribed to habitat destruction and collection for the pet trade and local consumption. It would therefore appear to meet the biological criteria for inclusion in Appendix I. The species is or may be affected by trade, as defined in Annex 5 of the Resolution.

Reviewers of summary information only: M. Auliya and S. Altherr.

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Inclusion of Mount Kenya Bush Viper *Atheris desaixi* in Appendix II

Proponent: Kenya

Summary: The Mount Kenya or Ashe's Bush Viper *Atheris desaixi* is a medium-sized primarily arboreal venomous snake confined to mid-altitude forests in central Kenya, with two known populations, one in Igembe and Ngaya forests in the Nyambene hills, and the other at Chuka, south eastern Mt Kenya forest. These areas total less than 10km^2 between them. However, the species is secretive and may be more widespread than records indicate[1]. Population data are limited, but a rapid field assessment of the species in 2010 yielded only 12 individuals in Chuka, whilst searches around Igembe and Ngaya forests were unfruitful. Local snake collectors reported that numbers had declined remarkably over the years. Females bear 10 to 13 live young per brood.

Ngaya forest is a government protected community forest. The species is believed to be affected by habitat loss and degradation, the forests in which it occurs being under high pressure from livestock grazing, fuel-wood collection, logging and agricultural expansion. It has been suggested that collection for illegal trade may also have had an impact on the species.

Atheris desaixi has been protected in Kenya since 1982. Current legislation prohibits both collection from the wild and export. In 1999/2000, 27 individuals were reportedly illegally exported by one trader[2], and US trade data records the import into the USA of 16 wild individuals in 2007 to 2008. Three snakes were rescued from a local snake collector during the rapid assessment survey in 2010. The species has been reported to fetch values of up to USD 4500 in Europe. However, there are no data to indicate whether this trade is increasing or not. The species appears to be relatively rare in captivity and the size of the market (specialist collectors) is almost certainly small[3]. Hobbyists report some captive breeding success.

Analysis: Available information indicates that *Atheris desaixi* has a very restricted range where it is subject to habitat loss and degradation. A 2010 survey found it rare, with reports of a decline in population at one site. There is evidence of demand on international markets. However, the market (specialist collectors) is almost certainly small. Recorded instances of such trade are at a very low level. All current trade from the range State is illegal. Because all trade in wild specimens is theoretically already fully regulated, the species does not appear to meet the criteria for inclusion on Appendix II (it may meet the criteria for inclusion in Appendix I).

Reviewers of summary information only: S. Spawls, J. Penner, S. Chng and J. Janssen.

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Inclusion of Kenya Horned Viper *Bitis worthingtoni* in Appendix II

Proponent: Kenya

Summary: The Kenya Horned Viper *Bitis worthingtoni* is a small venomous snake endemic to Kenya, occurring in high altitude areas of grassland and scrub. The range is limited, the species having been recorded patchily within an area of some 10,000km^2. The main population occurs within the Rift Valley plateau in areas adjacent to Lake Naivasha. There may be additional populations in Hells Gate and Lake Nakuru National Parks and in the areas south of Naivasha towards Mt. Longonot and Kedong valley and north through Gilgil and Elmenteita into Nakuru. There are no recent records from Uasin and Kinangop plateaus, where it has been recorded in the past. There are no population or density estimates, although the species has been described as relatively rare. Population depletions have been inferred based on degradation and loss of suitable habitat. It has also been suggested that populations are depleted in areas where it has been collected in the past, but the basis for this is not clear. Females bear between seven and 12 live young per brood. At least part of the range is within national protected areas and private wildlife sanctuaries.

Bitis worthingtoni has been protected in Kenya since 1989. Current legislation prohibits both collection from the wild and export. In 1999/2000, 37 individuals were reported to have been illegally exported from Kenya by one trader[1]. In 2013 a British man was charged with possession of five specimens[2] and one individual was recently offered for approximately USD 1100 on the European market.

Analysis: *Bitis worthingtoni* has a moderately extensive range in Kenya, although is patchily distributed within it. The species has been described as relatively rare, although suitable habitat probably remains extensive within its range. There is evidence of demand in international markets. However, the market (specialist collectors) is almost certainly small[3]. All current trade from the range State is illegal. Recorded instances of such trade are at a very low level. Given observed volumes of trade and likely limited demand, it seems unlikely that the species meets the criteria for inclusion in Appendix II in *Res. Conf. 9.24 (Rev. CoP16).*

Reviewers of summary information only: S. Spawls, J. Penner, S. Chng and J. Janssen.

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Information not referenced in Summary section is from the Supporting Statement.

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Inclusion of the following six species of the Family Trionychidae in Appendix II: *Cyclanorbis elegans, Cyclanorbis senegalensis, Cycloderma aubryi, Cycloderma frenatum, Trionyx triunguis*, and *Rafetus euphraticus*

Proponents: Burkina Faso, Chad, Gabon, Guinea, Liberia, Mauritania, Nigeria, Togo, and United States of America

Summary: Softshell turtles belonging to the Family of Trionychidae are highly aquatic species that generally prefer slow-moving water with muddy or sandy bottoms. Currently some 33 species in 13 genera are recognised. Three species occur in North America, six in Africa, the Mediterranean and the Middle East, and the remainder in more eastern parts of Asia. With the exception of the widely farmed *Pelodiscus sinensis,* the eastern Asian species are all variously included in either Appendix I or Appendix II. One North American subspecies, *Apalone spinifera atra,* is included in Appendix I. This proposal seeks to add all Trionychidae native to Africa, the Mediterranean and the Middle East to Appendix II.

Most softshell turtles reach maturity at 10 to 15 years old, and may live for 60 years or more. They may lay several clutches a year of 10 to 100 eggs (depending on the species) but few reach maturity.

Softshell turtles are in generally very difficult to survey and there is virtually no quantitative information on overall populations, population densities or trends for any of the species. For species that are used there is in almost all cases no market information to indicate changes in supply or in rates of use. Changes in population have sometimes been inferred from habitat changes or evidence of use, but appear often to be based on supposition.

Cyclanorbis elegans is known from wide rivers with muddy substrate in disjunct locations in the Sahel zone of sub-Saharan Africa. The species is difficult to distinguish from *C. senegalensis.* It has rarely been recorded in surveys. Known international trade is at an extremely low level. Collection of turtle eggs for consumption occurs within its range and changes in water management may affect its habitat, although there is no information on the impact of either of these. The species was observed in a pet trade market in Hong Kong between 2000 and 2003 (quantity unknown)[1]. Classified by IUCN as Lower Risk/near threatened (1996 – needs updating)[2].

Cyclanorbis senegalensis is widespread in West Africa and found in a range of aquatic habitats[3]. Some populations are reported as harvested, with an inference that this is leading to declines; other, unharvested populations reportedly remain abundant. Known international trade, presumed for the international pet trade, is at a very low level. Trade data indicate 70 live specimens were imported into the USA between 2005 and 2013, 54 of which were recorded as wild-sourced[4]. More than 50 live specimens were reported as exported from Togo in 2013[5]. In the past, this species has been exported under the name of *Trionyx triunguis* from Togo[5]. Classified by IUCN as Lower Risk/near threatened (1996 – needs updating)[6].

Cycloderma aubryi occurs in waterways in rainforests in central Africa. Reported to be collected extensively for local consumption which is inferred as having led to declines. Trade data records the import of negligible numbers (20 between 2007 and 2013) into the USA[4].

Cycloderma frenatum occurs chiefly in Malawi but extends into Mozambique, United Republic of Tanzania and Zimbabwe. It is reportedly common in the shallow southern waters of Lake Malawi, rare in the deeper northern waters. Historically this species has been collected for consumption across much of its range, but in some areas only eggs are consumed. Until recently the level of harvest was not thought to have a significant impact on the population. In 2013 an illegal butchery said to be processing 50 adults per day was shut down in Malawi[7]. This was reportedly in an area where it has not traditionally been consumed and was apparently to meet demand from Asian nationals in Malawi, although it has been suggested that the processed meat and shell were for export to East Asia[8]. Trade data records imports into the USA of 52 live specimens between 2008 and 2013, 50 of which were in 2013[4]. The species was observed in a pet trade market in Hong Kong between 2000 and 2003 (quantity unknown)[9]. Classified by IUCN as Lower risk/near threatened (1996 – needs updating)[10].

Trionyx triunguis inhabits fresh and brackish waters across Africa, and around the eastern Mediterranean. It is a large species, growing up to 80cm carapace length. It is best known from its Mediterranean populations which are generally believed to be declining. Populations in Africa are less well known; there is anecdotal evidence of major decline in catch per unit effort in parts of West Africa. Populations in central

Africa are suspected to be declining at a low rate because of harvest. In Egypt, there have been no recent records from the Nile below Lake Nasser but is reportedly considered abundant upstream of the dam. The species is believed to be affected by habitat alteration and incidental catch in nets. It is consumed for subsistence in parts of its range; its shells are sold in fetish markets in Togo and Benin[11]. In Israel the species is regarded as highly threatened, with the largest subpopulation believed to comprise some 50 individuals. Illegal harvest for local consumption has been identified as a threat. Trade data show a very small number imported as live specimens into the USA (ca. 100 declared as wild in 2004-2013 in total[4]).

Rafetus euphraticus is a little known species from Iran, Iraq, Syria and Turkey. It is reported to now be rare in Turkey, but has been reported as very abundant in marshes in Iraq. Habitat in Iraq had been reduced by draining but some has now been restored. Habitat degradation, pollution and killing by fishermen have been identified as factors affecting the species. It is not harvested for meat but parts of it are reportedly consumed for medicinal purposes in some of its range. There is no evidence of international trade. Classified by IUCN as Endangered (1996 – needs updating)[12].

Softshell turtles are heavily exploited in Asia. Demand, primarily for human food consumption and also traditional medicine, is not species-specific. The main parts in trade (meat and shell processed to varying degrees) are generally extremely difficult to identify to species level in the form in which they are traded.

Analysis: Information on all six species is generally scarce. Although declines in some parts of the range of some species have been reported, in no case is there any indication of major species-wide declines. It is not known if there is any significant level of international trade at present in any of the species. Where any information on harvest is available, this may be largely or entirely for domestic consumption. Therefore, there is insufficient information to determine whether any of the species meet the criteria for inclusion in Appendix II in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)*.

Species of softshell turtle in trade resemble each other in the parts in which they are mainly traded. If it is concluded that some of the species considered here meet the criteria for inclusion in the Appendices then the others would meet the criteria in Annex 2 b of *Res. Conf. 9.24 (Rev. CoP16)* (lookalike). The species proposed here cannot necessarily be identified with ease from other members of the Trionychidae family from other parts of the world in the form that they are traded, although the trade routes may assist enforcement agents in distinguishing between species from Asia, from Africa and from North America.

Reviewers of summary information only: P.P. van Dijk and G. Segniagbeto.

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Transfer of Tomato Frog *Dyscophus antongilii* from Appendix I to Appendix II

Proponent: Madagascar

Summary: The Tomato Frog *Dyscophus antongilii* is an attractive orange-red coloured frog, one of three members of the genus *Dyscophus*, all endemic to Madagascar. The two other members of the genus, *D. insularis* and *D. guineti,* are the subjects of Prop. 38, for inclusion in Appendix II; neither is currently listed in the Appendices.

The species has a relatively wide distribution in the east and northeast of Madagascar. The precise limits of the range are uncertain, in part because of possible confusion with the very similar *D. guineti*. Although there is no overall population estimate, one expert has noted that based on a mark-recapture study in part of the range, and the life history characteristics of the species, it is reasonable to presume that populations may reach hundreds of thousands individuals[1]. In 2008 it was said to be locally abundant, especially in and around Maroantsetra[2] and in the Ambatovaky Special Reserve region. Urban expansion is taking place in parts of the range, notably around Maroantsetra and this may be leading to some reduction in population[3]. However, the species is said to be adaptable, and has been recorded in urban areas and other altered habitats. It breeds several times per year after rainfalls, and lays 1000-15,000 eggs.

Dyscophus antongilii was included in CITES Appendix I in 1987 as it was harvested for the international pet trade and believed at the time to have a restricted range[4]. The CITES Trade Database includes a small amount of exports from Madagascar between 2000 and 2007, including 75 live frogs, and 400 specimens for scientific purposes. There was a small amount of reported trade between non-range States (76 live frogs over the same period, all bred in captivity or born in captivity); the majority of which were exported by Germany or Latvia. No trade in *D. antongilii* has been reported to CITES since 2007 in wild, or captive-bred specimens. However, a seizure in Malaysia in 2010 of 47 *D. antongilii* of Madagascan origin indicates demand for the species continues[5]. The similar species *D. guineti* and *D. insularis* are traded in some volume (see analysis for Prop. 38).

Dyscophus antongilii is currently listed as a protected species in Madagascar (Category I Class I Decree 2006-400) which means harvest is only allowed for scientific purposes[6]. Under domestic legislation, a transfer to Class II, which would allow for some harvest for commercial purposes outside of protected areas, would necessitate additional studies including population inventories[6]. Current the population is not actively monitored nor is it the subject of specific management measures. The Supporting Statement notes that the Madagascar Scientific Authority will recommend conservative quotas for commercial collecting, but does not provide any detail on any proposed export quota. All Madagascan amphibian species currently included in Appendix II are subject to conservative export quotas[7].

Dyscophus antongilii is classified in the IUCN Red List as Near Threatened (2008).

Analysis: *Dyscophus antongilii* does not have a restricted range nor a small population. There are no indications that the population is undergoing a marked decline. The species does not therefore appear to meet the biological criteria for inclusion in Appendix I.

The precautionary measures in Annex 4 of *Res. Conf. 9.24 (Rev. CoP16)* should be met. A conservative collection quota is proposed, only to be permitted once population inventories have taken place. Currently other Appendix-II listed Madagascan amphibian species are exported under similar measures. Export of these other species has been closely scrutinised by Parties, and has been agreed to demonstrate compliance with Article IV of the Convention[8] and therefore one would expect trade of this species, if transferred to Appendix II to comply with Article IV as well. Inclusion in Appendix II of the similar *Dyscophus guineti* and *D. insularis*, as proposed by Madagascar in Prop. 38, would help ensure enforcement controls for this species were effective.

Reviewers of summary information only: M. D. Kusrini and C. Ratsimbazafy.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[3] Ratsimbazafy, C. (2016) *In litt*. to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
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Inclusion of False Tomato Frog *Dyscophus guineti* and Antsouhy Tomato Frog *D. insularis* in Appendix II

Proponent: Madagascar

Summary: The False Tomato Frog *Dyscophus guineti* and the Antsouhy Tomato Frog *D. insularis* comprise two of three species in the genus *Dyscophus,* all of which are endemic to Madagascar. The third species, *D. antongilii* was included in Appendix I in 1987. It is subject to a separate proposal to be transferred from Appendix I to Appendix II (Proposal 37). All three are attractive red-orange coloured frogs.

Dyscophus are known to breed explosively with the availability of water during the rainy season (typically January-March) and during that time they can be found in abundance at breeding sites. Hundreds of eggs are laid in water following mating.

Dyscophus guineti
The known distribution of *D. guineti* includes a number of patches in the remnant central eastern rainforest of Madagascar. The species is secretive and believed likely to be more widespread than records indicate[1]. Overall population is unknown; locally the species can vary from extremely common to very rare[1]. Sexual maturity is attained between two and four years, comparatively earlier in males than in females[2].

The habitat of the species is affected by conversion of forest to agriculture, timber extraction, charcoal production and potentially small-scale mining activities. The species reportedly does not tolerate severe degredation[1]. There is not known to be local use of the species.

As a consequence of the Appendix-I listing in 1987 of the similar *Dyscophus antongilii*, collectors interested in "red *Dyscophus*" have shifted their attention to *D. guineti* which is now collected for export[3]. USA trade data indicate that the USA imported some 5300 live wild *D. guineti* from Madagascar in the period 2004 to 2013 with average annual imports higher at the start of this period than at the end (ca. 780 annually for 2004 to 2007; ca. 360 annually for 2008 to 2013)[5]. Madagascan export data indicate an increase in numbers exported and number of importing countries in recent years, from ca. 150 exported to three countries in 2013 to 2400 exported to 11 countries in in 2015. In recent years the USA has reported the export of significant numbers of captive-bred individuals[5].

There is a lack of evidence regarding the impact of harvest for trade on *D. guineti*. The species is reported mainly to be harvested in one area (around Fierenana). Populations there have been said to have been affected by harvest[3], although the view has also been expressed that levels of harvest, at least in 2008, were too low to have a serious impact on populations[1].

The species is classified in the IUCN Red List as Least Concern (2016).

Dyscophus insularis
This species has a wide distribution throughout western Madagascar, from Ambanja to south of Toliara. It has been suggested that this species could in fact be multiple species based on the wide range of habitats it occupies and the obvious geographic variation in morphology and colour[4]. No quantitative information on the population size in the wild could be found, but it is reported to be a common species[4] and is said to be quite locally abundant[3].

The species is likely to be affected by loss of habitat, although quantitative data are lacking.

According to data from Madagascar, exports of *D. insularis* have increased in recent years, as is the number of importing countries. In 2012, Madagascar did not record any exports, whereas in 2015 this had increased to 720 frogs exported to six countries. The USA reported importing 4,503 wild *D. insularis* from Madagascar between 2004 and 2013; exports were generally higher at the beginning of this period but have risen again somewhat in recent years[5]. As with *D. guineti,* the USA has been reporting the export of significant numbers of captive-bred individuals in recent years.

International trade levels are not thought to constitute a major factor affecting the species[4]. The species is classified in the IUCN Red List as Least Concern (2016).

Both species are protected under national legislation (Decree 2006-400) which allows harvest outside of protected areas with authorisation.

Both species, particularly *Dyscophus guineti*, resemble *D. antongili*[3, 6].

Analysis:

Dyscophus guineti

Dyscophus guineti has a wide distribution, and although the population size and trend are not known, it is reported to be at least locally abundant. Recorded levels of trade for the international pet market are relatively low, although have been reported as increasing recently. Individuals captive-bred outside the range State appear to meet at least some of the demand for the species. A possible decline has been reported from the main collection site but this is likely to represent only a small part of the range of the species. It seems unlikely that the species meets the criteria for inclusion in Appendix II in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)*.

Dyscophus insularis

Dyscophus insularis has a wide distribution and although the population size is not known, it is said to be common. As with *D. guineti* recorded levels of trade for the international pet market are relatively low, although have been reported as increasing recently. Individuals captive-bred outside the range State appear to meet at least some of the demand for the species. It seems unlikely that the species meets the criteria for inclusion in Appendix II in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)*.

Both species, particularly *D. guineti*, resemble *D. antongilii* which is already listed in the Appendices, and therefore both species meet the criteria for inclusion in Appendix II under Annex 2 b of *Res. Conf. 9.24 (Rev. CoP16)* (lookalike criteria). If this proposal and Proposal 37 (to transfer *D. antongilii* to Appendix II) were accepted it will have the effect of placing the entire genus in Appendix II.

Reviewers of summary information only: M. D. Kusrini and C. Ratsimbazafy.

References:

Information not referenced in the Summary section is from the Supporting Statement.

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[1] IUCN SSC Amphibian Specialist Group (2016) *Dyscophus guineti.* The IUCN Red List of Threatened Species 2016. View on 6th July 2016.

[2] Tessa, G., Guarino, F.M., Randrianirina, J.E. & Andreone, F. (2011) Age structure in the false tomato frog Dyscophus guineti from eastern Madagascar compared to the closely related *D. antongilii* (Anura, Microhylidae), African Journal of Herpetology, 60:84-88.

[3] Andreone, F. (2016) *In litt*. to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

[4] IUCN SSC Amphibian Specialist Group (2016) *Dyscophus insularis*. The IUCN Red List of Threatened Species 2016. Viewed on 6th July 2016.

[5] Analysis of US Fish & Wildlife Service Law Enforcement Management Information System (LEMIS) data, May 2016.

[6] Ratsimbazafy, C. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

Inclusion of Burrowing Frogs *Scaphiophryne marmorata, S. boribory* and *S. spinosa* in Appendix II

Proponent: Madagascar

Summary: *Scaphiophryne boribory*, *S. marmorata* and *S. spinosa*, known as burrowing frogs, are members of a genus endemic to Madagascar in which nine species in total are currently recognised. One species, *S. gottlebei,* was listed in Appendix II in 2003. All three now proposed are green-brown in colour with attractive patterning. All species of the genus are assumed to be explosive breeders that only reproduce once per rainy season after the first heavy rains[1]. They spend much of their time underground.

Because of their attractive colouration there is some demand in the international pet trade for these species. However, their burrowing habits likely limit the extent of this demand, their appeal being confined mainly to specialist hobbyists[2]. There is no known local use for any of the species.

Scaphiophryne boribory was described in 2003 from the Fierenana region of eastern Madagascar. Its distribution is more extensive than had previously been thought, the species currently also being known from Bemanevika forest and Marotondrano Special Reserve[3]. It has been reported to be locally common but is presumed to be in decline because of loss and degradation of habitat through conversion of land to agriculture and, locally, mining activities. The species was classified in the IUCN Red List as Endangered (2008), in part due to its limited known distribution at the time[4].

There is very limited known trade in the species, which is reported to be collected for export around Fierenana and Marotondrano. No imports into the USA are recorded in USA trade data for 2004 to 2013. Madagascan export data for the period 2014 to 2015 record export of 40 to Japan in 2015.

Scaphiophryne marmorata occurs in east central Madagascar from Zahamena south to the region of Andasibe. It has an extent of occurrence of around 15,000km^2. Population size is unknown. The species has been reported to be locally abundant, but the overall population is presumed to be declining as a result of loss and degradation of habitat. The species was classified in the IUCN Red List as Vulnerable (2016).

Reported trade is limited, though higher than that for the other two species in the proposal. The USA reported importing 2387 live *S. marmorata* from Madagascar between 2004 and 2013 all of which were wild. Annual imports at the beginning of this time period were higher (ca.740 between 2004 and 2005, compared with ca. 115 between 2006 and 2013). Madagascan export data indicate export of 245 in 2015 to five different countries.

Scaphiophryne spinosa has a wide distribution in eastern Madagascar from Masoala south to the Chaines Anosyennes in the far south[5]. There are no population estimates. The species can reportedly tolerate some habitat modification and it is not regarded as under threat at present. It was classified in the IUCN Red List as Least Concern (2016).

Very limited trade has been reported in this species. USA trade data for 2004 to 2013 indicate that the USA imported 41 live individuals from Madagascar in 2008. Madagascan export data for 2012 to 2015 records export of 180 to five countries in 2015.

Scaphiophryne spinosa was split from *S. marmorata* in 2002; the ranges of the two species overlap and they are reportedly still sometimes confused in trade[6]. All three species may also be included in exports of *Scaphiophryne* that are not reported to species level.

All three species are nationally protected (2006-400 Category I and Class II) meaning trade is legal if there is evidence that the specimens were harvested outside of protected areas, or a permit has been obtained for harvest within protected areas for scientific purposes[7].

Other *Scaphiophryne* species (e.g. *S. madagascariensis,* and *S. pustulosa*) have been reported in trade. It may be difficult for a non-expert to identify all the different species in the genus *Scaphiophryne*[8,7], although one expert noted it may be possible with guidance[6]. However, the three species subject to this proposal are relatively distinct from the other species in the genus, and therefore with guidance should be able to be identified from non-proposed *Scaphiophryne,* and *S. gottlebei*[9].

Analysis: *Scaphiophryne boribory, S. marmorata* and *S. spinosa* all have relatively wide distributions in eastern Madagascar. There is no information on overall population status of any, although all three have been described as at least locally common. *S. boribory* and *S. marmorata* are likely to be declining overall owing to habitat loss and degradation. All three have appeared in international trade, but only *S. marmorata* in any quantity; even for this species overall trade levels are relatively low. Demand for these species on the international market is likely to be limited. It seems unlikely that any of them meets the criteria for inclusion in Appendix II in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)*. However, given the difficulties in distinguishing between these species, were one of them to be listed in Appendix II, then the others would meet the criteria in Annex 2b A (lookalike criteria).

The Appendix-II listed *Scaphiophryne gottlebei* was included in the Review of Significant Trade process in 2008. The CITES Animals Committee subsequently decided that Madagascar was implementing the requirements of Article IV of the Convention for this species. It appears that none of the species proposed here meet the criteria for inclusion in Annex 2 b of *Res. Conf. 9.24 (Rev. CoP16).*

Reviewers of summary information only: M. D. Kusrini and C. Ratsimbazafy.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[5] IUCN SSC Amphibian Specialist Group (2016) *Scaphiophryne spinosa.* The IUCN Red List of Threatened Species 2016.
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Inclusion of Titicaca Water Frog *Telmatobius culeus* in Appendix I

Proponents: Plurinational State of Bolivia and Peru

Summary: The Titicaca Water Frog *Telmatobius culeus* is a totally aquatic frog occurring in Plurinational State of Bolivia (Bolivia) and Peru where its range is limited to Lake Titicaca (ca. 8500km^2 divided into the Minor Lake (2100km^2) and the Major Lake (6450km^2)), which straddles the Bolivian/Peruvian border, and a few nearby water bodies in Peru within an overall area of ca. 17,500km^2. It is sometimes referred to as the Titicaca Scrotum Water Frog on account of its loose skin. It is most frequently recorded in shallow water but has been observed at 100m depth[1]. Generation time is taken as five years.

Estimating population size is difficult because of the size of Lake Titicaca and the difficulty of making observations in deeper parts of the lake. Two estimates from 2002 differed very significantly: one gave the population as ranging from 17 million (+/- 14 million) in the dry season to 51 million (+/-34 million) in the wet season; another estimated the population at 2.5 million, assuming presence to depth of 40m. Local estimates of population density have also varied greatly. One estimate in a sampled area in one part of Lake Titicaca calculated 0.57 adults and 1.63 frogs per m^2 [2]. Most frogs were encountered at two to three metres depth, but the study did not assess occurrence below five metres. Another study[3] found densities on the Bolivian side of the lake of 1.14 individuals per 100 m^2 in the Minor Lake and 2.05 in 100 m^2 in the Major Lake. Given these were made on the basis of short transects close to the shore and observations to two metre depth it does not seem reasonable to extrapolate numbers to the whole lake.

One 2008 report estimated a decline of ca. 40% in the population between 1999 and 2008 in the Bolivian part of the Minor Lake[4]. In areas monitored by the Bolivian Amphibian Initiative/Alcide d'Orbigny Natural History Museum project, the observed population has decreased by 70% from previous years in some locations.

Mass mortality events associated with algal blooms as a result of organic pollution occurred in the Minor Lake area in 2009, 2011 and 2015. It was concluded that *T. culeus* was no longer present over an area of 500km^2 following one such event in April and May 2015[5]. “Chytrid fungus” and ranavirus have been detected in the population, but there is no information on their impact. Virtually all information on potential impacts on the species comes from the Minor Lake. The largest city in the lake’s catchment (El Alto, Bolivia) is in this part of the Lake, which is believed to be most affected by anthropogenic influences. The majority of the lake’s water, notably in the Major Lake, is still clean.[6]

Despite harvest being prohibited, the species is known to be used nationally in both Bolivia and Peru; it is often made into a soup or blended into a juice believed to have medicinal and aphrodisiac properties. Frogs legs are served as an exotic dish, mainly for tourists. It appears that the bulk of harvest (estimated at around 55,000 a year) is for local consumption although there have been reports that harvest on the Bolivian side of the lake is occurring for use in the markets in Lima, Peru[4, 7].

Evidence for international trade other than the possible transboundary trade mentioned above is extremely limited. There have reportedly been two instances, in 2009 and 2016, of very small numbers of live individuals intercepted in Ecuador, reportedly en route to Europe. In Germany, one of the major countries in Europe where amphibians are kept, it is not known to be held in any private collections, nor is it held in any European zoo[8, 9]**.** Evidence for export of the species for food in inconclusive. Data from the UN Comtrade Database reported one record of an import of 33,700kg of frogs’ legs (HS code 020802) from Peru into France in 2002 but it is not possible to determine the species involved; and there is no indication that it was *T. culeus*.

This species is classified in the IUCN Red List as Critically Endangered (2004; needs updating).

Analysis: *Telmatobius culeus* does not have a restricted range. Estimates of its overall population vary widely, but the species clearly does not have a small population. There are indications of declines, some marked, in parts of Lake Titicaca, the lake which comprises the great majority of its range. All such information relates to the Minor Lake, which comprises around one quarter of the area of Lake Titicaca. There is no information on population trends in the Major Lake or the other water bodies where the species occurs, although anthropogenic impacts are lower in the Major Lake than in the Minor Lake. There are indications of some transboundary trade between Bolivia and Peru, but indication of any other international trade is extremely limited. There is, overall, insufficient information to determine whether the species meets the criteria for inclusion in Appendix I.

Reviewers of summary information only: R. Melisch.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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Inclusion of the Hong Kong Warty Newt *Paramesotriton hongkongensis* in Appendix II

Proponent: China

Summary: *Paramesotriton hongkongensis,* the Hong Kong Warty Newt, is a relatively large, stocky newt endemic to China, where it is found in Hong Kong (Special Administrative Region) and the coastal Guangdong Province of mainland China. Its total area of distribution is estimated at approximately 20,000km[2]. It is primarily terrestrial, spending on average 45 to 60 days per year in streams and the rest in terrestrial habitat[1]. It has high critical requirements for water quality. Breeding is seasonal, females producing ca. 120 eggs per year. Maturity is reached in three to five years.

Available information suggests that the species may be relatively abundant in at least parts of its range. Monitoring at seven breeding pools in Hong Kong (SAR) between 2007 and 2014 indicated the population in these was generally stable. Much of the population in Hong Kong (SAR) is within protected areas and it is considered that these populations are generally stable[1]. Populations in Hong Kong (SAR) are believed likely to be larger than those in mainland China[2].

The species is reported to be affected by habitat alteration, stream channelization and water pollution. It has also been collected for both domestic use and export, in both cases as a pet and in research institutions.

Paramesotriton species were frequently recorded in a pet market survey in Guangdong Province in 2006 to 2008 and there were reports of large numbers appearing at interior pet markets in large cities in mainland China in the early 1990s. However, it is likely that these reports represent other as then-undescribed species of *Paramesotriton*[1].

Trade data indicate that an average of around 40,000 *P. hongkongensis* per year were imported into the USA between 2004 and 2013[3]. Imports increased from 2006 to 2010, before dropping considerably in 2013. Import of all Asian Caudata (newts and salamanders) into the USA has now been suspended because of concerns over disease[4, 5]. There are no European Union trade records of this species despite it having been listed in Annex D of the EU Wildlife Trade Regulations since 2009.

It is likely that the species has been confused with other *Paramesotriton* species or with species of *Cynops, Hyselotriton* or *Pachytriton* in trade. It has been imported into the USA under the generic names *Paramesotriton, Triturus* and *Trituroides*.

The species has been included in legislation in Hong Kong (SAR) since 1997 and in China since 2000; collection in both requires approval from competent departments and is not permitted in protected areas.

An increasing proportion of the trade into the USA has been reported as in captive-bred individuals[3]; however large scale captive-breeding for commercial purposes in Hong Kong (SAR) is not known and is believed unlikely to be economically viable as the species is of relatively low value[2]. The species has been successfully bred in captivity by hobbyists in Hong Kong (SAR)[2], in Europe and the USA.

Whilst it does appear possible to distinguish *P. hongkongensis* from other similar species based on morphological characteristics, non-professional identification may be difficult.

This species was classified in the IUCN Red List as Near Threatened (2004 – in need of updating).

Analysis: *Paramesotriton hongkongensis* has a reasonably large area of distribution, within which much of the population is reported to be in protected areas and apparently stable, at least in Hong Kong (SAR). It has been collected in the past in some numbers for domestic use and export as a pet and laboratory animal; the impact of such collection is unclear. The only country known to have imported significant numbers is the USA, which has currently suspended import of all Asian Caudata, therefore it seems unlikely that the species meets the criteria for inclusion in Appendix II.

Reviewers of summary information only: A. Lau, S. Chng, M. Lau and J. Janssen.

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Inclusion of Silky Shark *Carcharhinus falciformis* in Appendix II

Proponents: Bahamas, Bangladesh, Benin, Brazil, Burkina Faso, the Comoros, the Dominican Republic, Egypt, the European Union, Fiji, Gabon, Ghana, Guinea, Guinea-Bissau, Maldives, Mauritania, Palau, Panama, Samoa, Senegal, Sri Lanka and Ukraine

Summary: The Silky Shark *Carcharhinus falciformis* has a global distribution in oceanic and coastal tropical waters. It occurs in territorial waters of over 110 range States and in international waters.

Silky Sharks are highly migratory. They are often associated with seamounts; juveniles often congregate around floating objects. They can be long lived, believed normally up to 20 years[1] but sometimes longer, exhibit slow growth, late maturity (seven to 15 years for females), and production of few young (four to 18 pups[2] after a nine to 12 month gestation, with at least one resting year between litters). Their productivity is generally assessed as low. Overall population size is unknown and nearly all estimates of changes in population size are derived from fisheries data. Interpretation of such data is difficult, as landings are rarely reported at individual species level, there is a general lack of information on sizes, weights and numbers of individuals caught, and changes in management and reporting make analyses of time series data, particularly those covering long periods, challenging. Where declines are observed these are ascribed to fisheries-induced mortality.

The Silky Shark is taken in very large numbers mainly as incidental catch from longline tuna fisheries but also in purse seine fisheries and in some targeted shark fisheries. There are not known to be any major unexploited populations.

Catch and landings of Silky Shark are believed to be underreported. According to FAO data, Iran, Sri Lanka, Taiwan (Province of China), Ecuador and Costa Rica have been the main harvesters of Silky Sharks in recent years with total catch reported by those at around 7500t in 2010 declining to just under 5000t in 2014.

Silky Sharks may be used for meat, particularly in Oman and Taiwan (POC), and to a lesser extent for skin, liver oil, cartilage and teeth. The principal part in trade is the fin, in demand in East Asia, particularly China. An assessment based on 2000 data estimated that at that time a minimum of 500,000 to 1.5 million Silky Sharks were used annually for their fins; this being an estimated 5% or so of shark fins in trade at that time[3]. An assessment in 2014 using different methods found the species to be the second most important (by weight) in the world's largest shark fin market in Hong Kong (Special Administrative Region), accounting for ca. 5% of the total weight in the sample; absolute amounts in trade were not assessed[4].

Numerous declines in catches and in stocks, some widespread, have been reported. In the western and central Pacific, according to one estimate made in 2013, stocks had declined to 30% of theoretical equilibrium unexploited biomass; exploitation at that time was estimated at 4.5 times greater than a sustainable level. Recent analysis of Catch per Unit Effort (CPUE) data for the region found high fluctuations from which it was not possible to determine a trend[5]. Analysis of the most recent available information from the Eastern Pacific indicates a decline in CPUE of 77% in the southern stock based on comparison of 1994 to 1996 data with that for 2004 to 2013; data for 2014 to 2015 showed a slight increase in CPUE. Data for the northern stock indicated an overall 37% decline in catch rates in floating object sets for the period 1994 to 2015[5].

Data from the Indian Ocean are sparse. The reported annual catch in Sri Lanka declined from an average of ca. 20,000t in 1997 to 2000, to below 5000t from 2005 onwards, and ca. 3500t in 2012 to 2014. Changes in fishing effort are not reported. Fishers in the Maldives report declines of 50 to 90% in landings of the species over the past 20 years.

In the North Atlantic, one study in 2007 found a 50% decline in CPUE in longline fisheries between 1986 and 2005; the same study reported a 46% decline between 1992 and 2005 in longline fisheries based on observer data. Analysis of catches in the Gulf of Mexico from the 1950s to the 2000s shows a decrease in average size of Silky Sharks landed from ca. 100kg to 23kg. Declines in mean size and increasing proportion of juveniles have also been reported in Costa Rica and the southeast USA.

Total fishery-induced shark mortality caught in Indian Ocean purse seines was ca. 80% in 2011 to 2012, with about half of live discards from purse seines suffering delayed mortality. Pelagic longline fisheries off the

southeast coast of the USA reported 26% of Silky Sharks caught were released alive (with 44% discarded dead and 30% retained), although post release survival is not known.

Silky Sharks are protected under national legislation in over 10 countries and shark finning bans are implemented by 21 countries, the European Union (EU), and nine Regional Fisheries Management Organizations (RFMOs), which could help reduce Silky Shark mortality, if they cause a larger proportion of the catch to be released alive. Silky Sharks are listed in Annex I, Highly Migratory Species, of the UN Convention on the Law of the Sea. In 2014, the Convention on the Conservation of Migratory Species (CMS) listed the Silky Shark on Appendix II and in 2016, it was added to the Migratory Shark Memorandum of Understanding.

Fisheries management for this species on the high seas falls under the remit of the tuna RFMOs. The International Commission for the Conservation of Atlantic Tuna (ICCAT) and the Western and Central Pacific Fisheries Commission (WCPFC) prohibit retaining on board, trans-shipping, or landing any part or whole carcass of Silky Shark in the fisheries covered by these Conventions. However, there are concerns that there is little or no compliance monitoring of these measures in place. The Indian Ocean Tuna Commission (IOTC) recognises the depleting stock status of Silky Sharks in the Indian Ocean, however it has not adopted a management measure to date.

The species was classified by IUCN as Near Threatened in 2009.

In the shark fin trade, Silky Sharks are labelled as Wu Yang. A 2006 study found that 80% of samples labelled Wu Yang came from Silky Sharks, the remainder were from a variety of other species, including some that could not be identified[6].

Analysis: The Silky Shark is a low productivity species with a global distribution in coastal and oceanic water. It is widely caught, chiefly as incidental take in longline tuna fisheries. Retention of catch is chiefly to supply the trade in shark fins, particularly in East Asia. There is evidence of declines, some marked, in much of the range. Such declines are attributed to overharvest. Information is sparse for the Indian Ocean, although there are localised reports of declines in catches here. Longline tuna fisheries are widespread in the Indian Ocean and there is no reason to believe that these do not have a similar impact on the Silky Shark population here as observed elsewhere in its range. There are not known to be any major unexploited populations. It would appear therefore that the Silky Shark meets the criteria for inclusion in Appendix II in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16),* in that regulation of harvest for trade is required to ensure that the species is not reducing the population to a level at which it becomes threatened.

Reviewers of summary information only: V. Mundy, O. Sosa-Nishizaki, S. Clarke, A. Harry and G. Sant.

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Inclusion of all species of Thresher Sharks in the Genus *Alopias* in Appendix II

Proponents: Bahamas, Bangladesh, Benin, Brazil, Burkina Faso, Comoros, Dominican Republic, Egypt, European Union, Fiji, Gabon, Ghana, Guinea, Guinea-Bissau, Kenya, Maldives, Mauritania, Palau, Panama, Samoa, Senegal, Seychelles, Sri Lanka and Ukraine

Summary: Species in the genus *Alopias,* known as Thresher Sharks, are migratory sharks occurring extremely widely in tropical and temperate oceanic and coastal seas. They are characterised by very long upper lobes of their caudal (tail) fins. There are three species: the Bigeye Thresher *Alopias superciliosus,* Common Thresher *A. vulpinus* and Pelagic Thresher *A. pelagicus.* They occur in surface temperatures of 16 to 25°C, but have been tracked as deep as 723m where temperatures are around 5°C[1]. They have overlapping distributions. Common Threshers are recorded circumglobally and have a noted tolerance for cold waters; highest concentrations tend to occur in coastal temperate waters. The Bigeye Thresher is also circumglobal but is generally found at low latitudes and, in the Pacific at least, in pelagic rather than onshore areas. The Pelagic Thresher is the least known of the three species. It occurs widely in the Indo-Pacific but is not known in the Atlantic[4].

The **Bigeye Thresher** reaches a maximum length of ca. 4.6m. Age at maturity is estimated at eight to 15 years for females and seven to 13 years for males. Lifespan is estimated at 20 to 21 years. Litters are small (two). Gestation is estimated at 12 months[2]. The species has extremely low productivity. The **Common Thresher** is the largest species, reaching nearly six metres in total length (including the tail). Historical and recent unconfirmed records indicate even greater total length. A recent study indicates longevity may be greater than previously thought, reaching at least 38 years[3]. Average age at maturity is estimated at five years. Litter sizes are typically small (two to seven) and may vary geographically[3]. Gestation is around nine months. The **Pelagic Thresher** may reach nearly four metres in length. Maximum reported age is 29 years. Maturity is reached at 2.6-2.9m in length; litter size is two[4]. The species is reported to have lower productivity than the Common Thresher.[8]

Overall population size of any of the species is unknown and nearly all estimates of changes in population size are derived from fisheries data. Interpretation of such data is difficult, as landings are rarely reported at individual species level, there is a general lack of information on sizes, weights and numbers of individuals caught, and changes in management and reporting make analyses of time series data, particularly those covering long periods, challenging. Where declines are observed these are ascribed to fisheries-induced mortality. Bigeye and Common Thresher have recently been the subject of a detailed review undertaken by the US National Marine Fisheries Service (US NMFS) of the National Oceanic and Atmospheric Service[3]. This contains information and analysis that in many cases supersedes information, particularly on estimated declines, that is included in the Supporting Statement of the proposal. Much of the information in this analysis is drawn from the US NMFS review.

Thresher sharks are taken as incidental and target species in many coastal and oceanic pelagic fisheries where they occur. They are primarily caught in longline fisheries, but are also caught with anchored bottom and surface gill nets, and accidently caught in bottom trawls and fish traps. Directed fisheries and retention of incidental catch are driven by demand for meat (in the case of Common and Pelagic Threshers) and fins. Meat is generally consumed locally (although any catch from international waters would be considered trade, as introduction from the sea, under CITES) while fins enter international trade, being chiefly destined for East Asia, particularly China. Bigeye Thresher meat is reported not to be widely eaten[3].

Thresher shark catches in general are considered underreported globally[3]. FAO catch data indicate harvest of 183,000t of threshers between 1999 and 2014. Total reported catches increased greatly from 3400mt in 2004 to ca. 12,000mt to 2005 (most likely due to changes in reporting practices[5]), and peaked in 2011 at ca. 22,000mt decreasing marginally to ca. 19,000mt in 2014. Indonesia, Ecuador, Sri Lanka and the USA have reported the highest level of catch. Almost all (ca. 85%) catch data are reported as *Alopias* spp although some have been reported at the species level: ca. 3000mt as Bigeye Thresher, ca. 6000mt as Common Threshers and ca. 20,000mt as Pelagic Threshers, this last all reported by Ecuador for the Southeast Pacific[6].

The majority of reported catch has been in the Pacific (68%) followed by the Indian Ocean (29%); reported catches in the Atlantic, Mediterranean and Black Sea are negligible in comparison.

Information for the **Indian Ocean** is patchy. Data from the Indian Ocean Tuna Commission (IOTC) database record an increase in catch in **Common Thresher** in the late 1990s, reaching a peak of just under 1000mt in 1999 and then declining. Reported catches of **Bigeye Thresher** have increased recently from negligible levels to ca. 200mt in 2002. Most catch is reported in thresher shark complex; this has risen from ca. 1000mt in 1990 to ca. 5000mt in 2012[3]. Catch is believed to be very underreported in the region, and it is not possible to derive reliable Catch Per Unit Effort (CPUE) estimates from it. One 2013 study estimated that actual thresher catch in the **Indian Ocean** might be of the order of 25,000mt per annum[7].

In the northern part of the **Eastern Pacific** most information relates to **Common Threshers**. The stock of this species along the western coast of North America is believed to have undergone a marked decline beginning in the late 1970s as a result of fishing mortality. Management measures have led to an improvement in the status of this stock, which has recovered to levels near those estimated for the early 1970s[3].

In the southern part of the **Eastern Pacific**, reported catch from Ecuador is of **Pelagic Threshers** (see above). Catch in the largest shark fishery (Peru) also appears to comprise very largely of Pelagic Threshers. Shark landings in Peru have declined by roughly 3% per year for 2000 to 2010 despite an increase in the size of the fishery but trends in catch in threshers are unclear[3]. **Bigeye Thresher** are known to be taken as incidental catch in purse-seine and longline fisheries in the **Eastern Pacific** but generally comprise a small proportion of overall shark catch. There is no information on CPUE trends for the species in this fishery[3].

In the **Western and Central Pacific** there is generally a paucity of species-specific information, although the **Bigeye Thresher** is believed to be the predominant thresher species in offshore areas here. Species-specific observer information indicates that the **Bigeye Thresher** may be stable or possibly increasing in the area in which the Hawaiian longline fishery operates. A 2015 analysis of standardised CPUE in the wider **Western and Central Pacific** (which did not include data from the Hawaiian fishery) using aggregated data for all three species indicated a relatively shallow decline between 2003 and 2011 and a much steeper decline from 2011 to 2014, although information for 2014 was based on few data points[3]. **Pelagic Thresher** are caught in Taiwan POC longline fisheries. An analysis of spawner per recruit for the species in eastern Taiwanese waters for 1990-2004 suggested that the stock was slightly overexploited at that time.[8]

In the **South Atlantic** most thresher catch is of the **Bigeye Thresher**. CPUE of the species in the Uruguayan longline fishery is low, although data available only cover a short time period from which it is not possible to discern trends. In the Brazilian longline fishery, slight declines in CPUE were observed up to 2006, at which point the species disappeared from the fishery, although this is believed likely a result of the fishery moving to more temperate latitudes not favoured by the species rather than reflecting actual population changes[3].

In the **Northwest and Central Atlantic** abundance trend estimates derived from standardized catch rate indices of the USA pelagic longline fishery suggest that both **Common** and **Bigeye Threshers** have likely undergone historical declines in abundance. Standardised abundance indices derived from observer data indicate that populations of both these species may have stabilised since 1990[3].

In the **Mediterranean**, **Bigeye Thresher** are considered scarce. Very severe declines in stocks of **Common Thresher** here (perhaps as much as 99%) have been documented, ascribed to fishing mortality[3].

The proportion of threshers landed live globally is not known. One study in the Pacific Islands Countries and Territories in the Western and Central Pacific found that roughly half of landed **Bigeye Threshers** landed were dead or judged unlikely to survive after release.

The quantity of thresher shark fins identified in Hong Kong (Special Administrative Region) fin markets in the early 2000s equated to between 350,000 and 3.9 million individual thresher sharks, or a biomass of 12,000 to 85,000mt being killed and traded per year. This comprised roughly 2.3% of the estimated global shark fin trade. Much of this trade goes through Hong Kong (SAR), where thresher shark fins are traded as “wu gu” ; the majority of fins in this category are from threshers although some mixing with Longfin Mako *Isurus paucus* has been documented[9].

In a 2014 study, threshers made up a very small proportion (0.1%) of shark fin samples analysed[10]. Although there has been a reported decline in shark fin trade and consumption generally in recent years[11] debate remains regarding the causes, which may include increased regulation of catches, declining stocks and catch per unit effort or falling consumer demand.

All three species of Thresher Shark were classified in the IUCN Red List as Vulnerable (2009). Regional assessments for the Mediterranean for the Bigeye Thresher and Common Thresher classified them as Endangered in 2016.

The proponents indicate that the proposal concerns the inclusion in Appendix II of Bigeye Thresher Shark as satisfying the criteria in Annex 2a of Resolution Conf. 9.24 (Rev. CoP16); and the inclusion of all other species of thresher sharks as satisfying the criteria in Annex 2b of the Resolution.

Analysis: The three Thresher Sharks in the genus *Alopias* are widespread oceanic species that are harvested in large numbers, particularly as incidental catch in longline fisheries. Their fins enter the international fin trade. There are no overall population estimates for any of the species. Much fisheries information is recorded only to genus level, making determination of species-specific trends particularly problematic. Where population declines have been identified, these are invariably ascribed to fishing pressure.

The **Bigeye Thresher** has extremely low productivity. There are indications of historic declines in the Northwest Atlantic, where populations may have stabilised at a low level. Reported catch rates in the South Atlantic are low. In the West and Central Pacific, where the species occurs widely, there are indications from 2003 onward of decline in threshers in general, which may be accelerating; however, information from one extensive fishery (the Hawaiian longline fishery) indicated stability of the Bigeye Thresher population in the region covered by the fishery. Reported catch of threshers in the Indian Ocean has increased and it is believed that unreported catch (of all threshers) may be many times that of reported catch but there are no stock assessments or analyses of changes in CPUE.

The **Common Thresher** has low productivity. There are indications of extremely marked declines in the Mediterranean that are believed to be of this species, and of historic declines in the Northwest Atlantic, where populations may have stabilised at a low level. In the Northeast Pacific, Common Threshers underwent a decline in the 1980s and 1990s but populations appear to have recovered here because of improved management.

The **Pelagic Thresher** has very low productivity. It is known to be taken in large numbers in the Eastern Pacific and to be harvested in the Indian Ocean and West and Central Pacific but there is very little species-specific information on stocks or changes in CPUE.

Given known intensity of fishing pressure in much of the range of all three species and their low productivity (particularly the Bigeye Thresher) it is likely fisheries in a number of areas are unsustainable. In others, thresher stocks may be relatively stable, but in at least some of these populations are very likely to be at significantly lower than historic levels. Overall it is unclear for any of the species whether this level of decline would satisfy the criteria for inclusion in Appendix II in Annex 2 a *Res. Conf. 9.24 (Rev. CoP16*).

If any of the species were listed in Appendix II, the others in the genus would meet the criteria in Annex 2 b (lookalike criteria).

Reviewers of summary information only: S. Clarke, G. Sant, T. Curtis and R. Jabado.

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Inclusion of all species of Devil Rays in the Genus *Mobula* in Appendix II

Proponents: Bahamas, Bangladesh, Benin, Brazil, Burkina Faso, Comoros, Costa Rica, Ecuador, Egypt, European Union, Fiji, Ghana, Guinea, Guinea-Bissau, Maldives, Mauritania, Palau, Panama, Samoa, Senegal, Seychelles, Sri Lanka and United States of America

Summary: The genus *Mobula*, the Mobulas or Devil Rays, includes nine described species of marine ray that grow to wingspan or disc widths (DW) of one to over five metres. The genus is widely distributed in temperate and tropical seas. Different species may be confused with each other and precise distributions in most cases are poorly known. All are believed to have very low productivity. Females give birth to a single live young following a gestation period of around one year and are thought often to undergo protracted periods, perhaps two or three years, between pregnancies. All species are believed to be largely epipelagic (i.e. occur mainly in surface waters) although some also dive deep[1]. The three largest species are *Mobula japanica, M. mobular* and *M. tarapacana*; of these, *M. japanica* and *M. tarapacana* are the most widespread and feature most prominently in fisheries.

Mobula japanica is circumglobal in temperate and tropical seas, but its distribution is not completely defined. The species is usually encountered as solitary individuals but has also been recorded travelling in schools and tagged individuals have been monitored travelling long distances. In parts of the range populations appear to make regular migratory movements and form seasonal aggregations. Elsewhere the species is recorded year-round. *M. japanica* grows to a maximum of 310cm DW, and age at maturity has been estimated at five to six years (ca. 210cm DW). Generation time has been estimated at approximately 10 years. Pupping may take place offshore, around seamounts or islands. Population size and structure are unknown. The species was classified by IUCN as Near Threatened in 2006.

Mobula tarapacana has a circumglobal range in temperate, subtropical and tropical waters of the Indian, Pacific and Atlantic Oceans, however is patchily distributed within this range. It is primarily oceanic but is also found in coastal waters[1]. As with *M. japanica,* individuals are known to migrate long-distances and at least some populations make seasonal movements and form regular aggregations. Maximum verified size is 314cm DW. Estimated age of maturity is five to six years, minimum life span is 14 years, and generation length has been inferred at approximately 10 year[1]. Population size and structure are unknown. The species was classified as Vulnerable by IUCN in 2016.

Mobula mobular is confined to the Mediterranean and possibly adjacent North Atlantic, although it is believed that records here may be of *M. japanica* with which it is easily confused. It appears to occur at low densities, generally as solitary individuals or in groups of two to four, although larger seasonal aggregations are known. The species may reach over 5m DW and generation time has been estimated at 20 years. Recent aerial surveys estimated a population in the south central Adriatic of ca. 1600 individuals (coefficient of variation, CV, 25%), and of >12,700 individuals in the north western Mediterranean (CV 53%)[2]. The species was classified as Endangered by IUCN in 2015, and a Mediterranean regional assessment classified this population as Endangered in 2016.

Mobula thurstoni is probably circumglobal in all temperate and tropical seas, but its distribution is not completely defined. It is usually found in pelagic, but shallow waters (<100m), reaches a maximum DW of 180cm and maturity at 150cm DW. The species was classified as Near Threatened in 2016 by IUCN.

Mobula eregoodootenkee is locally common within its wide tropical Indo-west Pacific and northern Indian Ocean distribution, with a DW up to 100cm[3]. The species was classified in 2003 as Near Threatened.

Mobula hypostoma is endemic to the western Atlantic, found from North Carolina (USA) to northern Argentina, including the Gulf of Mexico, and Greater and Lesser Antilles. It has a DW up to 120cm and occurs primarily in coastal waters, although it occasionally enters oceanic waters[4]. The species was classified as Data Deficient in 2009 by IUCN.

Mobula kuhlii is an uncommon, inshore ray with a patchy distribution in the Indian Ocean and western central Pacific, reaching 120cm DW. Of 409 mobulid rays observed at four landing sites in eastern Indonesia from April 2001 to October 2005, *M. kuhlii* was the most rarely recorded and composed only 2% of the total rays in this part of its range[5]. The species was classified as Data Deficient by IUCN in 2009.

Mobula munkiana is an inshore species occurring in the Eastern Pacific from the Gulf of California, Mexico to Peru. It reaches 110cm DW; females mature at 97cm DW and males at ~87cm DW. It is known to form large aggregations[6]. The species was assessed as Near Threatened in 2006 by IUCN.

Mobula rochebrunei has a status that is still uncertain[7], although from currently available information is believed to be found in the eastern Atlantic from Mauritania to Angola and from two possible records off Brazil in the Southwest Atlantic, where it is probably rare. It is known to form large aggregations; maximum size is 133cm DW[8]. IUCN classified *M. rochebrunei* as Vulnerable in 2009.

Mobula species are closely related and similar to, though generally smaller than, Manta Rays *Manta* species which were included in CITES Appendix II at CoP16 (2013). Collectively the two genera are often referred to as mobulids. Most catch and trade data do not distinguish the two.

Mobula species may be affected by various factors such as climate change, pollution and ingestion of marine debris but by far the most important impact on populations is believed to come from targeted and incidental catch in both artisanal and large-scale fisheries. Studies of the small-scale artisanal Mexican fishery (which ceased in 2007) concluded that fishing rates were twice as high as their estimate of maximum intrinsic rate of population increase[9].

The meat is generally not highly sought after although some artisanal fisheries do target *Mobula* species for food and local products. In the past individuals caught incidentally were often discarded or released. The principal driver for directed fisheries and retention of incidental catch is the international market for gill plates, demand for which has increased greatly in recent years in Asia. This has led to *Mobula* species being increasingly targeted and retained.

A 2015 review identified 13 fisheries (mostly artisanal) in 12 countries that specifically targeted *Mobula* species and 30 fisheries in 23 countries that incidentally caught them. They were reported as incidental catch in nine large-scale fisheries in 11 countries using driftnets, trawls and purse seines, and in 21 small-scale fisheries in 15 countries using driftnets, gillnets, traps, trawls and longlines[10]. Five countries - Sri Lanka, India, Peru, Indonesia and China (the latter fishing in international waters) – are between them believed to account for the great majority, perhaps as much as 95%, of worldwide *Mobula* catch. There are examples of evolving/new *Mobula* fisheries in response to demands for gill plates for East Asian markets such as a new mechanized gillnet fishery formed in India in 2005 and offshore gillnet fishing in Myanmar, which started in 2014. Individual fisheries generally catch more than one *Mobula* species, complicating analysis. There is a single-species targeted fishery for *M. mobular* in the eastern Mediterranean - these are used for meat; gill plate export from the region is not confirmed[11].

FAO catch data do not distinguish between catch of *Manta* and *Mobula*, and are apparently incomplete. FAO fisheries data reported in the “manta/devil ray” category, is restricted mainly to Indonesia (24,059t reported between 2004 and 2013) and Liberia (3651t between 1998 and 2006, with no reported landings since then). Other countries report very small quantities in this specific category, but they may be reporting large quantities under “rays, stingrays, manta nei (not elsewhere included)”. There is, however no way of establishing the proportion of *Mobula* species within these more general categories. Total reported catches of “manta/devil rays” increased from 342t in 1998 to 931t in 2000, decreasing to around 100t per year between 2001 and 2003, and increasing again to over 4000t in 2008 and 6000t in 2013.

Mobula japanica and *M. tarapacana* are both known to be targeted and landed in Indonesia, Malaysia, Sri Lanka and the Philippines, and previously Mexico. *M. japanica* is also landed in China, Taiwan (Province of China), India, Myanmar and Oman and *M. tarapacana* in Senegal. *M. thurstoni* is known to be landed in Indonesia, Philippines, India, Sri Lanka, Malaysia, Myanmar, Guatemala, Peru and Guinea and likely elsewhere across its range. *M. eregoodootenkee* is known to be fished in Philippines, India and marketed in Thailand and probably elsewhere in Southeast Asia. In 2009, *M. hypostoma* was known to be caught in longline, net and possibly other fisheries but not landed for international trade. Given trends in other species, it is conceivable that some catch is now retained for trade. *M. kuhlii* is taken in fisheries in Indonesia, in small scale fisheries in Mozambique, Tanzania, South Africa, the Arabian/Persian Gulf[12] and Gulf of Oman and likely throughout much of its range. *M. munkiana* was the dominant mobulid landed round Bahía la Ventana, Baja California Sur, Mexico in 2001 and is known to be landed in Peru. *M. rochebrune*i is said to be the predominant species in mobulid catches in Guinean catches at three survey sites.

Population trend information for all *Mobula* species is restricted to population declines inferred from landing data/observations, market surveys and community interviews from a few specific fishing areas, and gill plate markets in East Asia. At Cocos Island (eastern Pacific, Costa Rica), a 78% decline in *Mobula* species was estimated from diver surveys (who reported that *M. tarapacana* is generally the species sighted in the area) over 21 years[13]. In Sri Lanka, the overall decline in catch landings of *Mobula* species was 51% over three years[14]. Despite an increase in fishing effort, recent declines in *M. japanica* and *M. tarapacana* landings between 2001/2002 and 2014 of between 50 and 99% have been reported in three different regions of Indonesia. Reported landings of *Mobula* species in Guinea declined by 60% between 2004 and 2008 despite reported increase in fishing effort; landings of *Mobula* species at Tumbes, Peru, declined by 90% between 1999 and 2013; landings would have included both *M. japanica* and *M. thurstoni* and to a lesser extent *M. tarapacana*. The Inter-American Tropical Tuna Commission (IATTC) catch and bycatch data for *Mobula* from purse seine fisheries in the eastern Pacific between 1998 and 2009 show a significant increase from <1t in 1998 to >80t in 2006, and a subsequent decline over three years until 2009, when the reported catch was 40t[15].

Mobulid gill plates, used medicinally, are commonly sold under the trade names "Peng Yu Sai" (translated as "Fish Gills"). Three types of gill plates have been identified; 1) *Manta* species 2) *M. tarapacana* specifically referred to as "Flower Gills" (white or bi-coloured gills) and 3) smaller gill plates of *M. japanica* (black gill plates), *M. thurstoni* and possibly other mobula species.

Based on gill plate market surveys, the total estimated global market for Mobulids tripled between 2011 and 2013, from ca. 48,000 individuals to ca. 130,000 individuals. Of this around 4500 per year were *Manta* species and the remainder *Mobula*. In 2013 the global mobulid market was estimated to comprise 110,000 (83%) *M. japanica* and other 'black gill' *Mobula* species, 17,000 (13%) *M. tarapacana*, and 5,000 (4%) *Manta* species.

Plates of *M. tarapacana* and *M. japanica* are said to be the most important *Mobula* products in international trade, with the largest plates selling for a few hundred USD per kilo[16]. It can be difficult to distinguish visually between the dried gill plates of small *Manta* and large *M. japanica*, and dried gill plates from *M. japanica* are very similar in size and appearance to *M. thurstoni,* and *M. kuhlii*. Gill plates of *M. tarapacana* are bi-coloured and resemble those of some *M. thurstoni* and *M. hypostoma*.

Several countries prohibit harvest of all *Mobula* species, but globally there is little or no protection for most coastal and high seas habitats. There have been no stock assessments, monitoring, or management of *Mobula* fisheries in the range States with the largest fisheries.

All species of *Mobula* were recently listed in Appendices I and II of the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and in July 2015 the IATTC passed a resolution to prohibit retention, unless accidentally captured on purse seine vessels, and mandate safe release of all *Mobula* species in the RFMO fisheries in the eastern Pacific Ocean. The General Fisheries Commission for the Mediterranean (GFCM) has also passed a resolution to regulate catch of *Mobula* species. Publication of a field guide for *Mobula* and *Manta* species and increased awareness of the vulnerability of this group of species has reportedly improved data collection in industrial tuna fisheries.

Analysis: *Mobula* species are widely distributed in tropical and temperate seas worldwide. All have very low productivity and are taken in artisanal and large-scale fisheries, both as directed and incidental catch. The major driver for retention of catch is believed to be the international trade in gill plates, which are used for medicinal purposes in Asia, particularly in China. Market surveys indicate a significant increase in the market in recent years, with the most important products in trade being the plates of *Mobula japanica* and *M. tarapacana*. There is very little numerical population information although there is an estimate for one species (*M. mobular)* of ca. 15,000 individuals in the north western Mediterranean and south central Adriatic combined. Population declines have generally been inferred from declining catches despite increases in fishing effort in a number of locations. Some such declines have been very steep. Given the very low productivity of these species, the marked increase in the international market and evidence of declining catches it is possible that at least some species meet the criteria for inclusion in Appendix II in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16*).

Large gill plates of *M. japanica* resemble smaller plates of Appendix-II listed *Manta* species. This species appears to meet the criteria in Annex 2 b of *Res. Conf. 9.24 (Rev. CoP16*) (lookalike criteria). There is general similarity between gill plates of different *Mobula* species although some gill plates are bi-coloured and some are not. If any *Mobula* in either category (bi-coloured or black) were to be listed under the criteria in Annex 2 a, the others in that category would meet the criteria in Annex 2 b (lookalike criteria).

Reviewers of summary information only: V. Mundy; J. Kiska; L. Couturier; G. Notarbartolo di Sciara and G. Sant.

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Inclusion of Ocellate river stingray *Potamotrygon motoro* in Appendix II

Proponent: Plurinational State of Bolivia

Summary: The Ocellate River Stingray *Potamotrygon motoro* is a freshwater stingray from South America in the family Potamotrygonidae. It has a wide distribution in Argentina, Plurinational State of Bolivia (Bolivia), Brazil, Colombia, Ecuador, French Guiana, Guyana, Paraguay, Peru, Suriname and the Bolivarian Republic of Venezuela (Venezuela).

There are a number of taxonomic uncertainties surrounding Potamotrygonidae; it is possible that *P. motoro* may comprise a cluster of different species[1, 2, 3]. There are also reports of hybridisation between *P. motoro* and *P. falkneri* in the wild[4].

There is limited biological or population information available, but the species is believed to exhibit low fertility (with only the left ovary normally being present and functioning), long gestation periods (six months) and slow growth. Size at maturity has been estimated at various sizes, ranging from 20cm to 39cm for males and 24cm to 44cm for females. Age at maturity has been estimated at between 18 months and four years[2].

Information on population size is sparse. *P. motoro* was the predominant fish species around Marajó Island in the Amazon River in Brazil in 2005 to 2007, making up 50% of all catches; a 2009 study reported that artisanal fishers in Soure, Brazil, indicated that they frequently caught the species[1]. A 2016 study reported it was the most abundant *Potamotrygon* species found in a census of the Tomo River in Colombia, with a reported density of 0.3 individuals per 1000m[2 5]. It was found at low densities in fisheries captures in the Rio Negro of the Orinoco in Colombia[6]. Only 79 specimens of *P. motoro* (52 males and 27 females) were found during night-time visual surveys in the dry season in 2010 to 2011 in the Estrella Fluvial de Inírida region in Venezuela and Colombia, where *P. motoro* was reportedly historically abundant[7]. Detection rates, however are dependent on a number of factors, including water levels and temperature, and time of the year and day[8]. Encounter rates by fishers in the Amazon in Peru have reportedly declined in more accessible areas[9].

The main impacts negatively affecting *P. motoro* are believed to be commercial and artisanal fisheries for the ornamental fish trade, particularly targeting juveniles. *P. motoro* is also caught for local consumption/use of its meat, oil and spines and it also may be affected by habitat modification.

Catch and trade data for *P. motoro* exist for Peru, Colombia and Brazil, however there are many uncertainties over the quality of the data, due to identification issues and specimens being traded using their common names, and concerns that numbers may be overestimates[3]. There are inadequate catch data available in most of the key exporting countries to determine the proportion of total catch destined for domestic and international markets, although in Iquitos, Peru, a high proportion of all rays caught were exported[9].

From 2000 to 2014 recorded annual exports from Peru averaged 25,000 specimens, with a peak in 2008 of ca. 45,000 specimens, after which annual exports declined again to similar levels as 2000 to 2002 (fewer than 20,000 individuals)[10]. Between 1999 and 2011 Colombia reported the export of an annual average of ca. 6500, increasing steadily from 1999, with a peak of 20,200 in 2008[2], after which quantities declined again to below 10,000 specimens in 2010 to 2011. Brazil exported ca. 6000 per year in 2003 to 2005, after which their exports declined greatly, although data are not available for recent years[3].

Based on USA trade data for 2004 to 2013, imports of *Potamotrygon* species were reported only from Brazil, Colombia and Peru. Most trade is reported at genus level. This has shown an increase from negligible levels in 2004 to 2008 to over 1500 per year in 2012 to 2013[11].

Some export of freshwater rays in general has been reported from Argentina but at a very low level (ca. 75 per year for 2004 to 2013); no export has been reported from Bolivia, Ecuador, Paraguay, Uruguay or Venezuela. There is reported to be transborder movement of *Potamotrygon* species, including *P. motoro* from Venezuela to Colombia via Puerto Carreño and Puerto Inírida[3].

The principal destinations for exports are Europe, the USA and increasingly East Asia[2]. The species is offered for sale on the internet[12, 13, 14, 15, 16] and appears readily available in the aquarium trade, however the provenance of many of these specimens is unknown. This species is reported to breed easily in captivity, and captive-breeding is known to be occurring in Europe, Southeast Asia and the USA. There is reportedly a surplus of captive-bred *P. motoro* individuals in European public aquaria, and therefore any demand for

wild-caught specimens in Europe is believed to come mainly from private collectors[17, 18]. The USA reported the export of over 3500 captive-bred *Potamotrygon* specimens in 2004 to 2013[11].

Brazil, Colombia and Peru have specific regulations in place governing harvesting and trade of ornamental species, including *P. motoro*, and Bolivia reports having draft legislation to control trade in ornamental fish in Bolivia underway. At an international level, this group of species has been the focus of several CITES Decisions aimed at improving the available information on their taxonomy, biology, population sizes/trends, harvesting and trade, with an expert workshop in 2014 identifying possible priority species and future actions, including Appendix II and III listings. The Supporting Statement does not provide details on how it fits within the wider picture and recommended actions. A proposal to list this species and *P. schroederi* in Appendix II was submitted to CoP16 but was not accepted.

Potamotrygon motoro can supposedly be differentiated from other *Potamotrygon* species by their colour patterns/markings, however these taxonomic uncertainties are problematic for monitoring use and trade in this species.

Potamotrygon motoro is classified by IUCN as Data Deficient (2005 – needs updating), although there may be some confusion with taxonomy in that assessment as a much more limited range is reported[8].

Analysis: *Potamotrygon motoro* has a very wide distribution in South America. Information on population status and trends is sparse and variable. It has been reported as abundant in some places and as occurring at low densities in others. There are some indications of declines in some locations. The species is taken in fisheries for local consumption of its meat and export of live specimens for the ornamental fish trade. Three out of eleven range States (Brazil, Colombia and Peru) are known to export the species. Reported export has been of the order of a few tens of thousands of specimens annually. These exporting countries comprise a reasonably large proportion of the overall range of the species, although it is not known how extensive harvest for export in any of the three known exporting range States is, or in general what proportion of the catch in these States is destined for export rather than for local consumption, although it may be significant in Peru. There is overall insufficient information to determine whether this species meets the criteria for inclusion in Appendix II.

Reviewers of summary information only: J. González Sanz, H. Ortega Torres, M.L. Goes de Araujo and G. Sant.

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Inclusion of Banggai Cardinalfish *Pterapogon kauderni* in Appendix II

Proponent: European Union

Summary: The Banggai Cardinal Fish *Pterapogon kauderni* is a small marine fish endemic to the Banggai Archipelago off Central Sulawesi, eastern Indonesia. It has a natural range of ca. 5500km^2 and occurs in isolated small populations in the shallow waters of 34 islands, primarily in sea grass beds and on coral reefs. The total area of potential habitat within the range is estimated at around 23km^2. There are also a few introduced populations in a restricted area outside the species natural area of occurrence, which account for a very small proportion of the total[1]. *P. kauderni* is a benthic, site-attached species with a generation time estimated at about two years. It exhibits relatively low fecundity, often with reduced fertility, direct development and extended parental care. There is no planktonic phase so dispersal and potential to colonise or re-colonise areas are very limited.

Harvesting for international trade is considered by most experts to be the principal factor affecting the species; it is believed to have led to substantial population declines and local extinctions within its natural range. The species is also adversely affected by habitat loss and deterioration through loss of coral cover, primarily as a result of destructive fishing methods, and declines in abundance through harvest of sea urchins, anemones and anemone-like corals on which it is dependent. The extent, if any, to which the species can adapt to severely altered habitats is unknown, but believed likely to be limited.

Live individuals for the aquarium market are the only product in trade. The majority are exported to the USA, Europe and Asia. The export fishery started in the mid-1990s, and by 2007 the annual harvest was estimated at some 900,000 fish.

There is an estimated mortality rate of 25 to 50% between actual harvest and compilation of trade/import figures. In 2015, holding nets containing thousands of *P. kauderni* were recorded on several islands, indicating that collection pressure may not have decreased.

A new shipping method, with increased use of “public” transportation (small and medium size boats, and speed boats) for shipping *P. kauderni* directly out of the Archipelago, has meant that captures are not being reported to the local (Banggai) fisheries/quarantine office, and assessment of trade volumes and shipping mortality has become more challenging.

There have been many studies of this species by different authors, and some areas and populations have been tracked over 15 years with data available for 2001, 2002, 2004, 2007 and 2015. The overall consensus is that the population is declining. Information on population levels before exploitation started is lacking. Studies in 2002 and 2004 of a population that was not then exploited estimated it to have a density of ca. 0.6 individuals per m^2.

Censuses conducted between 2001 and 2004 covering the entire range produced mean densities of between 0.07 to 0.08 fish per m^2 and an overall population estimate of 2.4 million fishes, based on 34km^2 of suitable habitat. A more restricted survey in 2011 to 2012 found a mean density of 0.05 fishes per m^2 indicating overall abundance of around 1.7 million fishes. Seven of the major sites surveyed in 2001 to 2004 and in 2011 to 2012 showed declines in mean density and overall abundance, presumed to be due to over-exploitation. In 2015, no population was found with a density near to 0.6 individuals per m^2. At sites where fishing intensity is high, the mean number of groups per census site declined by 27% from 2007, and the mean group size of censused populations in 2015 showed a ca. 40% reduction from the mean size group of 2007.

Extirpation of local populations ascribed to exploitation has been documented in several islands. Research suggests that once population densities decline to ca. 0.02 individuals per m^2, they may be unable to recover. Temporary local recovery of populations has been recorded at sites where fishing has been stopped, although in two documented cases these populations subsequently collapsed.

In Indonesia, a Banggai Cardinalfish Action Plan (BCF-AP) was drawn up for the period 2007 to 2012, and included the establishment of the Banggai Cardinalfish Centre (BCFC) to coordinate conservation and management actions. Trade quotas were proposed by local stakeholders in 2010 but were not continued, mainly due to a lack of legal support. By 2012, there was reportedly still no effective long-term conservation, management or monitoring system in place. A marine protected area was established in 2007 in part to help

conserve the species, but there has been no evidence of implementation or management of the area, and a large part of the protected area falls outside the range of the species.

The species can be bred relatively easily in captivity although wild-caught fish are currently cheaper. An NGO captive-breeding facility has just opened in Indonesia and sent its first exports to the UK[2]; other commercial captive-breeding facilities are reportedly exporting, according to internet sites, but details are not available.

The species was categorised as Endangered by IUCN in 2007.

Since *P. kauderni* are harvested from nearshore environments within the Indonesian Exclusive Economic Zone (EEZ), "introduction from the sea" is not an issue for this species.

Analysis: *Pterapogon kauderni* is a marine species with a very restricted range whose biological characteristics make it vulnerable to overexploitation. It has been harvested in large numbers since the mid-1990s for the international aquarium fish trade, with exploitation continuing. Available evidence indicates that this has led to significant and continuing reductions in population density and overall population size. The species is also affected by habitat loss and degradation. There appears to be no effective long-term management in place. It would appear that the species meets the criteria for inclusion in Appendix II in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)* in that regulation of harvest is required to ensure that the wild population does not become threatened through continued harvesting or other influences.

Reviewers of summary information only: G. Lilley, A. Vagelli, K. Carpenter, E. Wood and A. Rhyne.

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Information not referenced in the Summary section is from the Supporting Statement.

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Inclusion of Clarion Angelfish *Holacanthus clarionensis* in Appendix II

Proponent: Mexico

Summary: The Clarion Angelfish *Holacanthus clarionensis* occurs in the Pacific territorial waters of Mexico, in the Revillagigedo Archipelago and off Baja California Sur, and Clipperton Island (France). It is demersal and is found associated with coral reefs and rocky areas, including drop-offs, to a depth of 30m. It is commonest at around three to five metres depth, particularly around the cleaning stations of mantas (*Manta birostris*). Maturity is believed to be reached at between 18 and 30 months and life expectancy to be around 10 years. Observed population densities at sites vary seasonally, which may be associated with spawning migrations. A low frequency of juveniles in reef populations indicates that recruitment may take place elsewhere, in shallower or deeper waters.

The maximum potential area of occupancy for this species is probably ca. 4000km^2 [1]; the amount of suitable reef within this is smaller, so that actual area of occupancy is probably considerably less than 2000km^2. The majority of the population reportedly occurs within less than 50km^2 in the Revillagigedo Archipelago, where average population density has been estimated at around one individual per 200m^2 in suitable habitat, compared with fewer than one per hectare (10,000m^2) found in surveys off Baja California Sur. The population at Revillagigedo has been reported as apparently stable[2]. Report of a decline in the late 2000s at Cabo Pulmo, Baja California, where the species is generally extremely scarce, is not considered reliable[3]. In 1997 it was reported as rare at Clipperton Island[4].

Loss of habitat, and particularly damage to coral reefs, may affect the species, although its range in the Revillagigedo Islands is within a protected area. There is speculation that an increase in duration and frequency of El Niño–Southern Oscillation (ENSO) events associated with climate change might have an impact on the species[2].

Clarion Angelfish are very brightly coloured and have appeal in the ornamental fish trade. However, they are aggressive fish which are not suitable for community reef tanks; demand for them has therefore always been limited[5]. They do, however, command high prices in trade.

Domestic trade in Mexico is regarded as negligible. The majority of exports reportedly go to California, USA, although exports to Japan are also known[5]. Data on exports of live specimens from Mexico are variable, but no more than a few hundred per year (generally 200 to 600 based on the statistics available) enter the market. Between 2007 and 2013, Mexico gave permits for the collection of just over 3000 specimens and reportedly around 2750 individuals were exported although it has not been possible to confirm this. Trade data for the period 2006 to 2013 report only 625 Clarion Angelfish in total were imported into the USA from Mexico[6]. There is no information on mortality of individuals between capture and export.

Historically, there is a report from the early 1990s that over 1000 specimens were being collected at the Revillagigedo Archipelago on individual fishing trips, and that this was leading to significant depletion of the population, but no further information is available.

In Mexico, the species is considered subject to special protection, meaning that harvesting should only be undertaken if it is sustainable. Capture under permit is allowed in only three zones in the Gulf of California; in other parts of its range, including the Revillagigedo Archipelago, harvest is not permitted. It has been speculated that specimens taken under licence may in fact originate in the Revillagigedo Archipelago, although evidence is lacking.

The species was classified as Vulnerable by IUCN in 2010.

The species is bred at a commercial aquarium exporting facility in Bali, Indonesia; exports at a low level are known to take place from here to the United Kingdom and USA.

Analysis: The Clarion Angelfish is collected for export for the international marine aquarium fish trade and has a relatively limited range and population for a marine fish. Harvest for export from the main range State (Mexico) is limited and controlled by licence. The major part of the population occurs in a protected area where collection is not allowed, and is believed to be stable, although it may be vulnerable in the long term to pressures related to climate change. The species does not appear to meet the criteria for inclusion in Appendix II.

Reviewers of summary information only: K. Carpenter and E. Wood.

References:

Information not referenced in the Summary section are from the Supporting Statement.

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[1] Wells, S. (2016) *In litt.* to IUCN/TRAFFIC Analyses Team, Cambridge, UK.

[2] Pyle, R., Myers, R., Rocha, L.A. & Robertson, R. (2010) *Holacanthus clarionensis*. The IUCN Red List of Threatened Species 2010.

[3] Wood, E. (2016) *In litt.* to IUCN/TRAFFIC Analyses Team, Cambridge, UK.

[4] Allen, G.R. and Robertson, D.R. (1997) An annotated checklist of the fishes of Clipperton Atoll, tropical eastern Pacific. *Revista de Biologia Tropical* 45: 813-843.

[5] Jones, R. (2016) *In litt.* to IUCN/TRAFFIC Analyses Team, Cambridge, UK.

[6] Analysis of US Fish & Wildlife Service Law Enforcement Management Information System (LEMIS) data, May 2016.

Inclusion of all species of Chambered Nautiluses in the Family Nautilidae in Appendix II

Proponents: Fiji, India, Palau and United States of America

Summary: The Family Nautilidae, or Chambered Nautiluses, is a highly distinctive group of marine molluscs occurring in tropical, reef, and deep-water habitats in the Indo-Pacific. Two genera are recognised *Allonautilus* and *Nautilus.* The genus *Allonautilus* is generally considered to have two species: *A. perforatus* (known from Indonesia and possibly Papua New Guinea[1]) and *A. scrobiculatus* (known from Papua New Guinea and possibly the Solomon Islands[1]). The number of species in the genus *Nautilus* ranges from two to 12 according to different authors; the most recent taxonomic work considers that there are two species, *Nautilus pompilius* and *N. macromphalus*[1]. *N. pompilius* has a wide range, being known from 11 range States and possibly occurring in five others, from India in the west to American Samoa in the east. The Supporting Statement lists four other species: *N. macromphalus* endemic to New Caledonia and also *N. belauensis, N. repertus* and *N. stenomphalus* which are from Palau and Australia and are now considered to be part of *N. pompilius*. It is thought there may also be other as yet unrecognized but separate species existing as genetically distinct, geographically-and reproductively-isolated populations.

Chambered Nautiluses are extreme habitat specialists living in close association with steep-sloped fore reefs and associated silty, muddy or sandy-bottomed substrates, in preferred depths of 150 to 300m and rarely down to 700m. Distribution is patchy and erratic and they may be absent from apparently suitable habitats. They have a relatively narrow temperature range tolerance. Geographic barriers to movement include shallow areas where water temperatures exceed 25°C and open water areas which Chambered Nautiluses avoid, presumably because they are vulnerable to predation there.

They are slow-growing, late-maturing (10 to 15 years) and long-lived (at least 20 years), producing one large egg at a time[1] that requires a lengthy incubation period (about one year) and lacking a mobile larval stage. It is not known how many eggs a single wild female might lay over an entire year. Trapping data indicate that juvenile Chambered Nautiluses represent less than 10 to 20% of populations, indicative of a low-productivity species. The majority of animals captured in traps are male suggesting a male-biased sex ratio and a population structure based on multiple paternity; there is no evidence to suggest that adult males might be more likely to enter baited traps[1].

There are no global population estimates but there is good evidence that populations are naturally sparse, small, and isolated. Surveys have found abundance of most unexploited populations of *N. pompilius* to be low and in some cases less than one individual per km^2 (Australia, Fiji, American Samoa), but on one reef in Australia, abundance was found to be 10 to 15 individuals per km^2. Their attraction to baited traps and ease of recapture[1] may give a false impression of their abundance.

Chambered Nautiluses are the object of targeted fisheries and may also be caught incidentally in other fisheries. Commercial harvesters use fish traps baited with meat dropped to depths of 130 to 250m. The largest commercial fisheries are in the Philippines and Indonesia. The shells and the meat are both used, although the latter is thought to be essentially a by-product, with the shell the primary product in trade. Shells are sold whole as decorative objects or collectors' items and also in pieces, for example in inlay. There is both domestic use and international trade. Much of the domestic sale of shells is to tourists; a proportion of this is likely to be destined for export as personal effects. Some, though probably only a small proportion, of the supply of Chambered Nautilus shells is provided by beach-drift specimens. There is a small amount of use of live specimens for display in aquaria and research.

The Philippines and Indonesia have the largest commercial fisheries. In the Philippines harvest and trade of Chambered Nautiluses has occurred at least since the 1970s. A catch survey in 2001/2002 in Panay estimated an annual harvest of some 12,200 *N. pompilius*; in Palawan, about 9000 animals were reported harvested in 2013 and 37,000 in 2014. More than 18,500 whole shells were found in a recent survey of 162 shops across the Philippines. Commercial harvested reportedly occurs widely in Indonesia with products sold locally and exported. Export in recent years has been notable. According to USA trade data, between 2004 and 2013, some 3700 shells, were exported from Indonesia to the USA, the majority (2630) in 2007[2]. In addition between 2007 and 2010 up to 25,000 specimens were reportedly exported for their meat from Indonesia to China. Overall, however, there is little information on the relative importance of harvest for export compared with that for domestic use in Indonesia.

Targeted fisheries are reported to have taken place in the past in New Caledonia (France), Palau and Vanuatu. Harvests are also thought to take place in China (notably Hainan) and Papua New Guinea, but the extent and impact of these and the proportion, if any, of the harvest that enters international trade is unknown.

There are numerous importing countries but data are available for the USA only. In the period 2005 to 2014, an annual average of around 12,000 whole individuals and over 85,000 parts was imported into the USA, almost all from the Philippines (85%) and Indonesia (12%)[2]. Virtually all imports were reported as *N. pompilius*, but all other species except *A. scrobiculatus and N. stenomphalus* were reported in trade albeit in very small numbers (some in under 10 specimens). Total annual imports to the USA declined over this period. Imports from the Philippines to the USA declined markedly after 2009, and a shift to imports from Indonesia suggests there may have been a switch from Philippine suppliers to Indonesian suppliers.

Most information on population changes comes from the Philippines. Abundance estimates on a reef in Bohol that is subject to commercial harvest were one to three orders of magnitude lower than those of unfished populations. Trap yields from Tañon Strait reportedly declined by 97% between the 1970s when a fishery started and the 1980s when the fishery ceased as the species was considered commercially extinct; Chambered Nautiluses now appear to be completely absent here[1]. Anecdotal reports and results of surveys of harvesters and traders indicate declines, some severe, elsewhere (Palawan, the Visayan Regions and Tawi-Tawi Province). It has been suggested that *N. pompilius* populations in the Philippines are being serially depleted and that trade may be shifting to Indonesia and elsewhere.

There are reports of declines associated with harvest in India (where *N. pompilius* occurs), Indonesia (where *A. perforatus* and *N. pompilius* occur) and New Caledonia (where *N. macromphalus* and *N. pompilius* occur), although very little quantitative information is available.

The habitat on which Chambered Nautiluses are dependent is affected by pressures that have an impact on deep reefs (150m and deeper) such as pollution, sedimentation, deep water mining and fishing[3], and climate change (sea water warming and acidification).

Chambered Nautiluses are not known to be included in any fisheries management plans. Chambered Nautiluses have been protected in Indonesia since 1990. Enforcement is reportedly poor, as evidenced by the quantities exported to the USA, although seizures of shells have been made. Harvest of *N. pompilius* in China requires a permit. Captive-breeding has never been successful; eggs have been hatched but none has been raised to maturity.

The shells of different Chambered Nautilus species resemble each other. Experts are generally able to distinguish between different species but non-experts have difficulty doing so, and species are usually not differentiated in international trade.

Analysis: Chambered Nautiluses are believed generally to occur in small, scattered populations. They are highly vulnerable to overexploitation and are known to be targeted in fisheries, with the products, chiefly shells, known to enter international trade. The main species in trade, *Nautilus pompilius* has an extensive range in the Indo-Pacific. In one range State – the Philippines – harvest has been associated with severe local population declines; the country has exported large quantities of Chambered Nautiluses and it seems that international trade is a significant driver of the harvest. There are indications that harvest for trade has now shifted elsewhere. There are reports of historic and ongoing declines associated with harvest in other parts of the range. It is unclear how extensive such declines are or how important international trade is as a driver of harvest relative to domestic consumption. However, given the extreme vulnerability of Chambered Nautiluses to overharvest, any additional fishing pressure as a result of harvest for export is likely to lead to depletion or local extirpation of populations. Given this and the absence of management plans for the species, it is likely that *N. pompilius* at least meets the criteria for inclusion in Appendix II in Annex 2a of *Res. Conf. 9.24 (Rev. CoP16).*

Chambered Nautilus species resemble each other in the major form in which they appear in trade (shells) so given that *N. pompilius* appears to meet the criteria, all other species in the Family Nautilidae would therefore appear to meet the criteria in Annex 2b (lookalike criteria).

Reviewers of summary information only: P. Ward and E. Woods.

References:

Information not referenced in the Summary section is from the Supporting Statement.

[1] Ward, P. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team. Cambridge, UK.
[2] Analysis of US Fish & Wildlife Service Law Enforcement Management Information System (LEMIS) data, May 2016.
[3] Woods, E. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team. Cambridge, UK.

Inclusion of all species of Cuban land snails in the Genus *Polymita* in Appendix I

Proponent: Cuba

Summary: The genus *Polymita*, commonly known as Cuban land snails, contains six species; *P. brocheri*, *P. muscarum*, *P. picta* (the largest species of the genus), *P. sulphurosa*, *P. venusta* and *P. versicolor.* All are endemic to Cuba. They are arboreal and adapted to live on a variety of plants including coffee trees and coconut palms. Shell size is 2 to 3cm. In the wild, *Polymita* are estimated to live between 12 and 19 months, reaching sexual maturity at between nine and ten months. Clutch size varies, but is generally from 30 to 100 or more. Individuals (which are hermaphrodite) breed only once.

The shells of *Polymita* are beautiful with a wide range of colours, and are sold as handicrafts (such as necklaces) or to shell collectors.

Potential distribution maps (provided in the Supporting Statement) for each species have been calculated based on known distribution maps that have had suitable habitat that no longer exists and extirpated sub-populations removed; these provide the basis for the estimated of area of distribution provided below.

Polymita brocheri has a current distribution estimated at just over 50km^2 of fragmented habitat; original distribution is estimated at ca. 70km^2 (ca. 30% reduction). Population can reportedly be locally abundant (3.5 individuals per m^2); density overall has been calculated at 0.1 individuals per m^2 [1]. It is reported to be used domestically for fishing bait[1]. Not known to be present in any protected area[3].

Polymita muscarum has a current distribution estimated at ca. 3600km^2; original distribution is estimated at ca. 8000km^2 (56% reduction). Population size is unknown. Inland populations have reportedly been greatly reduced or have disappeared and the species is now largely confined to a narrow strip of coastal vegetation where development for tourism has led to habitat fragmentation. In 22 coastal towns the population was at a very low density (0.002 to 0.3 individuals per m^2). Known to occur in at least one protected area[3].

Polymita picta has a current distribution estimated at ca. 2200km^2, marginally reduced from the estimated original distribution (ca. 2400km^2). Population size is unknown. Surveys of 39 localities found an average density of between 0.01 and one individual per m^2 [1]. Populations not targeted for harvest had higher reported densities than exploited populations[4] but it is not clear whether this was due to surveys taking place at different times of the year, or a genuine indication of harvesting impact. Local extinctions of populations have been reported[1]. The species is not present within any protected areas[3] and it is impacted by habitat modification and fragmentation, as well as pesticide use in coffee plantations which is known to have caused a mass die-off in one plantation[1].

Polymita sulphurosa is believed to have an extremely limited distribution, potential habitat within its range comprising patches totalling a few square kilometres (there are differing estimates of 1.3 to 2.5km^2 and ca. 7km^2)[2, 3]. Original range has been estimated at ca. 200 km^2 (98% reduction). Remaining habitat suffers a severe level of fragmentation. Studies conducted from 1995 to 2004 at different times of year found populations at only 25% of localities reported in the literature. Reported densities are low (estimates of 0.08 to 0.4 and 0.001 to 0.1 individuals per m^2 [1]). An expedition in 2015 located only one individual in one patch of 1.31km^2. Habitat is impacted by intense land use for sugarcane agriculture, livestock grazing and subsistence crops. Depleted populations are believed to be at risk from over-harvest, continuous habitat fragmentation and incidental killings[4]. The species is not present in any protected areas[3].

Polymita venusta has a current distribution estimated at ca. 8000km^2 and the habitat is fragmented; original distribution is estimated at ca. 20,000km^2 (ca. 60% decrease). Studies at three sites found population density to fluctuate greatly over time. In some areas only relict populations are found, in areas of low agricultural interest. Known to occur in at least one protected area[3].

Polymita versicolor has a current distribution estimated at ca. 100km^2 and the habitat is fragmented; original distribution is estimated at ca. 130km^2 (ca. 25% decrease). Average population density at four locations has been estimated at 0.02 individuals per m^2 [1]. Local extinctions have been reported, ascribed to changes in land use for agriculture, gypsum mining, grazing and housing development and, in one case, possibly to over-collection. Known to occur in at least one protected area[3].

International exports were reported in the 1940s to be high (estimated at 0.5 million per year), and although an export ban (domestic and international trade prohibited without licenses) was introduced in 1943, trade

continued to the USA and Canada until the 1960s. In the last 20 years, only two legal exports have been recorded (55 shells and 35 live specimens). Between 2012 and 2016, Cuban Customs made 15 seizures totalling more than 23,400 shells being exported to the USA. One expert noted that the bulk of the trade goes to Europe and from there to Asia[4].

Species that are illegally traded in the greatest numbers are said to be *P. picta, P. sulphurosa, P. versicolor* and *P. venusta,* although all species can be found in international trade, and the most attractive and varied morphs are selected for harvest[1]. Most international trade is in fresh shells taken from the wild, rather than recycled from old collections[1,4], and the main demand now is said to be from tourists[1]. Collectors will gather live snails and empty shells when harvesting[1]: it is not clear what proportion of shells is from live snails. The majority of live snails are collected before they reach maturity[1, 5].

Polymita brocheri has a distinctive shape to its shell, which differs from that of the other species. Some of the other species have distinctive patterning and colouration, making them relatively easy to identify, but in some species (e.g. *P. venusta* and *P. picta*) there is considerable intraspecific variation, so that enforcement agents may have difficulty identifying specimens to species level with confidence[5, 6]. Photographs to facilitate their identification are available.

Analysis:

Polymita sulphurosa has a very limited and fragmented range in which it appears to be rare, with evidence of marked historic decline in area of distribution. It would appear to meet the biological criteria for inclusion in Appendix I.

Polymita brocheri (52km^2) and ***P. versicolor*** (99km^2) have relatively small ranges. An average population density of 0.1 individuals per m^2 (equivalent to 100,000 per km^2) has been estimated for *P. brocheri* and of 0.02 individuals per m^2 (equivalent to 20,000 per km^2) for *P. versicolor*. Even if these species are only found at these densities in a portion of their ranges, these figures indicate that their populations are not small. Declines in population have been deduced largely on the basis of reduction in available habitat, but indications are that these declines have not been marked in the sense of *Res. Conf. 9.24 (Rev. CoP16*). These species would not appear to meet the biological criteria for inclusion in Appendix I in Annex 1 of *Res. Conf. 9.24 (Rev. CoP16)*.

The three remaining species, ***P. muscarum*** and ***P. venusta*. *P. picta,*** have relatively extensive areas of distribution (2200 to 8000km^2) and, on the basis of population density figures, very large populations. All are believed to have undergone population declines largely as a result of declines in available habitat, but these are unlikely to be near the guidelines in Annex 1 of *Res. Conf. 9.24 (Rev. CoP16)* (in this case a reduction of 50% or more in ten years, generation time being ca. one year). These species would not appear to meet the biological criteria for inclusion in Appendix I in Annex 1 of *Res. Conf. 9.24 (Rev. CoP16)*.

All species are potentially affected by trade (which is illegal) and therefore appear to meet the trade criterion for inclusion in Appendix I.

Reviewers of summary information only: R. Kramer.

References:
Information not referenced in the Summary section is from the Supporting Statement.

[1] Hernández, N. (2016) *In litt*. to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
[2] Perez, E., Osa, E., Matamoros, Y., Shillcox, J. & Seal, U.S. (1998) *Conservation Breeding Specialist Group (SSC/IUCN). Report of Conservation Assessment and Management Plan Workshop For Selected Cuban Species*: Cbsg, Apple Valley, Minnesota 55124, USA.
[3] Mauriño, E. R. (2001) Proyecto de investigación - Ecología y conservación del molusco gasterópodo *Polymita sulphurosa* en Cuba. *Cuadernos de biodiversidad*. 7:14-17.
[4] González, A. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
[5] Torres, M.M. (2016) *In litt*. to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
[6] Cowie, R. (2016) *In litt*. to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

Inclusion of all species of Ponytail Palms in the Genus *Beacarnea* in Appendix II

Proponent: Mexico

Summary: *Beaucarnea* species, known as Ponytail Palms (although not strictly palms), occur in Mexico and northern parts of Central America (possibly as far south as northern Nicaragua). According to the Kew Plant Checklist there are nine accepted species*: Beaucarnea compacta, B. goldmanii, B. gracilis, B. guatemalensis, B. hiriartiae, B. pliabilis, B. recurvata, B. sanctomariana* and *B. stricta.* Two other species listed as synonyms (*B. inermis*, *B. purpusii*) are sometimes recognised as separate species[1] *Beaucarnea* species feature in the horticultural plant trade, with *B. recurvata* the main species in trade. It is still frequently traded under the synonym *Nolina recurvata*.

Beaucarnea recurvata can reach almost 15m in height and is endemic to the low deciduous forests in rocky and mountainous areas of Veracruz and Oaxaca, Mexico. Regeneration is reportedly limited. Individuals do not flower every year, and establishment rates are apparently low, owing to a lack of water, herbivory grazing, extraction and other causes. Adult plants may live for hundreds of years, and are reported to start flowering when they reach three metres tall after around 30 years (artificially propagated plants reportedly flower sooner)[1]. There is no accurate estimate of the number or size of wild populations. Maximum recorded density is of 135 individuals per ha (calculated from an area of 1.2ha in Veracruz). A study currently underway in central Veracruz has found few populations with more than 30 adult individuals, although it is believed that such populations exist in inaccessible areas, and that isolated individuals are still relatively abundant in the range[1]. Observed population structure apparently varies according to site accessibility, with relatively few seedlings and juveniles observed along flat roads near areas of human population.

There are nurseries in Mexico (registered under the Unidades de Manejo para la Conservación de la Vida Silvestre (UMA) or Predios Intensivos de Manejo y Vida Silvestre (PIMVS) systems) that legally propagate *Beaucarnea* species; it is reported that volume of production and sizes of available specimens do not satisfy the demand and that seeds, seedlings, juveniles and adults are consequently harvested from the wild to supplement artificial propagation. The main *B. recurvata* mass producers are reported to depend entirely on seeds from the wild. Flowering specimens produce an average of approximately seven inflorescences, with each inflorescence typically producing more than 2000 seeds. Only one nursery is known to have a closed cycle of production, and it is very small compared to mass producers in the main region of production of *B. recurvata* in Mexico[1]. While examples of sustainable management with scientifically-based harvest limits for seed are in place, such production is reportedly undercut by cultivation of specimens illegally harvested from the wild[2]. Mexican seizure data indicate that over 2000 specimens were confiscated from nurseries in the period 2004 to 2014.

Beaucarnea recurvata is also widely propagated outside Mexico and very common in ornamental plant markets in Europe and elsewhere. Denmark has reported an annual average export of 200,000 specimens. Information collected from various countries in the European Union noted that China is a major source of propagated specimens. The origin of parent material of live plants offered outside Mexico is unknown, although the species has been widely in cultivation since the first half of the nineteenth century[3].

There is virtually no information on export from Mexico. Trade data from the USA do not report any imports from Mexico in the period 2004 to 2013[4]. It has been stated that wild-collected plants are exported after acclimatisation in nurseries in Mexico although there does not appear to be clear evidence for this.

Other *Beaucarnea* species that are known to be in cultivation are *B. inermis, B. goldmanii, B. pliabilis, B. hiriartiae* and *B. guatemalensis.* Limited trade in seeds of *B. gracilis, B. stricta* and *B. sanctomariana* has also been recorded.

Juvenile and adult *Beaucarnea* specimens resemble each other to varying degrees. They can be distinguished to species level with some training, with information on identification published in a number of manuals[5, 6, 7]. However, seeds and seedlings cannot be easily identified to species level by non-specialists. There are a number of synonyms used for species in this genus, including species falling under four other genera (*Dasylirion, Dracaena, Nolina* and *Pincenectitia*). *B. inermis* is considered a synonym of *B. recurvata*[8] (however it is included as an accepted species in this proposal) and *B. recurvata* var. *stricta* is considered a synonym for *B. stricta*.

The species has not been assessed by IUCN, but the Norma Oficial Mexicana NOM-059-SEMARNAT-2010 (the Mexican National Red List) classifies *B. recurvata* as threatened (A) although more recent consideration by experts, indicated that it could instead be classified at risk of extinction.

Analysis: *Beaucarnea recurvata* is an extremely widely grown ornamental plant, cultivated both within its range State (Mexico) and elsewhere. Wild populations are scattered over a relatively wide area; there is no information on total numbers or trends. There are reports of wild-collection of seeds as source material for nursery propagation in Mexico, and of collection of plants of varying sizes for the horticultural plant trade. It has been stated that some of the trade in wild-collected plants is destined for export although there appears to be no clear evidence for this. There are indications of fewer young plants in accessible populations than in less accessible ones, but no other information on the possible impact of harvest on wild populations. The species has been in cultivation outside the range State for many years; it is likely that established cultivated stock can provide both large plants for sale and source material (seeds) for propagation in adequate quantities to meet market demands. The species seems likely not to meet the criteria for inclusion in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16*).

Evidence for trade in wild-collected plants of other *Beaucarnea* species is extremely limited and there is no indication that any of them meets the criteria in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16*).

Species of *Beaucarnea* resemble each other to varying degrees. If any species were to be included in Appendix II, the others would meet the criteria in Annex 2 b of the Resolution (lookalike criteria).

Reviewers of summary information only: L. Hernandez Sandoval, M. Chazaro and A. Contreras Hernandez.

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Information not referenced in the Summary section is from the Supporting Statement.

[1] Contreras Hernandez, A. (2016) *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
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[8] The Plant List (2013) Version 1.1. http://www.theplantlist.org/. http://www.theplantlist.org/tpl1.1/record/kew-300342, and http://www.theplantlist.org/tpl1.1/record/kew-300352.Viewed on 4th July 2016.

Deletion of *Tillandsia mauryana* from Appendix II

Proponent: Mexico

Summary: *Tillandsia mauryana* is a bromeliad plant endemic to Mexico. It has a limited range in Hidalgo State where it occurs on the vertical faces of limestone cliffs that are difficult to access. Surveys have located 31 populations of this species but, due to site inaccessibility, it has only been possible to evaluate abundance and population density in 9 of these. These contained between 3 and 304 individuals. Only a small proportion of the population at each site reproduces each year and the overall population may be decreasing[1]. Its range is located mainly in the Metztitlán Gully Biosphere Reserve, an area affected by rock mining, road building and urban development. The area's management programme contains specific actions for the protection of the species.

There are around 540 species of *Tillandsia* ranging from the southern USA to Argentina and Chile. A few species are widely distributed, but most have limited ranges. *Tillandsia* species in general feature in the horticultural plant trade. Some forms are artificially propagated in very large numbers and widely sold as ornamental plants. Others are grown largely by enthusiasts. *T. mauryana* was included in Appendix II in 1992 owing to concerns regarding the possible impact on it of wild-collection for international trade. The original listing proposal at CoP8 covered all *Tillandsia* spp. At the CoP it was agreed to include only seven species, including two from Mexico: *T. mauryana* and *T. xerographica* (the latter also found in Guatemala).

Since the species was listed, around 190 plants have been recorded in trade, mainly between Hungary and Switzerland, and all reported as artificially propagated. No trade in this species has been recorded from Mexico, no exports of wild specimens have been reported and there is no evidence of ongoing wild collection or illegal trade. The CITES Trade Database records a very small number of specimens of unidentified *Tillandsia* spp. as confiscated by the USA (90 specimens between 1993 and 2014); some 175 *Tillandsia* spp. occur in Mexico.

Artificial propagation of this species from seed is known to occur in nurseries in Germany and Hungary, and artificially propagated plants are offered for sale on the internet in a number of other countries, including the Czech Republic, Switzerland and the USA. Demand for this species by enthusiasts appears to be low and seems to be fully supplied by artificially propagated specimens.

Tillandsia mauryana is not similar to other CITES-listed *Tillandsia* species but does resemble other species that are not listed in the Appendices.

This proposal has resulted from the Plants Committee's Periodic Review process.

Analysis: It would appear that *Tillandsia mauryana* does not fulfil the criteria for inclusion in Appendix II as regulation of trade is not required to prevent harvesting of specimens from the wild from threatening the survival of the species. No export of wild harvested plants has been recorded since the species was listed in Appendix II and it seems that the limited demand for specimens is met entirely with artificially propagated plants. The species has not been subject to a recommendation under the provisions of the Review of Significant Trade within the last two intervals between meetings of the Conference of the Parties. It seems unlikely that its removal from the Appendices would stimulate trade in wild specimens such that it would meet the criteria for listing in Appendix II in the near future, as outlined in the precautionary measures in Annex 4 of *Res. Conf. 9.24 (Rev. CoP16)*, nor is its retention required to ensure that trade in any other Appendix-II listed species is effectively controlled.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] Valverde, T., Mondragón D., & Hernández-Apolinar, M. (2013) *Evaluación de la situación de Tillandsia mauryana en el Apéndice II de la CITES, según su estado de conservación y comercio*. Informe final CONABIO proyecto KE003. Facultad de Ciencias, UNAM. México.

Transfer fishhook cacti *Sclerocactus spinosior* ssp. *blainei* (= *S. blainei*), *S. cloverae* (CITES-listed synonym of *S. parviflorus*), and *S. sileri* from Appendix II to Appendix I

Proponent: United States of America

Summary: Cacti in the genus *Sclerocactus* are slow-growing, short, cylindrical, spiny plants occurring in southwestern USA and northern Mexico, with the majority of species endemic to the USA, including those covered by this proposal. Current CITES taxonomy recognises 20 species*;* eight species and one subspecies are included in Appendix I, the remainder are in Appendix II under the general listing for Cactaceae. A provisionally accepted revised taxonomy recognises the taxon currently listed as *Sclerocactus spinosior* ssp. *blainei* as *S. blainei* and splits *S. cloverae* from *S. parviflorus*[1]*.* This nomenclature is followed in this analysis.

Sclerocactus blainei (*S. s. blainei)* has a narrow distribution and is known from three occurrences in Nevada and Utah. Its population size is unknown. No information is available on population trends, but declining rainfall and prolonged drought conditions are believed to have impacted seedling recruitment and adult survivorship of *Sclerocactus* species in general. No trade in this taxon has ever been reported in the CITES Trade Database; a very small amount of trade in what is regarded as the parent taxon under current CITES taxonomy (*S. spinosior*) has been reported, in plants declared as artificially propagated and none from the USA. Seeds advertised as of *S. s. blainei* can be found for sale online outside the USA[2].

Sclerocactus cloverae (*S. parviflorus*) is known from 21 to 80 occurrences in Colorado and New Mexico. The estimated range is approximately 25,900km^2; individuals are generally scattered in suitable habitat but it may be locally abundant[3]; individual populations may be relatively short-lived[4]. A population estimate of ca. 10,000 has been made, but some of these may have been *S. parviflorus* (as currently recognised) or *S. whipplei* which are similar in appearance. Only a very small amount of trade in *S. parviflorus* has ever been recorded in the CITES Trade Database, all reported as of artificially propagated origin. It is not known if any of this trade was in the taxon regarded in this proposal as *S. cloverae*. Grafted (artificially propagated) plants advertised as *S. cloverae* are offered for sale in Europe[5].

Sclerocactus sileri is known from 10 to 12 occurrences in Arizona, and has a distribution there of approximately 1000km^2. According to recent work this species is also found in Utah. The species has reportedly suffered declines caused by fire, and is now uncommon[6]. Most of the populations are said to be very small (two to ten plants)[3]. The species has a generation length of four years (to first flower)[6]. No trade in this taxon has ever been reported in the CITES Trade Database and no evidence could be found of plants or seeds offered for sale. It was classified by the IUCN Red List in 2013 as Vulnerable.

Sclerocactus species are reported to be impacted by oil and gas exploration and extraction activities, recreational off-road vehicle (ORV) use, livestock trampling, collection of specimens, loss of habitat, and insect parasitism. Oil and gas development and ORV may increase access to plants by collectors.

There are concerns that *Sclerocactus* populations could be adversely affected by unauthorized and illegal harvest of plants and seeds. It is suggested that the harvest of even a small number of seeds or plants could adversely affect the species' reproductive potential and perhaps their long-term survival. As *Sclerocactus* species can be challenging to cultivate they may be of interest to a limited number of specialist collectors in the USA and elsewhere, but in general demand for them is likely to be low or very low.

All *Sclerocactus* species are protected by the U.S. Lacey Act, meaning, amongst other things, it is unlawful to import, export, transport, sell or receive any wild plant (including roots, seeds, and other parts) taken, possessed, transported, or sold in violation of any State law or regulation. In Arizona collectors must obtain a harvest permit and plants may not be moved from private property without contacting the Arizona Department of Agriculture. Collectors must obtain a permit to harvest and transport plants in Nevada. On land in Arizona and Nevada managed by the Bureau of Land Management (BLM), collection of *Sclerocactus* may be permitted only for scientific or educational purposes, or conservation or propagation of the species. Utah requires proof of ownership of the plant to collect and transport native plants within the State.

Sclerocactus blainei has previously been included in an assessment by IUCN of *S. spinosior* (classified as Least Concern, 2013) as a subspecies, but has not been assessed as a species in its own right[7]. S. cloverae has previously been included in the subspecies *S. whipplei heilii* as part of an assessment of *S. whipplei* by IUCN (classified as Least Concern, 2013), but *S. cloverae* has not been assessed in its own right[7].

Seeds of Appendix-II listed cacti (other than those from Mexico) are exempt from the provisions of the Convention under current annotation #4.

The proponents also propose amending the nomenclature of the Appendix-I listed *S. glaucus* but as this does not entail transfer or removal from the Appendices, or a change in any annotation, it has not been addressed here.

Analysis:

Sclerocactus blainei has a narrow distribution and is known from three occurrences. Its population size is unknown and no information is available on population trends. However, based on its apparently very restricted distribution, it may meet the biological criteria for inclusion in Appendix I. There is insufficient evidence to determine that the species is affected by trade.

Sclerocactus cloverae has a relatively widespread distribution and does not appear to have a small population. There is no indication that the species has undergone a marked decline. It would not appear to meet the biological criteria for inclusion in Appendix I. There is insufficient evidence to determine that the species is affected by trade.

Sclerocactus sileri has a relatively restricted distribution (although has recently been found to be more widespread than hitherto thought) and has reportedly undergone declines as a result of fire, although the severity of these declines is not known. Most known populations are reported to be small. The species may meet the biological criteria for inclusion in Appendix I. There is insufficient evidence to determine that the species is affected by trade.

Reviewers of summary information only: B. Goettsch.

References:

Information not referenced in the Summary section is from the Supporting Statement.

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[1] Hunt, D.R. (2016) CITES Cactaceae checklist. Third edition. Royal Botanic Gardens Kew & International Organisation for Succulent Plant Study. 174 pp.

[2] Zahradnictví Malej Jarda (2016) *Sclerocactus spinosior* ssp. *blainei* SB 1540. http://www.gerardo.cz/en/cacti-succulents-plantsseeds/623-sclerocactus-spinosior-ssp-blainei-sb-1540-currant-nv-10-seeds.html Viewed on 29th May 2016.

[3] NatureServe (2015) NatureServe Explorer: An online encyclopaedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. http://explorer.natureserve.org Viewed on 29th May 2016.

[4] Natureserve (2016) Conservation Status Assessment. http://www.natureserve.org/conservation-tools/conservation-status-assessment. Viewed on 29th May 2016.

[5] Nur kaffeemaschinen Hier finden Sie preiswerte *Sclerocactus* http://nur-kaffeemaschinen.de/shop/kat-29520/a-4/sclerocactus.html Viewed on 29th May 2016.

[6] Butterworth, C. & Porter, J.M. (2013) *Sclerocactus sileri.* The IUCN Red List of Threatened Species 2013.

[7] Goettsch, B. (2016) *In litt.* to IUCN/TRAFFIC Analyses Team, Cambridge, UK.

Delete annotation #5 to the listings of *Dalbergia cochinchinensis* and replace it with annotation #4

Proponents: Thailand

Summary: Siamese Rosewood *Dalbergia cochinchinensis* is a slow growing evergreen tree found sparsely in open semi-deciduous forests in Cambodia, Lao People's Democratic Republic (PDR), Thailand and southern Viet Nam. It is in demand internationally for its wood. It is included in the Chinese "Hongmu" standard of high-quality hardwoods used for furniture and cabinet-making. At CoP16 it was listed in Appendix II with annotation #5 to restrict the listing to logs, sawn wood and veneer sheets.

Harvesting of this species is either restricted (Viet Nam) or banned (Cambodia, Lao PDR, Thailand) within its range. A recent review of the trade in *D. cochinchinensis* indicates that a significant portion of the trade in this and similar species is currently in the form of secondary processed products, particularly furniture. By crudely processing timber in the source country to produce furniture it is possible to circumvent the current annotation #5. Since the Appendix-II listing, large shipments of timber believed to have been illegally harvested and exported in this way have been intercepted.

The proponents seek to amend the current listing with annotation #4 to include all parts and derivatives, except seeds and seedlings or tissue cultures obtained in vitro, in solid or liquid media, transported in sterile containers, and cut flowers of artificially propagated plants.

The intention of using this annotation is in order to regulate the products in trade that are of conservation concern.

Analysis: International trade in *Dalbergia cochinchinensis* appears to include products not included in the current annotation to the listing. This has been demonstrated by the interception of shipments of crudely processed timber exported as furniture. Annotation #4 would include all timber-related products including finished furniture, which are those of evident conservation concern.

Prop. 55 seeks to include all species of *Dalbergia*, except those in Appendix I, in Appendix II with no annotation. Depending on the order in which proposals are taken, if Proposal 53 is considered before Proposal 55, *D. cochinchinensis* would be included in the genus-level listing with no annotation if Proposal 55 were to be accepted, meaning that all readily recognisable parts and derivatives would be covered by the listing. There would be little practical difference between the two listings.

Inclusion in Appendix II of 13 timber species of the Genus *Dalbergia* native to Mexico and Central America without annotation: *Dalbergia calderonii*; *D. calycina, D. congestiflora, D. cubilquitzensis, D. glomerata, D. longepedunculata, D. luteola*, *D. melanocardium, D. modesta, D. palo-escrito, D. rhachiflexa, D. ruddae, D. tucurensis*

Proponent: Mexico

Note: *The entire genus Dalbergia apart from those already included in Appendix I or II is the subject of Proposal 55. Discussion of the genus as a whole is included in the analysis of that proposal.*

Summary: There are 20 *Dalbergia* species found in Mexico, six of which are endemic. Of the total, 15 produce high quality timber; two are already listed in Appendix II (*D. retusa* and *D. stevensonii*), the remainder are proposed here for listing in Appendix II.

Timber produced by many species of *Dalbergia*, often known as 'rosewood', is valued for the beauty, durability and physical properties of the wood; it is consequently in demand in international trade (see analysis for Proposal 55). None of the 13 species currently proposed for listing is named in the National Hongmu Standard of 33 species[1], or in the Chinese Industrial Hardwood Standard[2].

There is little information on the populations and trade in most of the species. Regeneration of many *Dalbergia* species is considered slow[3]. Mexico has now carried out risk assessments for the populations of Mexico.

Dalbergia calderonii occurs in Mexico, Guatemala, El Salvador and Honduras. This species is rare and slow growing, and occurs in a region with high deforestation. Mexico considers its population endangered and El Salvador has assessed it as threatened.

Dalbergia calycina occurs in Belize, Costa Rica, El Salvador, Guatemala, Mexico and Nicaragua. No data are available on the volume of trade; exports from Guatemala (which listed its population in Appendix III) were reported in 2014[4]. Known to occur in a number of protected areas, there are no specific data relating to the population size of this taxon, however, it is considered to be rare in Nicaragua and assessed as threatened by Mexico. In 2012 IUCN classified the species as of Least Concern.

Dalbergia congestiflora is distributed in Mexico, Guatemala and El Salvador, it is currently considered endangered in Mexico, but according to the most recent assessment it now qualifies as subject to special protection.

Dalbergia cubilquitzensis occurs in Belize, Costa Rica, Guatemala, Honduras, Mexico and Nicaragua. Mexico considers this species endangered. No data are available on the volume of trade; exports from Guatemala (which has listed the species in Appendix III) were reported in 2014[4].

Dalbergia glomerata is reported to occur in Costa Rica, Guatemala, Honduras and Mexico, although Mexico considers that it is endemic and the populations elsewhere are *D. congestiflora*. It was listed in Appendix III by Guatemala in 2015 and, according to the CITES Trade Database, since then 42m^3 of sawn wood have been reported as exported from Honduras to Taiwan (Province of China). The species is harvested for timber; populations are believed to be declining as a result. Also believed to be affected by decline in area and quality of habitat as a result of conversion to agriculture. Road construction is making areas more accessible for logging[5]. Classified as in need of special protection in Mexico and as Vulnerable by IUCN (2012).

Dalbergia longepedunculata occurs in Honduras and Mexico; considered endangered by Mexico.

Dalbergia luteola occurs in Guatemala and Mexico; considered endangered in Mexico.

Dalbergia melanocardium occurs in El Salvador, Guatemala and Mexico; considered endangered in Mexico.

Dalbergia modesta endemic to Mexico where it is considered threatened.

Dalbergia palo-escrito is endemic to Mexico where it is considered threatened, this species is in high demand for the manufacture of classical guitars and is subject to selective logging[6].

Dalbergia rhachiflexa is endemic to Mexico where it is considered threatened.

Dalbergia ruddae occurs in Costa Rica and Mexico; considered threatened in Mexico.

Dalbergia tucurensis naturally occurs in Belize, El Salvador, Guatemala, Honduras, Mexico and Nicaragua; introduced into Costa Rica. Nicaragua listed the species in Appendix III in 2014 and Guatemala added its population in 2015; the CITES Trade Database records just over 29,000m^3 in trade, mainly from Nicaragua to East Asia. Considered endangered by Mexico.

The 13 species proposed for listing in Appendix II have timber that is similar to that of species already listed in Appendix II from the same geographical region. Enforcement of the current listing is difficult due to problems in species identification. Trade is often reported at genus level and enforcement officers do not have a quick and easy technique to identify to species level. Under laboratory conditions, there are identification tests that can be done to species level but they are both costly and complicated. There is also reported to be illegal trade in *Dalbergia* species in the region.

For a broader discussion of trade in *Dalbergia* species see analysis of Proposal 55.

With no annotation, all parts and derivatives, live or dead, would be regulated. Most current *Dalbergia* listings have annotations (#5 and #6) that variously include logs, sawn wood and veneer sheets and plywood. However Proposal 53 notes that in a review of the trade in *D. cochinchinensis* a large portion of the trade in "rosewood" species from eastern Asia is currently in the form of secondary processed products, particularly furniture. The traders can crudely process the timber in the source country and then export it as furniture to circumvent the control. That proposal is to expand the scope of the listing by switching to annotation #4.

Analysis: The species of *Dalbergia* proposed here are timber-producing species that share range States with two *Dalbergia* species that are already included in Appendix II. There is insufficient information to determine whether any of the species proposed here meets the criteria in Annex 2a of the Resolution.

At least some of the species are known to be in trade and have timber that is difficult to distinguish from the Appendix-II listed species. Trade in timber from *Dalbergia* species may be reported at genus level. This creates problems in the implementation of the existing Appendix-II listing. It would appear therefore that these species meet the criteria for inclusion in Appendix II in Annex 2b of *Res. Conf. 9.24 (Rev. CoP16)* (lookalike criteria). With no annotation proposed, all parts and derivatives, live or dead, would be regulated; under the current Appendix-II listings for *D. retusa* and *D. stevensonii* the only products included are logs, sawn wood and veneer sheets and plywood.

References:
Information not referenced in the Summary section is from the Supporting Statement.

[1] Wenbin, H. & Xiufang, S. (2013) Tropical Hardwood Flows in China: Case Studies of Rosewood and Okoumé. Forest Trends.
[2] Chinese Industrial Standard of Precious Dark Color Hardwood Furniture (QB / T 2385-2008).
[3] EIA (2013) Report on CoP16 Proposals: https://eia-international.org/wp-content/uploads/EIA-COP16-Briefing_Proposals1.pdf. Viewed on 3rd July 2016.
[4] CITES (2015) PC22.Doc 17.2. Report of the Working Group for Neotropical Tree Species.
[5] Groom, A. (2012). *Dalbergia glomerata.* The IUCN Red List of Threatened Species 2012.
[6] Chatham House (2012) Chatham House Workshop: Tackling the Trade in Illegal Precious Woods 23-24 April 2012 Background Paper 1: Precious Woods: Exploitation of the Finest Timber Prepared by TRAFFIC.

Inclusion of the Genus *Dalergia* in CITES Appendix II without annotation, with the exception of the species included in Appendix I

Proponents: Argentina, Brazil, Guatemala and Kenya

Summary: *Dalbergia* is a genus of trees, shrubs and lianas with a pan-tropical distribution in Africa, Asia and Central and South America, ranging in habitat from tropical rainforests to seasonally dry tropical to subtropical humid and dry forest, woodland and wooded grassland. There are currently around 300 accepted names according to the Plant List but there is still substantial taxonomic uncertainty within the genus. Currently one species *Dalbergia nigra* from Brazil is in Appendix I, *D. cochinchinensis* from Southeast Asia, *D. granadillo*, *D. retusa*, and *D. stevensonii,* from Mexico and Central America, and all Malagasy species in the genus (ca. 70) are included in Appendix II. A number of Central American populations of various species are in Appendix III. Thirteen Mexican and Central American species are subject to a separate proposal (Proposal 54) for inclusion in Appendix II.

Some species produce high quality timber, often known as “rosewood”, which commands high prices in trade and is used in construction, cabinet work, marquetry, inlay, furniture construction, musical instrument manufacture, tools and carvings. The term rosewood is imprecise, and used differently in different contexts. Not all timbers characterised as rosewood are *Dalbergia* (the name is also variously applied to species in the genera *Jacaranda*, *Guibourtia* (the subject of Proposal 56) and *Machaerium*), and not all *Dalbergia* species produce rosewood. Some valued *Dalbergia* timber is known as ebony or “blackwood”.

Much of the current demand for rosewoods is associated with the demand in China for “Hongmu” furniture. However, not all Hongmu timber is necessarily rosewood. A national Hongmu standard (SAQSIQ 2000) of 33 species was issued in 2000 to identify those species whose density, texture and colour meet the requirements set in the Chinese National Hongmu Standard for legal marketing purposes (see Annex 1)[1]. Under the Hongmu standard *D. odorifera* is classified as “scented rosewood”. Fifteen other species of *Dalbergia* are included in the standard but none is classified as rosewood (all rosewoods other than *D. odorifera* in the standard are species of *Pterocarpus*). Eight Hongmu *Dalbergia* are classified as “blackwood”: *D. cultrata; D. fusca; D. latifolia; D. louvelii* (CITES Appendix II[2])*; D. melanoxylon; D. nigra* (Appendix I)*; D. spruceana; D. stevensonii* (Appendix II). Seven are classified as “mahogany”: *D. bariensis; D. cearensis; D. cochinchinensis* (Appendix II)*; D. frutescens; D. granadillo* (Appendix II)*; D. retusa* (Appendix II); *D. oliveri*. There is also an Industrial Standard of Precious Dark Color Hardwood Furniture in China. This classifies an additional species of *Dalbergia* (*D. greveana* (Appendix II[2])) as “Rosewood”.

Other *Dalbergia* species are also used for their hard wood. These include (but are not restricted to): Africa: some *Dalbergia* species from Madagascar; Latin America: *D. brasiliensis, D. cearensis, D. cubilquitzensis, D. cuscatlanica, D. decipularis, D. foliolosa*, *D. funera*, *D. glomerata, D. hortensis, D. miscolobium, D. spruceana, D. villosa, D. tucurensis, D. glabra, D. calycina.* Asia*: D. annamensis, D. cambodiana, D. mammosa, D. sissoo*[3]*, D. tonkinensis.* Various lists of commercial timber species exist that include *Dalbergias* (see “A Working List of Commercial Timber Tree Species”[4] although note that some species mentioned in the SS and here are not included in this); not all these species necessarily produce timber that resembles that of species already listed in the CITES Appendices.

Some *Dalbergia* species are used for making musical instruments. In particular the African Blackwood *D. melanoxylon* is the most highly-favoured wood for clarinets and oboes. Other species known for their musical qualities include *D. cochinchinensis* (Appendix II)*, D. glomerata, D. granadillo* (Appendix II)*, D. palo-escrito, D. retusa* (Appendix II)*, D. stevensonii* (Appendix II)[5], *D. tucurensis* and a number of Madagascan species (Appendix II)[6]. Recorded export of *D. melanoxylon*, a species widespread in sub-Saharan Africa, takes place almost entirely from Mozambique and Tanzania. Demand for musical instrument manufacture has been estimated at 255m^3 per year of semi-processed billets, equivalent to perhaps 1500m^3 of roundwood.

Harvest of different species of *Dalbergia* and similar timbers appears to follow a distinctive pattern in which as the most favoured and accessible timber stocks in a particular area are depleted, attention turns to others. As an example, with the commercial extinction of *D. odorifera* in China and *Pterocarpus santalinus* in India, the trade in *D. cochinchinensis* reportedly grew rapidly and it became the most sought-after Hongmu species globally. As *D. cochinchinensis* has subsequently been depleted the main species now dominating the Hongmu trade in Southeast Asia are reported to be *D. oliveri, D. bariensis, P. macrocarpus* and *P. pedatus*[7]*.*

There is generally very little quantitative information on the impact of logging on populations of *Dalbergia* species. Knowledge of the status of many of them is very limited and often out-of-date. In the 1998 IUCN Threatened Trees of the World, the following species were identified as under threat from overexploitation: *D. annamensis, D. bariensis, D. cambodiana, D. mammosa, D. oliveri*, *D. latifolia*, *D. odorifera, D. tonkinensis*. Of these *D. bariensis, D. latifolia, D. odorifera* and *D. oliveri* are classified as Hongmu species.

Dalbergia bariensis is native to Cambodia; Lao People's Democratic Republic (PDR); Thailand; Viet Nam where it is said to be widely distributed and scattered. At the time of the IUCN assessment (1998) there was said to be a rapid decline in the number of large trees because of overexploitation[8]. Millet and Truong (2011) recorded *D. bariensis* in Tan Phu forest in southern Viet Nam but noted that it was rare, showed limited regeneration and was "close to extinction"[9]. *D. latifolia* occurs in India, Indonesia and Nepal. In the 1998 IUCN assessment the timber was said to be of high commercial value and wild subpopulations widely overexploited including from illegal felling. *D. odorifera* was reportedly only known in 1998 from stands of coppiced individuals on Hainan Island, China. *D. oliveri* has a restricted distribution in Myanmar, Thailand and Viet Nam. Myanmar reported the export of 9000 m^3 of sawnwood to ITTO between 2000 and 2003[9].

Some species of *Dalbergia* are widely cultivated both within and outside their native range, occurring in plantations and used in agroforestry systems. Some, such as *D. latifolia* and *D. sissoo* have been regarded as invasive species outside their natural range[10, 11]. Some are shrubs or lianas with no international commercial use (e.g. *D. monetaria*[12], *D. hostilis*).

Use and trade of non-timber producing *Dalbergia* has not been assessed for this analysis. There may be some species in trade where the products in trade do not resemble those of species already included in the Appendices or proposed as meeting the criteria for inclusion in Appendix II in their own right rather than as lookalike species. However, there are no indications of large-scale international trade in such products[13].

Of the non-*Dalbergia* Hongmu species, *P. santalinus* is listed in Appendix II and *P. erinaceus* is the subject of Proposal 57 to be included in Appendix II.

The wood of some *Dalbergia* species has a characteristic colour and texture. Many species have the same wood anatomy[14] making identification by eye or using traditional anatomical methods only possible to genus level, if at all. However, in combination with chemical methods, such as mass spectrometry, DNA sequencing and profiling, near infrared spectroscopy and stable isotope analysis identification can consistently identify and distinguish between species[15, 16, 17]. Inexpensive and accessible tools are not available to enforcement officers at this time.

The intention of the proposal is to include all parts and derivatives of the species, live or dead and therefore no annotation is proposed for inclusion with the listing.

Analysis: The genus *Dalbergia* is a large and widespread one, comprising plants of many different forms. Some species produce high quality and sought-after timber, some of which is traded as "rosewood".

There is little available information on the status of, or impacts of harvest for trade on, non-CITES listed species of *Dalbergia* that produce rosewood, although there are indications of decline in some species, notably in Asia and Central and South America. There is insufficient readily available information to determine whether any of these meets the criteria for inclusion in Appendix II in Annex 2a of *Res. Conf. 9.24 (Rev. CoP16)*.

However, given the difficulty in distinguishing between different rosewood-producing species of *Dalbergia* in the principal form in which they are traded (timber) it would appear that such species would meet the criteria for inclusion in Appendix II in Annex 2b (lookalike criteria) owing to the resemblance of their timber in trade to that of species already listed in the Appendices. Determining which species should be treated as lookalikes may require some additional work; various lists of Dalbergia species timber in trade exist but these would need to be analysed as to which rosewoods resemble each other.

One species of African *Dalbergia* African Blackwood (*D. melanoxylon*) produces timber that is in trade principally in a form (semi-processed billets for the production of musical instruments) that is reasonably easily distinguished from other *Dalbergia* spp. in trade and other timber species included in the Appendices. There is insufficient information to determine whether this species meets the criteria for inclusion in Appendix II in Annex 2a of *Res. Conf. 9.24 (Rev. CoP16)*. It does not appear to meet the criteria in Annex 2b. No mainland African species of *Dalbergia* is known to produce rosewood that is in trade.

Many *Dalbergia* species are not known to be in trade, nor do they resemble species that are in trade. These do not meet the criteria for inclusion in Appendix II.

No annotation is proposed with this listing which would result in all parts and derivatives being included, if adopted. Current listings are annotated to include “Logs, sawn wood and veneer sheets” (#5) and plywood for those with annotation (#6). Some of those species which are currently listed are used for the manufacture of musical instruments, although musical instruments are excluded from the listings. Species that would be listed were this proposal adopted would include musical instruments where they are used for this purpose. A proposal to amend the annotation for *D. cochinchinensis* (Proposal 53) intends to widen the scope of products covered to include secondary processed products, particularly furniture as it appears that traders are crudely processing timber in the source country and then exporting it as furniture to circumvent the control. A genus level listing with no annotation would include un-processed, semi-processed and finished furniture.

References:

Information not referenced in the Summary section is from the Supporting Statement.

[1] Forest Trends (2014) China’s Policies for Hongmu Import Surveillance & Control Zhang Yue http://www.forest-trends.org/documents/files/doc_4368.pdf Viewed on 2nd July 2016.
[2] Under the general Appendix II listing for Malagasy *Dalbergia* spp.
[3] FAO (year unknown) *Dalbergia sissoo* Roxb http://www.fao.org/ag/agp/AGPC/doc/gbase/data/pf000385.htm. Viewed 2nd July 2016.
[4] Mark, J., Newton, A.C., Oldfield, S. and Rivers, M. (2014) The International Timber Trade: A working list of commercial timber tree species. BGCI, Kew, UK. http://www.bgci.org/files/Global_Trees_Campaign/Timber_list/TimberWorkingList_v2DImage.pdf
[5] Global Trees Campaign (2016) http://globaltrees.org/threatened-trees/tree-values/musical-instruments/ Viewed 2nd July 2016.
[6] Jenkins, A., Bridgland, N., Hembery, R., Malessa, U., Hewitt, J. & Keong, C.H. (2012) Background Paper 1: Precious Woods: Exploitation of the Finest Timber. TRAFFIC. Chatman House Workshop. Tackling the Trade in Illegal Precious Woods. 23-24 April 2012, http://www.traffic.org/non-traffic-papers/ Viewed on 2nd July 2016.
[7] EIA (2016). The Hongmu Challenge: A briefing for the 66th meeting of the CITES Standing Committee, January 2016.
[8] Oldfield, S., Lusty, C. & MacKinven, A. (1998) Threatened Trees of the World, IUCN/ WCMC, Cambridge.
[9] UNEP-WCMC (2014) Non-CITES timber species from South Asia (Leguminosae) potentially warranting further protection. UNEP-WCMC, Cambridge, UK.
[10] Global Invasive Species Database (GISD) (2016) Species profile: Dalbergia sissoo. http://www.iucngisd.org/gisd/species.php?sc=1186 on 26-05-2016.
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Annex 1: 33 species listed in Chinese National Standard for Hongmu.

Category	Species
Red sandalwood	*Pterocarpus santalinus*
Rosewood	*Pterocarpus cambodianus, P. dalbergioides, P. erinaceus, P. indicus, P. macrocarpus, P. marsupium, P. pedatus*
Scented rosewood	*Dalbergia odorifera*
Blackwood	*Dalbergia cultrata, D. fusca, D. latifolia, D. louvelii, D. melanoxylon, D. nigra, D. spruceana, D. stevensonii*
Mahogany	*Dalbergia bariensis, D. cearensis, D. cochinchinensis, D. frutescens, D. granadillo, D. oliveri, D. retusa*
Ebene	*Diospyros ebenum, D. crassiflora, D. pilosanthera, D. poncei*
Ebony	*Diospyros celebica, D. philippensis*
Wenge	*Millettia laurentii, M. leucantha, Cassia siamea*

Inclusion of *Guibourtia demeusei*, *G. pellegriniana* and *G. tessmannii* in Appendix II with annotation #4

Proponents: Gabon and European Union

Summary: The genus *Guibourtia* is currently considered to comprise between 14 and 16 species[1,2]; 13 occur in tropical Africa, and one in the Neotropics[1]. All three species in the current proposal are African forest trees.

Guibourtia tessmannii grows to 65m with a trunk that can exceptionally reach 2m in diameter at breast height (DBH) but is generally smaller. It is found at very low population density on firm ground in evergreen forest in Cameroon, Equatorial Guinea and Gabon. It is also believed likely to occur in southeast Nigeria, the Republic of Congo and extreme southwest Central African Republic, being present in logging concessions in Cameroon near these countries. It has not been confirmed in the Democratic Republic of the Congo[3].

Guibourtia pellegriniana grows to 30m, with a trunk typically around 40cm DBH. It also occurs at very low population density. Its known distribution is considerably smaller than that of *G. tessmannii*. Herbarium specimens originate from a narrow strip of coastal forests in Cameroon, Equatorial Guinea, Gabon and possibly the Republic of Congo. Recent work indicates it may be more widespread than this, also occurring in forests further inland where *G. tessmannii* is found, although at much lower density.

Guibourtia demeusei grows 25 to 40m tall with a trunk up to one metre DBH. It occurs in periodically flooded and swampy forest and gallery forest, often in pure stands[4]. It has a much larger range than the other two species, extending into the central Congo basin. It occurs in Cameroon, Central African Republic, Democratic Republic of the Congo, Equatorial Guinea, Gabon, and the Republic of Congo[5].

There is a general lack of biological data for these species, which are all from closed forest and have historically been problematic to study[6]. *G. tessmannii* and *G. pellegriniana* are said to be difficult to distinguish from each other in the field. Increment rate (increase in trunk diameter) of ca. 0.35cm per year has been estimated for *G. tessmannii* in Gabon and of ca. 0.4cm per year for managed populations of *G. demeusei* in the Republic of Congo[7]. Frugivorous animals are thought to play an important part in seed dispersal.

Some inventory data are available. Historic estimates for Gabon (of a probable stock of three to seven million m^3 of *G. tessmannii* and *G. pellegriniana* combined made in 1975 and of seven to 13 million m^3 made in 1995) are not considered reliable. More recent assessment in forestry concessions in Gabon found extremely low stocks of the two species combined that could be harvested on a sustainable basis (annual harvests from effectively zero to $0.0045m^3$ per ha). Inventories in Cameroon have found similarly low stocking densities, between 0.002 and 0.06 trees with DBH >20cm per ha. A considerably higher density of *G. demeusei*, of 0.4 stems per ha, has been reported in the Central African Republic[5].

The timber of all three species is commonly used locally and has a high socio-cultural value[8]. *G. tessmannii* and *G. pellegriniana* in particular are reportedly highly sought-after as timber for furniture-making within their range, although there is no information on quantities used.

All three species are traded internationally as Bubinga but are also known under other names, such as Kevazingo. *G. tessmannii* and *G. pellegriniana* are commonly referred to as Rose Bubinga and are reportedly indistinguishable in trade. *G. demeusei,* or Red Bubinga, can be distinguished and is generally considered of inferior quality, but reportedly may be easily confused with or substituted for that of the other two on the international market. Historically most exports were to Europe; more recently China has become the main market as the timber is used for the making of Hongmu-type (rosewood and blackwood or ebony) furniture, demand for which has increased greatly in the past decade. *Guibourtia* species are not a part of the recognised Hongmu standard in China, but their timber is a category A2 hardwood that is used as a substitute for Hongmu timbers[9].

Trade data, often reported under trade (i.e. non-scientific) names, may refer to one or other of the three species, or some combination of them. Reported exports of *G. tessmannii* and *G. pellegriniana* combined from Gabon have increased over time. Until 2009 all recorded export was of roundwood (logs), rising from an average annual export of ca. $25{,}000m^3$ in 1987 to 1992, to ca. $60{,}000m^3$ in 1993 to 1999 and just under $70{,}000m^3$ in 2000 to 2009. From 2011 onwards, only sawnwood has been officially recorded as exported.

Using a conversion factor from sawnwood to roundwood of three, on average the equivalent of 65,000m^3 roundwood was exported annually in 2011 to 2014.

Recorded exports from Cameroon do not distinguish between the three species. Exports are at a much lower level than those for Gabon, amounting for ca. 13,000m^3 roundwood per year in 1995 to 1998 and roundwood equivalent of around 6000m^3 in sawnwood per year for 1999 to 2014 with little discernible overall trend. Data from logging requests submitted by forest management units indicates that during the period 2008 to 2012, around 75% of logged volume of *Guibourtia* in Cameroon was of *G. demeusei*, with volumes requested for this species for 2011 to 2013 considerably higher than those requested in previous years. At the same time requested volume of *G. tessmannii* (probably including *G. pellegriniana*) halved.

Recorded exports of Bubinga from Equatorial Guinea are at a low level, although have risen from virtually zero in 2007 to ca. 400 m^3 in 2011[5]. The species involved are not identified, nor is it clear if the volume is for roundwood or sawnwood.

Bubinga exported from Democratic Republic of the Congo is *G. demeusei* (the other two species do not occur there); it is likely that all Bubinga exported from Central African Republic (where *G. demeusei* is known to occur and the other two have not been confirmed) is also *G. demeusei*. Reported exports from the Central African Republic rose sharply from zero or nearly zero in 2005 to 2009 to ca. 600m^3 in 2010 and 1700m^3 in 2011 (again, it is unclear if this is roundwood or sawnwood). Recorded exports from the Democratic Republic of the Congo also increased, from very low levels in 2005 to 2008 to ca. 700m^3 in 2009 and 2000m^3 in each of 2010 and 2011.

Roundwood of *G. tessmannii* and *G. pellegriniana* is advertised on the internet, indicating that it is available on the international market despite the fact that the two countries known to export these species prohibit export of roundwood. It is suspected that declared exports represent only a proportion of the actual volume exported, although exactly what proportion remains unknown. There are narrative accounts of extensive illegal and unauthorised logging of these species in Cameroon[5].

Current low population densities of *G. tessmannii* and *G. pellegriniana* have been ascribed to past exploitation[10]. However, there is an absence of baseline information on which to assess the effects of exploitation.

Minimum felling diameters have been set in some range States. The export of Bubinga logs has been prohibited in Cameroon since 1999 and Gabon since 2010. In November 2012 Cameroon suspended its harvest of *G. tessmannii* in the national forest domain.

Guibourtia ehie and *G. arnoldiana* are distributed in the same region as the proposed three species, and are also in international trade but are easily distinguished due to their brown wood.

The listing of these species is proposed with an annotation (#4) that would include all parts and derivatives, except seeds, seedlings or tissue cultures obtained in vitro and cut flowers of artificially propagated plants.

Analysis: Information on the status of *Guibourtia tessmannii*, *G. pellegriniana* and *G. demeusei* is sparse. There is very little information on recruitment rates or age and size at maturity. The species, particularly *G. tessmannii* and *G. pellegriniana,* are known to be in demand internationally for their rosewood-type timber, the market for which has grown very rapidly in Asia, particularly China, in recent years.

Populations of *G. tessmannii* and *G. pellegriniana* are of low density, although it is not known if this is a natural state or a result of past exploitation. Where there is information on trade in these species it appears to be at a low level. There are indications of illegal offtake and trade, the volume of which is not quantified but which may be relatively high. Given the evident scarcity of harvestable-sized *G. tessmannii* and *G. pellegriniana* it is likely that current harvest, including illegal offtake, for export is exceeding the rate at which such trees are entering the population, leading to probable commercial extinction of these species. It is unclear, however, whether this will lead to the species themselves becoming threatened by harvest or other influences, or becoming eligible for inclusion in Appendix I in the near future

G. demeusei is a widespread species that can be locally abundant. Reported harvest and export in a number of its range States increased around 2009 and 2010, which may be associated both with increasing demand for rosewoods in general at that time, and declining availability of *G. tessmannii* and *G. pellegriniana*. However, recorded harvest and export remain at a relatively low level, indicating that the species is unlikely to meet the criteria for inclusion in Appendix II in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)*.

Given the difficulties in distinguishing between *G. tessmannii* and *G. pellegriniana*, if either were to be included in Appendix II, then the other would meet the criteria in Annex 2b A of the Resolution (lookalike criteria). Information regarding the similarity of these two species to *G. demeusei* is conflicting. By some accounts the principal part in trade (timber) is relatively straightforward to distinguish, although all three may be traded under the same generic trade name. It is unclear, therefore, whether *G. demeusei* does meet the criteria in Annex 2b A of the Resolution.

Reviewers of summary information only: D. Mahonghol and T. Osborn.

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Information not referenced in the Summary section is from the Supporting Statement.

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Inclusion of *Pterocarpus erinaceus* in Appendix II, without annotation

Proponents: Benin, Burkina Faso, Chad, Côte d'Ivoire, European Union, Guinea, Guinea-Bissau, Mali, Nigeria, Senegal and Togo

Summary: *Pterocarpus erinaceus* is a slow growing, medium sized, generally deciduous tree found in open forest and wooded savannah in Sub-Sahelian West Africa, with a broad distribution including Burkina Faso, Cameroon, Côte d'Ivoire, Gambia, Ghana, Guinea, Guinea-Bissau, Mali, Niger, Nigeria, Senegal, Togo and possibly Central African Republic, Chad, Liberia and Sierra Leone. It is a pioneer species, readily colonising fallow land, and is drought tolerant once established. It regenerates quickly after coppicing and is reasonably resistant to fire, usually surviving the yearly savannah bush fires. The species can mature at around 5cm stem diameter[1]. It is an important species ecologically due to its atmospheric nitrogen fixing properties which may improve soil fertility.

Population studies undertaken show that average tree density varies widely: in Burkina Faso, Niger and Togo it ranges from 1.17 ± 0.75 trees/ha to 110.9 ± 1.15 trees/ha[2 3], with average stem diameter ranging from ca. 25cm to ca. 50cm. It has been observed that many mature trees remain in Sierra Leone: in the 100km^2 around Lake Sonfon in Sierra Leone there are an estimated total of 500,000 - 1 million individual trees[1], and trees here are of a larger average size than those observed in Burkina Faso[4].

The species is of high socio-cultural importance in the region and is widely used locally in construction and furniture making, as medicine, for musical instruments, charcoal, dyes and fodder for livestock.

The species has been heavily exploited for international trade in recent years with nearly all recorded trade going to China. It is amongst the species classified under China's Hongmu Standard, a list of 33 species, including *Pterocarpus* spp., *Dalbergia* spp., *Diospyros* spp., *Millettia* spp. and *Cassia* spp., whose density, texture and colour match the requirements for the manufacture of luxury Hongmu furniture. Recorded Chinese imports of logs increased from ca. 3000 m^3 in 2009 to 700,000m^3 in 2014[5]. A typical yield of 0.8m^3 is estimated for a relatively large (50cm dbh) tree[6], so that reported imports to China in 2014 would have required the harvesting of nearly 900,000 large trees.

The species has been protected under forest law in most range States, in some cases since 1996, due to concerns about failing population management and unsustainable use. There are total export bans in at least seven range States. Recommended average felling diameter ranges from 35-*65cm*. However, there is little evidence of forest management plans in place for this species, or of effective controls regulating national use or international trade. In some countries only specimens of ca. 30 cm stem diameter or over are being logged, but in others, for instance Benin, Burkina Faso, Ghana and Côte d'Ivoire trees of a smaller stem diameter are also targeted[7]. Logs of this species are widely available on the internet, in any quantity requested, and shipped from ports in countries with export bans in place.

In order to ensure the listing covers those parts of the species that first enter or dominate the international market (and as listings with annotations may be circumvented, for instance #5 by minimal working of the wood prior to export) the proposal is without annotation.

Analysis: *Pterocarpus erinaceus* is a tree species which it is harvested for timber and has a number of other local uses. There is evidence of rapid increase in export of timber from range States in the past six years largely to meet demand in China for Hongmu timber used in furniture-making. A proportion of this export, possibly the majority, appears to be unauthorised or illegal. The species is widespread and adaptable and may be at least locally abundant. It may also mature at a size considerably smaller than that at which it is harvested for timber. The current level of harvest for timber is likely to be unsustainable, in that it almost certainly exceeds the rate at which harvestable-sized trees are being replenished in the population, but it seems unlikely that regulation of trade is required to prevent the species from becoming eligible for inclusion in Appendix I in the near future, or that regulation of trade is required to ensure that harvest of specimens is not reducing the population to a level at which its survival might be threatened.

Reviewers of summary information only: C. Duvall, C. Hin Keong and S. Oldfield.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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Inclusion of Grandidier's Baobab *Adansonia grandidieri* in Appendix II with an annotation limiting the parts and derivatives to seeds, fruits, oils and living plants

Proponent: Madagascar

Summary: Grandidier's Baobab *Adansonia grandidieri*, one of six endemic *Adansonia* in Madagascar, is a large deciduous tree that occurs in west and southwest Madagascar. Recent studies based on analysis of satellite images and field observations[1,2], have found it to have a relatively extensive distribution (26,000 to 32,000 km^2) along the Mangoky River and in the western part of the Menabe region. Populations of the tree are scattered through this area. The satellite image study estimated its population at over one million individuals, many more than has previously been suggested.

Regeneration levels of *A. grandidieri* appear to be low, with few young trees in the population. A number of potential causes have been proposed, including reductions or loss of populations of natural seed dispersal agents (wild animals that eat the fruit without digesting the seeds), an increase in the population of feral pigs and cattle, increasing human harvesting of fruits and bark and changing land use through the conversion to agricultural land and pasture. Forest cover has declined considerably in the region in the past few decades and deforestation rates are said to have increased markedly in the past decade.

The species has established local uses for fruit, seed oil, tree bark, bark fibre and wood. Local demand for juice preparation in hotels in Morondava (one of the main towns in the region) is roughly estimated at 3000kg of fruits per year. Local harvest of trees for fibres in Morondava market is estimated at the equivalent of 50 trees harvested[6]. The amount of national level trade is not known but seed mounds (equivalent to ca. 2500kg) have been observed in the market in Antananarivo (the capital, not in the area of distribution of the baobab) during the fruiting season[6].

In Madagascar, several companies promote the commercialization and sustainable use of *A. grandidieri,* including through the production and marketing of baobab powder and baobab oil derived from fruit and seed[3]. The actual extent of this trade is unclear. A collection permit was granted to a company in Madagascar for 4000kg of *A. grandidieri* fruit for use in food and cosmetics; the full amount was reported as harvested in 2015[4]. Harvested fruit did not appear to be intended for international trade.

There is very little evidence of international trade in any products of *A. grandidieri.* Records from the Government of Madagascar indicate export of 150ml of seed oil (equivalent to 15 fruits) in 2014 and 35kg, apparently of seed oil (possibly seed), in 2015. Health products are advertised on online platforms as containing *A. grandidieri*[5], but it is not known if these do in fact contain *A. grandidieri* extract. Seeds for horticulture are advertised internationally and there is some local purchase of seeds by tourists, presumably intended for export[6]. The numbers involved are likely to be very small compared with national use.

The fruits and seeds (processed into powder and oil respectively) of another species of baobab, *A. digitata,* a species native to and widespread in Africa and southwest Arabia (and widely introduced elsewhere) have featured increasingly in international trade for use in pharmaceutical, cosmetics and food products[7]. *A. digiata* is registered by the US Food and Drug Administration as Generally Regarded as Safe (GRAS), which allows *A. digiata* to be used as an ingredient in food products. A similar classification (as Novel Food) for *A. digitata* exists in the European Union. There may be concerns that the future registration of *A. grandidieri* on the European and the USA export markets will lead to an increase in demand for fruit and seeds.

Local harvest comes under general regulations for non-timber forest products[8]. To date, there are no specific government regulations for *A. grandidieri.* The species is considered to have good representation in protected areas[1], where harvest is prohibited.

On-the-ground implementation projects are under way to establish sustainable production quantities of *A. grandidieri*; an experimental sustainable fruit offtake harvest has been piloted on a commercial basis for a few years[4].

The proposed annotation is to include seeds, fruits, oils and live plants. Both oils and products containing oils have been recorded in trade. It is unclear whether the use of term 'oil' in the proposal annotation is intended to include finished products, for example cosmetics.

The species was assessed as Endangered against the IUCN Red List criteria (1998); an updated assessment has been made but not yet published.

Analysis: *Adansonia grandidieri* is an endemic species to Madagascar where its population is reportedly still numerous, although affected by a range of factors including low regeneration rates and conversion of its habitat. There is harvest for domestic use. Some of this (principally for fibres) is destructive, but the major products harvested are fruit and seeds, harvested non-destructively. Such harvest might conceivably have some impact on regeneration rates in areas where it takes place, although there are no data to support this. Recorded levels of international trade (in seeds and seed products) compared with observed levels of domestic use are very small and appear highly unlikely to have an impact on the wild population. The species would not appear to meet the criteria for inclusion in Appendix II in *Res. Conf. 9.24 (Rev. CoP16)*.

The proposed annotation is to include seeds, fruits, oils and live plants. Both oils and products containing oils have been recorded in trade. It is unclear whether the use of the term 'oil' in the proposal annotation is intended to include finished products, for example cosmetics.

Reviewers of summary information only: V. Jeannoda, H. Ravaomanalina, D. Mayne, S. Andriambololonera, S. E. Rakotoarisoa, S. Wohlhauser and E. Creuse.

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Inclusion of Algerian Fir *Abies numidica* in Appendix I

Proponent: Algeria

Summary: *Abies numidica*, the Algerian Fir, is an evergreen coniferous tree which grows to a height of 20-35m[1]. It is native to Algeria where it occurs only at 1800-2000m elevation on Djebel Babor, part of the Petite Kabylie Mountain Range. The total extent of forests containing the species is estimated at less than 30km^2 with the fir only occupying a small portion of this. Access to the area is highly restricted because of security problems and there are no recent population estimates. A 2011 report stated that the number of trees had halved since the 1950s, although the basis of this is not known[2].

The area is a Nature Reserve to which entry is controlled; wardens try to prevent timber extraction, hunting and livestock grazing although these activities apparently still persist, with grazing in particular said to seriously affect plant regeneration. Fires are also reportedly a hazard[2].

A number of specific conservation proposals have been suggested and relevant authorities are said to be very supportive of action to protect the site, but no more recent information is available regarding its current status and management[1].

Algerian Fir is not exploited for timber[3] but it is grown as an ornamental tree in parks and larger gardens, being valued among firs for its drought tolerance and attractive appearance[4]. However it is sensitive to low temperatures and to air pollution in urban environments[3]. It is mostly cultivated in countries around the Mediterranean Sea, where it is sometimes planted in hedges as it takes trimming well[3]. Few cultivars are known. The species hybridises readily with other *Abies* spp., so that seed collected from cultivated trees is often hybrid[3]. As a result the species has reportedly mostly been grown from seed collected *in situ*, although some nurseries use grafting techniques as a method of propagation[5]. There are no indications of wild collection of plants, nor is it known if seed is collected from wild plants at present.

This species is reported to be present in 72 botanic gardens[6]. Availability of the species on-line appears to be very limited.

This species is classified in the IUCN Red List as Critically Endangered (2011).

Analysis: The species has a restricted range and has a population which is apparently declining, so that it appears to meet the biological criteria for listing in Appendix I. If trade from the wild population does occur, which is not known, it is almost certainly in seeds. Unless very substantial quantities were collected, or harvest were destructive (through felling of the trees) such trade would be highly unlikely to have an effect on the wild population. It is not clear therefore whether the species meets the criteria for inclusion in Appendix I.

A listing with no annotation would mean all parts and products were included.

References:
Information not referenced in the Summary section is from the Supporting Statement.

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Amendment of the listings of *Aquilaria* spp. and *Gyrinops* spp. in Appendix II

Amend Annotation #14 with the underlined text:
"All parts and derivatives except:
a) seeds and pollen;
b) seedling or tissue cultures obtained in vitro, in solid or liquid media, transported in sterile containers;
c) fruits;
d) leaves;
e) exhausted agarwood powder, including compressed powder in all shapes; and
f) finished products packaged and ready for retail trade, this exemption does not apply to <u>wood chips</u>, beads, prayer beads and carvings."

Proponent: United States of America

Summary: *Aquilaria* and *Gyrinops* are two genera of trees in the family Thymelaeaceae, distributed from India to New Guinea. The CITES Checklist currently recognises some 25 species of *Aquilaria* and eight of *Gyrinops*. In some trees, a still imprecisely understood combination of wounding, vectors of infection (bacterial infection, fungus) and resinous response induces the formation of a resinous heartwood (agarwood) that is fragrant and highly valued. The primary source of agarwood in reported trade is *Aquilaria malaccensis*. Agarwood is used in perfumes, incense and traditional medicines, and as an essential oil, distilled from the wood. Carvings and beads, including prayer beads, are also produced from the wood. So-called exhausted wood powder – the residue left after the distillation process – is often compressed to make incense sticks and small statues.

All agarwood-producing taxa are currently included in Appendix II; *Aquilaria malaccensis* was listed in 1994, and the rest of the genus *Aquilaria* and all *Gyrinops* spp. in 2004. They are currently covered by annotation #14, agreed at CoP16 (Bangkok, 2013), including all parts and derivatives except:

a) seeds and pollen;
b) seedling or tissue cultures obtained *in vitro*, in solid or liquid media, transported in sterile containers;
c) fruits;
d) leaves;
e) exhausted agarwood powder, including compressed powder in all shapes; and
f) finished products packaged and ready for retail trade, this exemption does not apply to beads, prayer beads and carvings.

International agarwood trade is complex, as it is traded in a variety of forms and at various stages of processing, from raw whole pieces to finished products such as perfumes, which may contain only small amounts of agarwood oil. Some processing of agarwood to produce end-products takes place in range States; some takes place elsewhere with resulting products, either sold domestically or re-exported to other consumer countries.

One of the major products in trade is wood chips, which may be traded for burning as 'incense wood', or for further processing to produce products such as beads, prayer beads, medicines, incense sticks, perfumes and tea. Such chips are exported in large quantities from range States and therefore adhere to the recommendations for inclusion in Appendix II set out in *Res. Conf. 11.21 (Rev CoP16*), which state that commodities listed should be those that dominate the trade and the demand for the wild resource.

It is not possible to distinguish wood chips destined for further processing from those intended to be used as an end-product. Because of this, concern has been expressed that substantial quantities of wood chips intended for further processing could be entering trade ostensibly as finished products packaged and ready for the retail trade. Under the current listing, wood chips that appear to be packaged and ready for retail trade are not covered by the Convention.

Consultations undertaken by the Standing Committee Working Group on Annotations indicated that there was variability in how trade in agarwood chips packaged for retail trade was regulated, with some such trade taking place with CITES permits, even though this was not currently required.

Conversely, in some cases it seemed as if such chips had been confiscated because they lacked CITES documentation, even though such documentation was not required.

Information from the CITES Trade Database confirms the importance of wood chips as a product in trade (ca. 7000mt of *Aquilaria* reported as exported and 10,000mt reported as imported between 2005 and 2015; ca. 180mt of *Gyrinops* reported as exported and 230mt reported as imported in the same period), and also indicates numerous transactions of less than 5kg, reported as grammes or kilogrammes. Reported small transactions such as these account for only a very small proportion of overall trade in wood chips (ca. 500kg in total, or 0.01% by weight of trade in *Aquilaria* as reported by exporters, and just 5kg of *Gyrinops* in total for the period 2005 to 2105). The amount of trade in wood chips packaged and ready for retail trade currently unregulated by CITES is unknown.

Analysis: Wood chips are a major trade item for agarwood, included in Appendix II as *Aquilaria* spp. and *Gyrinops* spp. The current annotation for agarwood exempts wood chips that are packaged and ready for retail trade from CITES controls. This exemption is reported to be inconsistently applied. Removal of the exemption, as proposed here, would ensure that all agarwood chips, however packaged, were subject to CITES controls (apart from those exempt under personal effects as specified in *Res. Conf. 13.7 (Rev CoP16)*), thereby bringing more of the agarwood trade under CITES control and, in theory, simplifying implementation.

Inclusion of Natal Ginger *Siphonochilus aethiopicus* (populations of Mozambique, South Africa, Swaziland and Zimbabwe) in Appendix II

Proponent: South Africa

Summary: *Siphonochilus aethiopicus,* the Natal Ginger or Wild Ginger, is a long-lived plant that grows in seasonally dry woodlands with a perennial rhizome and annual above-ground parts that die off during the dry season. It is widespread in tropical and sub-tropical Africa, occurring in 24 range States. The proposal only concerns populations of Mozambique, South Africa, Swaziland and Zimbabwe.

Although believed to be affected by habitat loss, large-scale commercial harvesting to supply the herbal medicine trade in southern Africa is considered to be the most important factor affecting the species, which is one of the most popular ingredients in traditional medicines, particularly in South Africa. Harvest for local medicinal use has been implicated in declines in South Africa, where it is now extinct over much of its former range, its extent of occurrence having reportedly declined by more than 90% over the last 100 years, now standing at just over 8000km^2 [1]. Thirty-nine known historical subpopulations were identified in South Africa in 2000, of which only 17 were still extant. More than half had fewer than 100 individuals, although some had up to 4000 plants. The species is now extinct in KwaZulu-Natal and has declined drastically in Limpopo and Mpumalanga Provinces. The majority of the remaining populations are reportedly not secure. Two-thirds occur outside formal conservation areas and three of the six populations that are theoretically protected are reportedly still being heavily exploited.

Historically South Africa exported the plant (e.g. to Lesotho in the early 20th century). The direction of trade appears to have reversed. Demand is South Africa is apparently too high to be met by current production from cultivated sources or locally sourced wild plants[2]. Wild plants are reportedly being imported in increasing quantities from neighbouring countries. One reported market observation in 2011 revealed thousands of plants said to have been harvested in Zimbabwe and there are accounts of people travelling to Zimbabwe from South Africa to harvest. Plants are also reportedly imported into South Africa and possibly Swaziland from Mozambique. Because any trade that occurs is part of the informal economic sector it is difficult to assess its volume.

Reports from Mozambique from 1987 and 2010 indicated it to be locally abundant in clumps in miombo woodland. Healthy populations apparently still exist in northern Mozambique, although it is suspected that some of those in the south may be depleted. In Swaziland, remnant wild populations are not effectively protected in protected areas and there is information on ongoing harvest of the species in at least one nature reserve. There is no information available on population status or trends in Zimbabwe.

Outside the four range States that are the subject of the proposal, there are reports of numerous *S. aethiopicus* populations in West Africa.

There is also some evidence of international trade through international online trade platforms, including from South Africa and from Australia (of *S. aethiopicus* of South African origin).

In South Africa, *S. aethiopicus* is listed as an endangered species in the Threatened or Protected Species (TOPS) Regulation list. Permits are required for harvesting, possession and trade. In 2015, South Africa has published (in draft)[3] its intention to revise the TOPS listing of *S. aethiopicus* to a critically endangered species of medicinal plants, which will restrict, through permitting, the import into South Africa of wild-sourced material of the species, as well as domestic trade within the country. Under the listing, all artificially propagated plants and their export would be exempt from controls[3].

In Swaziland, the Flora Protection Act lists *S. aethiopicus* as a specially protected flora species requiring permits for harvest or export[4].

Siphonochilus aethiopicus has not been assessed against the IUCN Red List criteria, but was assessed as critically endangered in South Africa, as endangered in Swaziland, and was reported to be endangered in Benin.

Analysis: This proposal is limited to *Siphonochilus aethiopicus* populations in Mozambique, South Africa, Swaziland and Zimbabwe. The populations in South Africa have evidently been seriously depleted by harvesting for domestic demand, and there are indications that harvesting for import into South Africa has spread to the other three range States in the proposal. Import is believed to take place through informal channels. Populations have been described as ‘remnant’ in Swaziland. Healthy populations reportedly exist in northern Mozambique; there are suspicions of depletion in the south of the country. There is no information available on its status in Zimbabwe. On this basis there is insufficient evidence to determine whether the species meets the criteria in Annex 2 a of *Res. Conf. 9.24 (Rev. CoP16)*.

Reviewers of summary information only: D. Newton, V. Williams, N. Crouch and G. Nichols.

References:

Information not referenced in the Summary section is from the Supporting Statement.

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[1] Williams and Crouch unpublished, obtained from Williams. V, (2016) *In litt.* to IUCN/TRAFFIC Analyses Team, Cambridge, UK.

[2] Crouch, N (2016) *In litt.* to IUCN/TRAFFIC Analyses Team, Cambridge, UK.

[3] South Africa Department of Environmental Affairs (2015) National Environmental Management: Biodiversity Act (10/2004): Threatened or protected species regulations. do.: Publication of lists of species that are threatened or protected, activities that are prohibited and exemption from restriction. Government Gazette, Notice 255 of 2015, No. 38600.

[4] Minister for Agriculture and Cooperatives (2000) The Flora Protection Act, Schedule A, Especially protected flora (Endangered). Mbabane.

Amendment of the listing of *Bulnesia sarmientoi* in Appendix II

Amend Annotation #11with underlined text: Logs, sawn wood, veneer sheets, plywood, powder and extracts. <u>Finished products containing such extracts as ingredients, including fragrances, are not considered to be covered by this annotation</u>

Proponent: United States of America

Summary: *Bulnesia sarmientoi* is a tree species occurring in the Plurinational State of Bolivia, Paraguay, Argentina and a small part of Brazil. It was included in Appendix II in 2010. The wood of *B. sarmientoi* is heavy, very strong and decay-resistant, even underground, because of its resin content, which also gives it aromatic properties. It has a wide range of uses including furniture, flooring, lathe work, manufacture of propeller shaft bearings for ships, and fence poles. The essential oil derived from *B. sarmientoi* wood, known as "Guayacol", "Guajol" or "Guayaco", is used in the perfume cosmetics industry and in mosquito repellents. Palo santo resin, derived from the residue of the distillation process can be used to produce dark varnishes and paints. The tree is also used for charcoal production and the leaves have been used for medicinal purposes.

The listing currently has annotation #11 covering "Logs, sawn wood, veneer sheets, plywood, powder and extracts".

A working group set up at CoP16 to review annotations concluded that finished products containing extracts of *B. sarmientoi* could be excluded from the listing with minimal impact on the conservation of the species. The proposed new annotation would ensure that extract, which is routinely exported, continues to be covered by the listing but that finished products containing extract are not. The wording does not specify that, in order to be exempted from CITES controls, finished products should be "ready for retail trade". This reflects the findings of the Annotations Working Group, based on consultation with the personal care products industry, that there are many different commodities along the production chain that are not yet packaged and ready for retail trade but whose trade has minimal conservation impact[1].

The CITES Trade Database shows that, along with timber, extract (including oil) is a key commodity of *B. sarmientoi* exported by range States – some 1000mt is reported as having been exported in the period 2010-2014. It is not clear to what extent "finished products" are exported from range States as these have not been reported as a separate term.

Analysis: According to *Res. Conf. 11.21 (Rev. CoP16)* annotations should concentrate on those commodities that first appear in international trade as exports from range States and include only those commodities that dominate the trade and the demand for the wild resource. Extracts (including oil) are clearly significant commodities in trade from range States. Although information is sparse, there is little indication that finished products are a major commodity exported by them.

The proposed amendment would closely align the annotation for this species to that for *Aniba rosaeodora* (annotation #12), which is similar in trade. The only difference is a reference to powders in the annotation for *Bulnesia*. This reference appears technically redundant, as powders are covered by the current definition of extracts (as solid – fine or coarse particles)[2].

References:
Information not referenced in the Summary section is from the Supporting Statement.

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[1] CITES (2016) SC66 Doc 25, paragraph 41. https://cites.org/sites/default/files/eng/com/sc/66/E-SC66-25.pdf. Viewed on 24th June 2016.
[2] Interpretation of the Appendices (2016) See definitions in paragraph 8 of the https://www.cites.org/eng/app/appendices.php Viewed on 24th June 2016.