ANALYSES

IUCN and TRAFFIC Analyses of the proposals to amend the CITES Appendices at the

18TH MEETING OF THE CONFERENCE OF THE PARTIES

Geneva, Switzerland, 17th – 28th August, 2019

IUCN SSC Species Survival Commission

TRAFFIC the wildlife trade monitoring network

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Prepared by IUCN Global Species Programme and Species Survival Commission and TRAFFIC

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IUCN – International Union for Conservation of Nature is the global authority on the status of the natural world and the measures needed to safeguard it. IUCN is a membership Union composed of both government and civil society organisations. It harnesses the experience, resources and reach of its more than 1,300 Member organisations and the input of more than 13,000 experts.

The IUCN Species Survival Commission (SSC), the largest of IUCN's six commissions, has over 8,000 species experts recruited through its network of over 150 groups (Specialist Groups, Task Forces and groups focusing solely on Red List assessments). Biodiversity loss is one of the world's most pressing crises, with many species' populations declining to critical levels. SSC is dedicated to halting this decline in biodiversity and to provide an unmatched source of information and advice to influence conservation outcomes, as well as contribute to international conventions and agreements dealing with biodiversity conservation.

TRAFFIC is a non-governmental organisation working globally on trade in wild animals and plants in the context of both biodiversity conservation and sustainable development. TRAFFIC plays a unique and leading role as a global wildlife trade specialist, with a team of 150 staff around the world carrying out research, investigations and analysis to compile the evidence needed to catalyse action by governments, businesses and individuals, in collaboration with a wide range of partners, to help ensure that wildlife trade is not a threat to the conservation of nature.

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FOREWORD

CITES is an international agreement between governments which aims to ensure that international trade in specimens of wild animals and plants does not threaten their survival. It originated from a resolution adopted at the 1963 IUCN Members' Assembly and entered into force on 1 July 1975.
To ensure that CITES is effective in achieving this aim, decisions taken by the Parties to CITES need to be based on the best available scientific and technical information. This is particularly the case when deciding whether or not to include species in the CITES Appendices, transfer species between Appendix I and II, or remove them from the Appendices altogether. To assist Parties in ensuring that such decisions are evidence-based, IUCN and TRAFFIC undertake technical reviews of the proposals to amend the CITES Appendices for each of the Conference of the Parties (CoPs). It is with great pleasure that we now produce the Analyses of the Proposals for CITES CoP18, which will take place in Colombo, Sri Lanka, in 2019. We would like to thank the team in TRAFFIC and IUCN for producing such a complex and helpful document in a very short time.

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

For the first time, the Analyses for CoP18 not only provide an assessment of whether or not each proposal meets the criteria specified by CITES, but also summarises any additional considerations that may be relevant to the decision on whether or not to adopt the proposal. These include, for example, any potential implementation challenges, benefits or risks that may be associated with the adoption of the proposal.

With unsustainable and illegal trade driving declines in many wild species (biological resource use generally is a threat to 10,647 species assessed as threatened on The IUCN Red List of Threatened Species), and the collective under-performance of governments, business and civil society to halt the global decline in biodiversity, CITES has a key role to play in the next decade. Wise, evidence-based decisions that are true to the Convention's aim of ensuring that international trade is not a threat to wild species, will be needed alongside the contributions of other sectors to deliver a post-2020 decade that halts species extinctions, slows declines and promotes recovery.

Dr. Jon Paul Rodríguez
Chair, IUCN Species Survival Commission

Dr. Thomas Brooks
Chief Scientist, IUCN

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 183 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 submitted to the Eighteenth Meeting of the Conference of the Parties to CITES (CoP18).

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 agreed by Parties and laid out in the listing criteria elaborated in *Resolution Conf. 9.24 (Rev. CoP17)* and other relevant Resolutions and Decisions. To ensure *the Analyses* are as accessible as possible to all Parties, we have created a bespoke webpage where *the Analyses* can be downloaded individually by proposal or in full (see https://citesanalyses.iucn.org/).

For each of *The Analyses*, a "Summary" section presents a synthesis of available information taken from each proposal's Supporting Statement and other sources, and a separate "Analysis" paragraph provides an assessment of whether or not the proposal is considered to meet the pertinent criteria in *Resolution Conf. 9.24 (Rev. CoP17)* or other relevant CITES Resolutions and Decisions. In response to feedback from Parties, for the first time, an additional paragraph is included for certain proposals to summarise any "Additional considerations" that may be relevant to the decision on whether or not to adopt the proposal (for example, implementation challenges and potential risks/benefits for the conservation of the species concerned). Information used to compile the "Summary", "Analysis" and "Additional considerations" is provided in the "Summary of available information" section. Only information from sources other than the Supporting Statement is referenced in this section, and for brevity, these references are not repeated in the "Summary", "Analysis" or "Additional considerations" sections.

To evaluate the proposals against the CITES listing criteria, 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 TRAFFIC has drawn on its own expert network and information sources to determine the nature and scale of any trade. Although draft versions of the "Summary", "Analysis" and "Additional considerations" sections were shared with relevant experts for review, the conclusions drawn do not necessarily reflect the opinions of the reviewers.

The 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 a high number of proposals to consider within the allotted timeframe and under a limited budget. We have nevertheless tried to ensure that the document is factual and objective, and consistent in how the criteria have been interpreted and applied across the range of taxa and proposals.

The Analyses were completed and made available online on 15th March 2019 to allow CITES Parties and other stakeholders sufficient time to consider the information in advance of the Conference of the Parties, which convenes on 23 May 2019 in Sri Lanka. The "Summary", "Analysis" and "Additional considerations" sections will be translated into French and Spanish and made available online. Printed versions of these sections will be made available to Parties at CoP18.

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We again acknowledge the generous support of all the project's donors in these economically difficult times.

Many individuals and institutions contributed to the compilation of *the Analyses*. Those to whom we would first like to extend our thanks are the reviewers (listed below), many of them members of the IUCN Species Survival Commission, TRAFFIC staff as well as the many other scientists and experts who have volunteered their time, data, contact list and expertise to this process. For this, we are immensely grateful.

We would also like to thank Daniele and Richard Devitre for French translation, and Wendy Byrnes for Spanish translation.

The Analyses team was made up of: Thomasina Oldfield, Willow Outhwaite, Lauren Brown and Simon Furnell (TRAFFIC), Patricia Cremona (IUCN), and a number of consultants including Martin Jenkins, James Stevens and David Whitehead. Alison Rosser, Steven Broad, Sabri Zain and Sara Oldfield also provided vital input when we unexpectedly needed additional capacity. Steven Broad, Richard Jenkins and Richard Thomas are thanked for their valuable input in reviewing *The Analyses*. Marcus Cornthwaite designed the cover. Nothando Gazi is thanked for her patience and administrative assistance. Yifu Wang provided assistance with research. Patricia Cremona led the fundraising for this project, without which it would not have been possible to undertake. All other colleagues within TRAFFIC and IUCN are thanked for their support and good humour, sugar and caffeine. Thomasina Oldfield, Willow Outhwaite and Martin Jenkins were responsible for overseeing the project.

Reviewers

We are very grateful to the reviewers who contributed their valuable time to this project. Reviewers were not asked to comment on IUCN/TRAFFIC's conclusion of whether each proposal met the relevant criteria (in the "Analysis" paragraph), for which IUCN and TRAFFIC take sole responsibility. The reviewers were:

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List of Analyses

Transfer of Heptner's Markhor *Capra falconeri heptneri* (population of Tajikistan) from Appendix I to Appendix II

Proponent: Tajikistan

Summary: The Markhor *Capra falconeri* is a large species of wild goat famed for its impressive corkscrew horns, which are sought after by trophy hunters. *Capra falconeri* was included in Appendix II in 1975 then transferred to Appendix I in 1992, and was classified on the IUCN Red List as Near Threatened in 2014. There are three subspecies currently recognised.

This proposal concerns only the population of the subspecies Heptner's Markhor *Capra falconeri heptneri* within Tajikistan. Other subspecies do not occur in Tajikistan although *C. f. heptneri* also occurs in the mountainous terrain of Afghanistan, Turkmenistan and Uzbekistan. Regarding split-listings, *Res. Conf. 9.24 (Rev. CoP17*) advises these should be avoided or, if they do occur, be on the basis of national or regional populations rather than subspecies. This Analysis assesses only the Tajik population against the criteria but takes into account information from other range States where appropriate.

The largest national population of *Capra falconeri heptneri* is in Tajikistan: nearly 2,000 were observed in 2017 in an intensive survey which covered most of the prime habitat for *C. f. heptneri* in the country (Dashtijum Strictly Protected Area and a small area of the range close to the border with Afghanistan could not be surveyed for security reasons and heavy snowfall). The distribution in Tajikistan totals 1,200 km^2. The two subpopulations in southern Tajikistan (likely not isolated from each other) are transboundary with Afghanistan in at least two areas (the Afghan population is low, sourced by the Tajik population), while a third isolated subpopulation is thought to consist of only a few dozen animals. Another subpopulation on the border with Uzbekistan is most likely extinct. Annual surveys indicate that the population appears to have steadily increased from 1,000 in 2012, although the figure reported in 2018 (2,650) is considered likely to be an over-estimate and the population may have reached carrying capacity in some areas. The population status varies by area: three out of seven surveyed areas reportedly had growing populations in 2017. Threats include overgrazing and disease transmission from livestock as well as poaching for meat or trophies.

The management of *Capra falconeri heptneri* in Tajikistan is considered by some to be a good example of sustainable use leading to improved conservation outcomes. From around 2004, several traditional local hunters established small enterprises dedicated to Markhor conservation and future sustainable use. "The Mountain Ungulate Project" led to the establishment of several community-based conservancies. In the 2013–2014 season, the government issued the first hunting quota of *C. f. heptneri* in Tajikistan of six permits, which increased to 12 by 2018–2019. Most, but not all, concessions in the subspecies' range are managed by local families. The revenue from permits, plus additional expenditure by hunters totals tens of thousands of dollars, and has the potential to generate significant revenue and benefit communities. *Capra falconeri heptneri* populations are said to be increasing in at least three conservancies, but concerns have been raised that in some areas unsustainable hunting is occurring and that any benefits to local people have been very limited. The subspecies is protected within Tajikistan (hunting is only allowed by special decree by the national government) and part of its range is within protected areas.

Tajikistan currently has a methodology in place for calculating quotas based on minimum numbers of *Capra falconeri heptneri* within a conservancy (including trophy-aged males) and limits on the percentage of the population that can be hunted. Surveys are conducted every one to two years. Quotas are allocated per season and Tajikistan states that it implements an adaptive management approach. If the Tajik population is transferred to Appendix II, the proponent indicates that it will continue to set a quota, but it is not clear if the current system to calculate future offtake will continue to be employed. Problems with enforcement of the current system have been identified, including the hunting of young males below the legal trophy age. The number of trophies reported as imports from Tajikistan is lower than the number of hunting permits used.

Analysis: The species is affected by trade: trophy hunting is permitted (based on a quota system) and successful community-based management has aided population recovery and benefited local communities. Unsustainable hunting and illegal trade have been reported.

The observed Tajik population of *Capra falconeri heptneri* is around 2,000. Although not all animals were counted, since the survey covered most of the prime habitat it is very unlikely that the actual total population exceeds the guidance of 5,000 given in *Res. Conf. 9.24 (Rev. CoP17)* for a small wild population. Overall the population in Tajikistan is increasing, although some of this is due to an apparent change in the survey area size. Therefore, the Tajik population may be considered to no longer meet the biological criteria for inclusion in Appendix I. Although the national population is growing, this recovery is still recent, restricted to certain areas and delicate.

Annex 4 of *Res. Conf. 9.24 (Rev. CoP17*) advises that species in demand in international trade should only be transferred to Appendix II if Parties are satisfied with the precautionary measures stipulated by the proponent. Given concerns expressed over the sustainability and legality of some hunts under the existing quota allocation system, it is not clear that the precautionary measures for down-listing *Capra falconeri heptneri* to Appendix II are met. Furthermore, as two of the subpopulations are contiguous with those in Afghanistan where poaching occurs, a split listing by country may be difficult to implement.

Other considerations: Successful community-based management has aided population recovery and benefited local communities. The majority of concessions have publicly stated they do not support a transfer to Appendix II (including those that have growing populations).

Difficulties in obtaining import permits for trophies have been reported by some hunters, and an Appendix II listing may facilitate imports. However, legal trade in Appendix I trophies is occurring as evidenced by imports reported in the CITES Trade Database to a number of countries (including the USA and European countries) so it is not clear whether this is a significant issue, or whether the problematic imports are due to the trophies being hunted in contravention of quotas or other requirements. This issue could be addressed directly between Tajikistan and the importing countries. There may also be potential to amend *Res. Conf. 10.15 (Rev. CoP14) Establishment of Quotas for Markhor Hunting Trophies* (to include Tajikistan), as it currently includes quotas for Pakistan and will be discussed at CoP18 since Pakistan seeks to increase their quota.

Transfer of Saiga Antelope *Saiga tatarica* from Appendix II to Appendix I

Proponent: Mongolia and the USA

Summary: *Note:* this proposal is to transfer *Saiga tatarica* from Appendix II to Appendix I. From the Supporting Statement it is evident that the Proponents consider this to refer to all living saiga. However, CITES adopted nomenclature recognises two separate species of saiga: *Saiga borealis,* endemic to Mongolia and elsewhere considered to be *Saiga tatarica mongolica*, and *Saiga tatarica*, elsewhere considered to be *S. tatarica tatarica,* comprising all other populations. Because proposals for species already included in the Appendices should follow CITES taxonomy, this proposal excludes *Saiga borealis* and only applies to the non-Mongolian populations of saiga, recognised under CITES as *S. tatarica* and elsewhere as *S. t. tatarica.*

Saiga borealis was included in CITES Appendix I in 1975 as *Saiga borealis mongolica*, but removed from the Appendices in 1979. In 1995 *Saiga tatarica* was listed in Appendix II, at that time the Mongolian population was considered a subspecies of *Saiga tatarica* and included in that listing, but subsequent adoption of Wilson and Reeder (2005) as the CITES Standard Taxonomic Reference for mammals, including saiga, resulted in the splitting of this taxon into *S. borealis* and *S. tatarica*, a division now widely recognised to have been in error but enshrined in CITES taxonomy until a new reference is adopted. Currently, both *Saiga tatarica* and *Saiga borealis* are listed in CITES Appendix II.

In this analysis information is provided on all saiga, divided where possible into the two CITES-recognised species: information from the Supporting Statement and IUCN Red List that refers to *S. t. mongolica* is considered as referring to *S. borealis* and all other populations as applying to *S. tatarica.*

***Saiga tatarica (sensu* CITES)**

Saiga tatarica, is a nomadic herding antelope that inhabits open dry steppe grasslands and semi-arid deserts across Central Asia. There are four distinct populations: one in the Russian Federation (the Kalmykia population), and three within Kazakhstan (the Betpak-dala, Ustyurt and Ural populations). Of these, the Ural population is somewhat transboundary with Russia, while the Ustyurt population makes seasonal migrations into Uzbekistan and Turkmenistan.

Historically, populations of *S. tatarica* numbered in the millions, until excessive hunting reduced them to low thousands of individuals at the beginning of the 20th Century. Since then, the population has undergone large fluctuations in size. From the early 1990s there was a decade of rapid decline caused by excessive hunting for meat and horns after the collapse of Soviet regulatory systems. Between 2006 and 2018 the population increased overall from an estimated 60,000 to in excess of 220,000 individuals (despite a large disease-related die-off in 2015). The next annual census scheduled for May 2019 is likely to show further growth in population size.

While hunting is prohibited in all range States, the species faces a range of threats, including disease, habitat loss, poaching and the blocking of migration routes by infrastructure. The greatest cause of mortality recently has been sporadic outbreaks of disease, which cause severe population crashes and large, temporary, fluctuations in population size. In 2015, a bacterial infection killed more than 200,000 saiga in Kazakhstan (more than 80% of the affected population and more than 60% of the global population), within a three-week period.

However, due to their high fecundity, (females mature at around eight months and usually produce twins), populations can rapidly rebound, with annual population growth in excess of 40% reported. Since the 2015 mass-die off, populations within Kazakhstan have undergone a strong recovery, increasing from 153,000 in 2017, to 215,000 in 2018.

Saiga are traded primarily for their horn, which is used widely in Traditional Asian Medicines. Reported trade in recent years has largely been between non-range States in Asia, including China, Japan, Hong Kong SAR and Singapore, much of it declared as originating in pre-Convention stockpiles.

While the range States currently prohibit all trade, horns from poached saiga also enter the market, particularly through trafficking routes to China. As only the males have horns, the selective poaching of males can skew the sex ratio, which in the early 2000s led to the reproductive collapse of the Russian population.

All range States are actively engaged with saiga conservation initiatives, which are coordinated through a Memorandum of Understanding (MoU) of the Convention on the Conservation of Migratory Species of Wild Animals (CMS), in partnership with CITES. Restoring saiga populations to a point where sustainable use is possible is the long-term goal of this MoU.

Saiga borealis
Mongolian saiga, *Saiga borealis*, are endemic to Mongolia, and isolated from populations of *Saiga tatarica* by the Gobi Altai Mountains. This species is nationally protected, with hunting and export of all saiga strictly prohibited. It faces a range of threats including harsh climatic conditions, competition for forage with livestock, and outbreaks of disease. Longer-term population trends are hard to assess due to changes in survey methods, but in the 2000s the population recovered from very low numbers to a high of around 15,000 individuals in 2014, due to conservation efforts. Since then, an outbreak of Peste des Petits Ruminants (PPR) disease in 2016–2017 killed 54% of the population, reducing it to fewer than 5,000 individuals. A harsh winter also contributed to further declines, and by 2018 the population was an estimated 3,000 individuals.

Saiga borealis is also subject to the conservation measures of the CMS MoU, which was amended to cover *Saiga* spp. in 2010.

Analysis
Saiga tatarica
Saiga tatarica has a large area of distribution, its population exceeds 220,000 individuals and is currently increasing. Historical decline has been significant. In the past decade, outbreaks of disease have caused large and sudden reductions in the size of the population. Global threat assessments made only a few years ago rightly reflected negative trends observable at that time. However, the ability of populations to quickly rebound at rates exceeding 40% per year, gives the species significant resilience to such mass-mortality events. In considering trends now, despite recent fluctuations, when measured over the last three generations (around 11 years) *S. tatarica* has not undergone a recent marked decline and is increasing overall. National protection measures, export bans from range States and collaborative conservation actions under the CMS MoU provide a significant degree of security at present.

The vast majority trade in saiga horn is believed to be derived from *S. tatarica*, with legal trade occurring outside the range States based on stockpiles of pre-Convention horns. Illegally sourced horns from poached animals are laundered into this market, although current levels of poaching are not considered to represent a threat to the survival of the species.

Saiga borealis
Saiga borealis is endemic to Mongolia. It has a small population of less than 5,000 individual that has decreased from over 14,500 since 2013/2014 due to an outbreak of disease and harsh winter conditions, although the population is subject to significant fluctuations. This decline would fall within the guidelines for marked recent declines for small populations given in Annex 5 of *Res. Conf. 9.24 (Rev. CoP17)* a percentage decline of 20% or more in the last 5 years or 2 generations (whichever is the longer). Large short term fluctuations have been caused by disease outbreaks (the most recent in 2016-17). It faces a range of threats including harsh climatic conditions, competition for forage with livestock, and outbreaks of disease. Horns of both species strongly resemble each other, however, it

appears that the majority of trade from pre-Convention stockpiles outside the range States are of *S. tatarica.* Although poaching does not represent a major threat to this species, horns from poached animals may be laundered into this legal market. It appears that *S. borealis* meets the criteria for listing in Appendix I, although this would result in implementation challenges with the Appendix-II listing for *Saiga tatarica*. However, this species is not within the scope of the proposal now under consideration according to current CITES adopted nomenclature.

Other Considerations:
Listing *S. borealis* in Appendix I would result in implementation challenges, due to the strong resemblance of its parts and derivatives in trade to those of *Saiga tatarica*.

Summary of Available Information
Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy
Wilson and Reeder (2005), the current CITES Standard Reference for saiga recognises two species – Saiga tatarica and Saiga borealis – and both are included in the CITES Appendices.

Range
Saiga tatarica:
Extant (resident): Kazakhstan; Russian Federation; Uzbekistan
Extant (non-breeding): Turkmenistan
Extinct: China; Ukraine

Saiga borealis:
Mongolia

IUCN Global Category
Taxonomy used for the IUCN Red List treats saiga as one species: Saiga tatarica with two subspecies (see below for details). Assessments have been made for the species and both of the subspecies:

Saiga tatarica tatarica (considered under CITES taxonomy as Saiga tatarica): *Endangered A4abde. Assessed February 2018 (The IUCN SSC Antelope Specialist Group 2018b).*

Saiga tatarica mongolica (considered under CITES taxonomy as Saiga borealis): *Endangered A4abde. Assessed February 2018 (The IUCN SSC Antelope Specialist Group 2018c).*

Saiga tatarica: (covering both species considered under CITES taxonomy): *Critically Endangered A2acd* (date of assessment, February 2018). However, the IUCN SSC Antelope Specialist Group (2018a) add that Saiga tatarica currently does not meet the Red List Criteria for Critically Endangered and is considered to have crossed the thresholds between Critically Endangered and Endangered in 2015. However, this reassessment falls under the IUCN's five-year rule. This rule applies to taxa that appear to be genuinely improving in status, and states that "a taxon may be moved from a category of higher threat to a category of lower threat if and when none of the criteria of the higher category has been met for five years or more". Saiga is considered to have crossed the thresholds between CR and EN around 2015, therefore its status will be re-evaluated again in 2020.*

Biological criteria for inclusion in Appendix I

A) Small wild population
Saiga population numbers undergo fluctuations and are capable of increasing and decreasing rapidly:

Saiga tatarica
Data from the last three years indicates that the current population of Saiga tatarica *doubled between 2016 and 2018, and now exceeds 220,000 (SCA Data, 2019).*

Saiga borealis
In 2018, there were an estimated 3000 Saiga borealis in Mongolia (B. Chimeddorj, in litt., *2019; SCA Data, 2019).*

B) Restricted area of distribution
Saiga tatarica

Saiga tatarica occurs in four populations, throughout south-eastern Europe and Central Asia. The Ural, Betpak-Dala, and Ustyurt populations are in Kazakhstan, with the Ustyurt population also in Uzbekistan and sometimes wintering in Turkmenistan, *and the Ural population also in the Russian Federation (E.J. Milner-Gulland,* in litt., *2019).* The north-west Pre-Caspian (or Kalmykia) population is in the Russian Federation. There is minimal interaction between these populations. Each year, some populations of *Saiga tatarica* undertake irregular long-distance migrations (sometimes between range States), between summer pastures in the northern parts of their range, where births occur, and overwintering grounds to the south, where they mate. *In 1958, the saiga's range covered an estimated 2.5 million km^2 (IUCN SSC Antelope Specialist Group, 2018a). While the range of* Saiga tatarica *has contracted since then, and recent figures estimating its current Area of Occupancy have not been calculated, this species' occupancy of four geographic areas and migratory lifestyle means that its range is not restricted.*

Saiga borealis

Saiga borealis is endemic to Mongolia and is isolated from all populations of *Saiga tatarica* by the Gobi Altai Mountains. *The population is somewhat sub-divided but this is likely to be to do with range contraction and low numbers rather than permanent isolation (E.J. Milner-Gulland,* in litt.*, 2019). The population* seasonally changes its pastures but does not migrate long distances. *While a precise Area of Occupancy has not been calculated, in 2017, the range of* Saiga borealis *within Mongolia was estimated to be 14,713 km^2 (Chimeddorj & Buuveibaatar, 2017). During the last few decades, the Mongolian Saiga's range has declined, although more recently, groups have become established within a number of new areas (IUCN SSC Antelope Specialist Group, 2018c).*

C) Decline in number of wild individuals

Both species are capable of undergoing rapid fluctuations in population size. Declines are driven by natural factors such as disease epidemics and severe weather, and by excessive hunting, while population rebounds are due to the saiga's high fecundity. Males become sexually mature at about 19 months, while females reach sexual maturity at around 8 months. Saiga are polygamous, and females exhibit a high level of fertility with two-thirds of females frequently producing twins. It is estimated that 95% of adult and 80% of young females conceive in an average year. *In favourable climatic conditions, a population can increase by up to 60% in a single year (Chan* et al., *1995). Very few animals in a population are more than 3.5 years old, indicating that the population is almost completely renewed after four years (Bekenov et al. 1998).*

Saiga tatarica

Historically, herds of Saiga tatarica numbered in the millions (IUCN SSC Antelope Specialist Group 2018a). By the early 20th century, excessive hunting reduced them to an estimated total of low thousands (Heptner et al. 1961; Milner-Gulland, in litt., *2019). The regulation of hunting then saw their numbers recover to over 1 million by the 1980s (Data supplied by the Saiga Conservation Alliance (SCA Data) 2019). From around 1995, their populations underwent rapid declines, primarily driven by excessive poaching for both meat and horns following the collapse of the Soviet Union (IUCN SSC Antelope Specialist Group 2018a; SCA Data, 2019).*

Since then, a wide range of conservation measures have seen saiga populations first stabilise, then increase (Milner-Gulland, 2010), although regional trends differ. By January 2018 the global population of Saiga tatarica *was estimated to be between 159,000 and 160,600, with the number of mature individuals estimated at 119,700 to 120,450 (IUCN SSC Antelope Specialist Group, 2018b). More recent estimates (based on aerial surveys in Kazhakstan and vehicle surveys in Russia), indicates that the global population further increased in 2018, to around 220,000 (SCA Data, 2019). Figure 1 shows the population data for the four subpopulations of* Saiga tatarica*, from 2000 to 2018.*

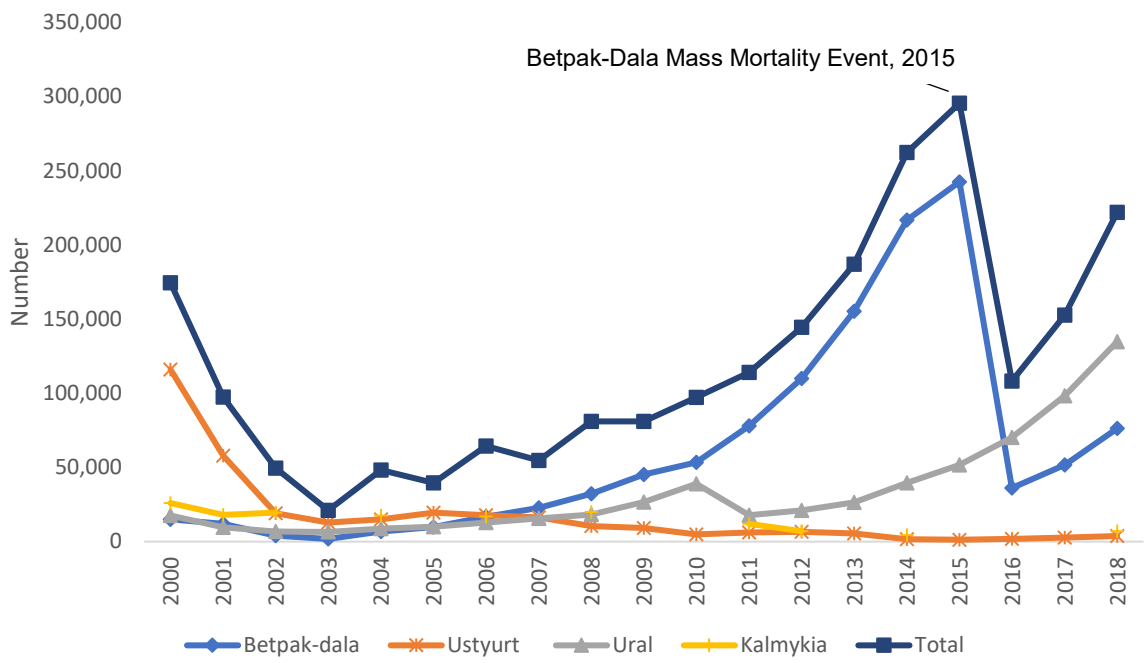

Fig. 1. *Estimated population sizes for* Saiga tatarica*, 2006-2018. Data supplied by Saiga Conservation Alliance, (2019).*

Although trends for the different subpopulations have varied over the last three generations (with the IUCN Red List recognising 3 generations as 11 years whereas Milner-Gulland suggesting a range between 9-12 years) an overall increase in population has been observed over that time period.

Between 2003 and 2015, the largest population at ***Betpak-dala****, (Kazakhstan) grew steadily from 1,800 individuals to 242,500 (SCA Data, 2019). In 2015, an outbreak of disease caused by the bacterium Pasteurella multocida, resulted in a mass-die off in the Betpak-dala population (Kazakhstan), with >200,000 (>80% of this subpopulation,* and more than half of the global population*) dying within a three-week period (IUCN SSC Antelope Specialist Group, 2018a; E.J. Milner-Gulland,* in litt.*, 2019). The population at Betpak-dala has, however, rebounded, increasing from around 31,300 following the die off to an estimated 76,361 in 2018, representing an annual population growth of >40% between 2016-2018 (SCA Data, 2019).*

The ***Ural*** *population the majority of which occurs in Kazakhstan (with a proportion of the population seasonally migrating into the Russian Federation (Singh,* et al.*, 2010)), also suffered a mass-die off due to disease in 2010, with the loss of around 12,000 animals (one-third of this subpopulation) (Dieterich & Sarsenova, 2012)This was followed a year later by a smaller die off of several hundred animals in the same location ((IUCN SSC Antelope Specialist Group, 2018c). Pasteurellosis was implicated based on clinical signs and bacteriology, but very little pathology was undertaken (Kock et al. 2018). Since this die-off, the population has recovered strongly, and in 2018 was estimated to be 134,823, making it the largest S. tatarica subpopulation at this moment in time (SCA, 2019)*

The ***Ustyurt*** *population (primarily Kazakhstan but also found in Uzbekistan and sometimes wintering in Turkmenistan), reduced from around 250,000 in the early 1990s to 1,270 in 2015, primarily due to poaching. The latest population trends for Ustyurt are increasing, with the latest official figures (2018) being around 3,600 individuals (SCA, 2019; E.J. Milner-Gulland,* in litt.*, 2019).*

Poaching and other factors, have also caused large long-term declines of the ***Kalmykia*** *population in Russia, from around 380,000 in 1980, to 17,600 in 2004 (IUCN SSC Antelope Specialist Group 2018a; SCA, 2019). For more recent years, however, as population surveys in Kalmykia are not performed annually and rely on vehicle surveys which are much less reliable than aerial surveys, (due to the need to adhere to road accessible areas, and the possibility of animals fleeing from the vehicle, Milner-Gulland,* in litt.*,(2019)), the population data for Kalmykia is rather incomplete. Between 2008 and 2014, the population appeared to fall from 20,000 to 4,500, although by 2018, was estimated to have climbed back to 7,000 (SCA, 2019). While there is some evidence for continuing poaching (Kühl* et al.*, 2009), and the population is possibly declining (E.J. Milner-Gulland,* in litt.*, 2019), some conflicting reports regarding the status of this population (D. Mallon,* in litt.*, 2019), and recent rough*

estimates of around 10,000 animals (E.J. Milner-Gulland, in litt.*, 2019), suggest that the population at its current level may have stabilised.*

Saiga borealis
In 1977 Saiga borealis *had reportedly been reduced to the low hundreds by uncontrolled poaching (IUCN SSC Antelope Specialist Group, 2018c).* Since *then, favourable climatic conditions and active conservation measures saw its numbers increase to a peak of nearly 15,000 in 2014 (SAC Data, 2019). In 2016, the population was around 11,000 (Chimeddorj & Buuveibaatar, 2017), before an outbreak of Peste de Petits ruminants (PPR) killed 54% of the population in 2016-2017, reducing it to around 3,000 individuals in 2018, in combination with a harsh winter and poor grazing (B. Chimeddorj,* in litt., *2019; 2019; SCA Data, 2019).*

Population estimates for Saiga borealis *from two sources are shown in Figure 2. As survey effort and methods vary (for example, surveys may be conducted from ground vehicles or aircraft), figures between the two are not directly comparable. Best available estimates from official Saiga Conservation Alliance data are indicated by solid bars, with other estimates compiled by Chimeddorj and Buuveibaatar (2017) indicated by hatched bars.*

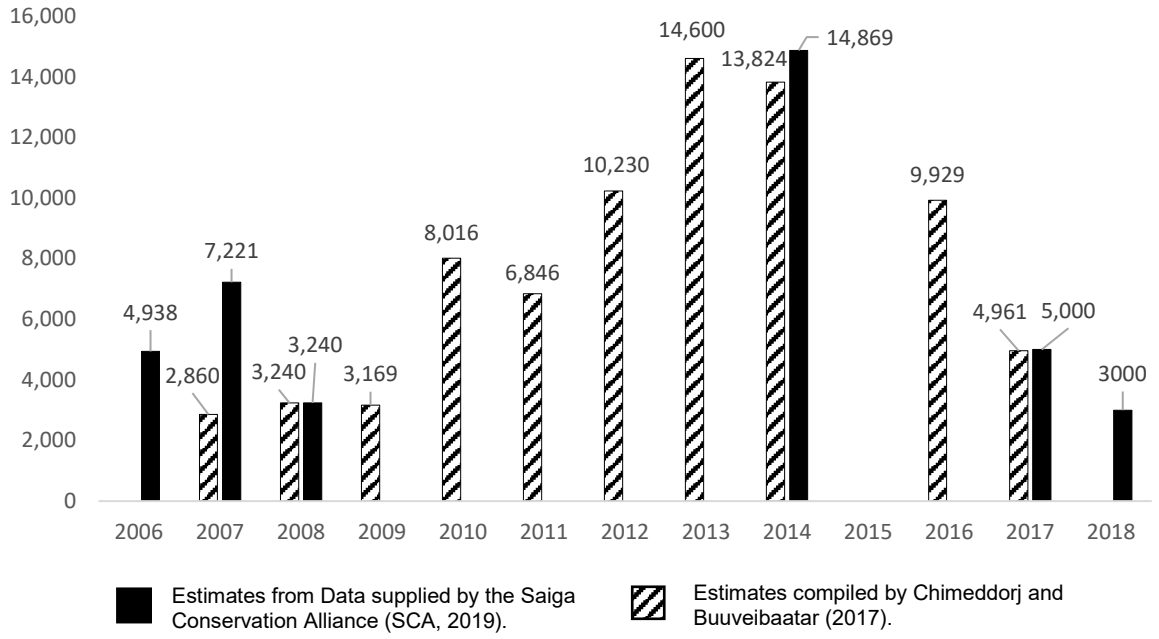

Fig. 2. *Population estimates for* Saiga borealis*, 2006-2018.*

As indicated by figure 2, despite a variety of methods having been used to generate the population estimates, it is apparent that Saiga borealis currently has a small population of <5,000 individuals. The population has also undergone a recent marked *decline of well over 20% over the last 2 generations (6-8 years), to an estimated 3,000 individuals in 2018. Annex 5 of Res. Conf. 9.24 (Rev. CoP17) gives a general guideline for marked recent decline where the population is small as a percentage decline of 20% or more in the last 5 years or 2 generations (whichever is the longer). However, as seen from recoveries from similarly small populations in 2008, Saiga numbers can recover swiftly.*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade
Legal trade can occur through the use of stockpiles of horn legally acquired before range State export bans came into force (UNEP, 2015), and which are predominantly comprised of S. tatarica *(E.J. Milner-Gulland,* in litt.*, 2019). The main trading countries/territories are China, Hong Kong SAR, Indonesia, Japan, Malaysia, Singapore and Viet Nam (CITES, 2018b),* where the most commonly used saiga horn products include bottled 'fresh' saiga water, shavings, bottled 'supermarket' saiga water, and tablets. *In 2006, Singapore reported a horn stockpile of around 33,000 tonnes, which had fallen to an estimated "less than 20,000 tonnes" by 2015 (Theng & Krishnasamy, 2017).*

According to an analysis conducted by UNEP-WCMC of reported trade in specimens of saiga for 2007-2016, legal international trade in horns and derivatives appears to have declined over the last decade, while trade in

medicine as a finished product has increased between Asian non-range States (CITES, 2018a). In 2018, the major saiga consumer and trading States did not report any particular difficulties or challenges in regulating the trade in saiga specimens (CITES, 2018a). Both S. tatarica *and* S. borealis *are reported (although the method by which the species were identified is unclear).*

*However, r*egularly reported seizures of horns provide evidence for the existence of illegal trading networks that enable poached saiga to be trafficked between range States and China *and laundered into legal trade*. Markets of saiga horn are found in several countries of South and East Asia, fuelling poaching in the saiga ranges of Russia and Kazakhstan. Saiga horns are mainly smuggled by trucks that cross the border between Russia and Kazakhstan into China. Downstream Asian markets include China, Hong Kong SAR, Japan, Singapore, and Malaysia.

Details of seizures made within range States and consumer countries made between 2007 and 2012 are compiled within CoP16 Inf. 4 (CITES, 2013). Other, more recent examples include:

- *In 2014, 162 horns were seized in Russia (the equivalent of 81 saiga), with the perpetrators reportedly intending to illegally transport the horns from Russia into China.*
- *In 2016, 468 horns, 72 horn pieces and 170 skins (total value ~USD 46,000) were seized in Uzbekistan (Saiga Conservation Alliance, 2016).*
- *In Kazakhstan, 79 criminal cases were initiated for saiga poaching in 2014, followed by 107 cases in 2015, with >3,000 horns confiscated as a result (Saiga Conservation Alliance, 2016).*
- *Between 2012 to 2015, China, Japan and Malaysia also reported additional seizures (CITES, 2016).*

Seizures of saiga horn are also made in Mongolia, although it is likely that these are often of S. tatarica *horns in transit from another range State (E.J. Milner-Gulland,* in litt.*, 2019). In 2013, a case involving 76 horns (whether* borealis *or* tatarica *horns in transit was not specified) was prosecuted, while 162 horns in transit from Russia were intercepted in 2014 (National Report for Mongolia, 2015). It was also observed that horns were cut from many male saigas that had died during the PPR outbreak in 2016-17, and it is possible that these horns have also illegally entered into trade (Chimeddorj & Buuveibaatar, 2019) although no further information on this could be found.*

Additional Information

Threats

The range of both species places them at risk of climatic extremes, including summer droughts and severe winters (IUCN SSC Antelope Specialist Group, 2018a). A severe winter in 2018 negatively impacted the population of S. borealis, *and another harsh winter is expected in 2019, exacerbated by the poor condition of grazing pastures following a dry summer (Webwire, 2019).* The development of infrastructure including fences, roads and railways can also negatively impact saiga movement and migrations. Saiga rarely cross railway tracks, and the projected expansion of road and train networks in Kazakhstan and Mongolia may therefore pose a serious threat to some populations in future.

The greatest cause of mortality over the last 3-5 years, however, has been from outbreaks of disease, which cause mass mortality events (MME) (E.J. Milner-Gulland, in litt., *2019), the impacts of which are described above. The 2015 MME that affected the Bepak-Dala population, was caused by the bacterium Pasteurella multocida type B, which is also carried by livestock and causes hemorrhagic septicaemia (Kock* et al.*, 2018). The pre-conditions that lead to outbreaks of* P. multocida *are not yet understood, but climatic factors may play a role, which* may be exacerbated in future by changing environmental conditions, with projected increases in temperature, precipitation and humidity in Kazakhstan *potentially increasing the prevalence of such pathogens (Kock* et al.*, 2018).* The saiga's unusually high birth rate, however, means that populations can recover rapidly from these events *(see discussion of recent population trends above). The geographic isolation of different populations of* Saiga tatarica *makes it highly unlikely that they could all be affected be disease at the same time, but the potentially severe impacts upon individual populations underlines the importance of improving the viability of each, to act as an insurance policy against this threat (Milner-Gulland, 2015).*

The 2016-17 MME that killed 54% of the population of S. borealis *was caused by the* virus peste des petits ruminants (known as PPR, or "goat plague"), which was *transmitted by livestock (E.J. Milner-Gulland,* in litt.*, 2019).* Both species also face anthropogenic pressures and are threatened by habitat loss and degradation through conversion of land for agricultural use and livestock grazing. Loss of habitat to grazing livestock land is also problematic because livestock can transmit diseases and parasites to saiga. *The habitat of* Saiga borealis *in Mongolia is described as severely fragmented (National Report for Mongolia, 2015), but recent range expansion occurred prior to the PPR outbreak in 2016 (D. Mallon,* in litt.*, 2019). However,* Saiga borealis *are still confined to the western part of their former range, expansion to the south-east now being severely limited by a narrow bottleneck between two mountain ranges that is densely occupied by people and livestock (D. Mallon,* in litt.*,*

2019). In Mongolia, high livestock density is considered a particularly serious threat to S. borealis*, through both competition for grazing and an increased risk of disease transmission (Kock, 2017). Veterinarians reported that the poor physical condition of saiga in Mongolia caused by habitat degradation, may have contributed to the severity of the disease outbreak that killed 54% of the population in 2016 (S. Zuther,* in litt.*, 2019).*

Saiga are also illegally hunted for their meat (for local consumption (CITES, 2016)), and horns, which are perceived to cure many diseases within Traditional Asian Medicine*. In a study of Russian and Kazakhstan communities, Kühl* et al*. (2009) found that this illegal exploitation was closely linked to levels of poverty and unemployment. Saiga are protected in all range States but the level of enforcement varies. Anti-poaching efforts in Kazakhstan and other countries have been intensified over the last 10 years (IUCN Antelope Specialist Group, 2018a).*

All populations of Saiga tatarica have been heavily impacted by poaching, the incidence of which dramatically increased during the 1990s following the collapse of the Soviet Union and the opening of the Chinese-Soviet border (Milner-Gulland, 2003). Poaching continues and authorities make regular seizures of horn (see trade section above).

Despite the regularity of poaching incidents, however, the available evidence does not suggest that the current levels of illegal exploitation are a significant threat to the future of the Saiga tatarica *as a species (D. Mallon,* in litt.*, 2019; E.J. Milner-Gulland,* in litt., *2019).*

Saiga borealis *is also targeted by poachers (IUCN Antelope Specialist Group, 2018c). According to national police and ranger patrol data in Mongolia, a total of 27 cases involving the poaching of 231 saiga were recorded during 2005-2017, with the number of poached saiga peaking in 2013 (Chimeddorj & Buuveibaatar, 2019). Two poaching cases (involving 16 saiga) in the period 2010-15 were reported by the National Report for Mongolia (2015).*

As poachers often target males for their horns, this selective offtake can also skew the sex ratios of both species. *As saiga are harem breeders, this can have a significant impact upon the reproductive success of populations (Milner-Gulland, 2003). In the late 1990s, heavy poaching of male saiga within the Russian Pre-Caspian population resulted in a lowering of female fecundity, leading to reproductive collapse (Milner-Gulland,* et al., *2003). In recent years, however, while the proportion of males in many populations remains low, it is currently thought to be sufficient to maintain a normal level of reproduction (S. Zuther,* in litt., *2019).*

Conservation, management and legislation

The saiga is legally protected in all countries of its breeding range, and hunting is not permitted in all five range States, at least until 2020. *The export of all saiga products from range States is also prohibited (CITES, 2016).*

The range of both species partially falls within a number of protected areas, but for most populations, protected areas do not cover the entire year-round range of this highly migratory species (S. Michel, in litt.*, 2019).*

Population monitoring has involved the joint initiatives of a range of conservation organisations, and includes aerial surveys of *Saiga tatarica* in Kazakhstan, and ground surveys and satellite collaring of *Saiga borealis* in Mongolia*. In Russia, since 2006, surveys have been undertaken by ground-based vehicles, although these are not performed annually (E.J. Milner-Gulland,* in litt.*, 2019; SCA Data, 2019).*

Saiga range States have implemented several conservation measures, including investment in new protected areas, ranger teams, training and anti-poaching initiatives, to improve population status (D. Mallon, in litt., *2019).* The Convention on the Conservation of Migratory Species of Wild Animals (CMS), *in collaboration with CITES,* developed a Memorandum of Understanding (MOU), *with an associated Medium-Term International Work Programme (MTIWP) updated on a five-yearly basis (Milner-Gulland, in litt., 2019)*, that provided a non-legally binding instrument for the conservation of saiga, which has been signed by all saiga range States. Ten international conservation organisations *have signed on as cooperating organisations, committing to working towards implementing the MTIWP (Milner-Gulland, in litt., 2019)*, which includes improving and increasing population monitoring; improving protected area networks; reducing poaching; creating alternative livelihoods; captive breeding; and raising awareness at all levels.

Furthermore, in 2014, 14 Asian countries agreed to the development of the Central Asian Mammals Initiative (CAMI) under the CMS. In 2017, CMS, under the MOU, created the first database of Central Asian migration routes, along with planned and constructed infrastructure.

In 2017, the CITES Standing Committee encouraged *signatories to the MoU* to address challenges in controlling illegal trade in saiga horns and derivatives, including ensuring effective stockpile management. *The next technical meeting of the MoU signatories is in April 2019, to develop a revised MTIWP for the period 2020-2025 which will be approved at a meeting of the signatories in 2020 (E.J. Milner-Gulland,* in litt.*, 2019).*

Artificial Propagation/captive breeding

In the past, captive breeding in zoos has shown poor success. There are currently eight known captive centres in the world for saiga (not including zoos). Six of these occur within the current range of saiga (four in Russia, two in Kazakhstan), and two occur in Ukraine (in a semi-captive reserve), *one of which has proved very successful at breeding saiga (D. Mallon,* in litt.*, 2019),* and China. Currently there are around 900 saiga in captivity. *More captive centres are planned (E.J. Milner-Gulland,* in litt., *2019).*

Implementation challenges (including similar species)

While Saiga tatarica *and* Saiga borealis *show morphological differences, products in trade including their horns cannot be differentiated by non-experts (E.J. Milner-Gulland,* in litt.*, 2019; S. Zuther,* in litt., *2019). Both may be represented in stockpiles outside the species' range States, the current source of legal trade, although it is likely that the vast majority of the stockpiles are S. tatarica (E.J. Milner-Gulland,* in litt., *2019).*

Potential risk(s) of a transfer from Appendix II to I

The long-term goal of the CMS Saiga MoU is to restore Saiga spp. populations to a point where sustainable use is again possible (MTIWP, 2015; CITES, 2018b), and listing Saiga tatarica in Appendix I would impede some aspects of this goal (D. Mallon, in litt.*, 2019).*

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Transfer of the Vicuña *Vicugna vicugna* population of the Province of Salta (Argentina) from Appendix I to Appendix II with annotation 1

Proponent: Argentina

Summary: The Vicuña is a South American member of the camelid family that produces high-quality wool. Populations were heavily depleted by hunting in the mid-20th century to supply wool fibre for export, the species was consequently listed in Appendix I in 1975. Following a rapid recovery of the species, some populations in Peru and far northern Chile were transferred to Appendix II to allow the export of appropriately labelled fabric woven from wool fibre sheared from live animals. Other populations have followed suit, including ones in Argentina and Bolivia and a small introduced population in Ecuador. The current conditions for export, which regulate how the fibre should be harvested and labelled for export, are set out in annotation 1.

In 2018 the global Vicuña population was estimated at approximately half a million animals. The species is currently classified by IUCN as Least Concern.

The Argentinian Vicuña population was estimated at between 73,000 and 127,000 individuals in 2006 (dependent on the census method). Wild populations occur in five provinces: Catamarca, Jujuy, La Rioja, Salta and San Juan. Populations in Jujuy and Catamarca were transferred to Appendix II in 1987 and 2003 respectively. Semi-captive populations in all provinces, including that in Salta Province, are all also currently included in Appendix II. The current proposal is to transfer the wild population of Salta to Appendix II. In Argentina, this would leave only the small wild populations of La Rioja and San Juan listed in Appendix I.

The wild population in Salta Province in 2018 was estimated at just under 60,000, compared with around 30,000 in 2013. Suitable habitat within the extent of occurrence in Salta is calculated to be around 26,000 km^2, population densities vary considerably within this area. The species is covered by a range of national and provincial laws and regulations and is present in protected areas, including the “Los Andes” faunal reserve in the south-west of Salta, which protects around 40% of Vicuña habitat in the province.

Analysis: The Vicuña population of Salta Province, Argentina, does not meet the biological criteria for retention in Appendix I – its population is large, increasing and distributed over a large area. The species is in trade and in this regard is intended to be managed in the same way as the adjacent and contiguous populations of Jujuy and Catamarca Provinces, also in Argentina. These have been included in Appendix II for over 20 and over 15 years respectively with no evident problems. It would appear therefore that precautionary measures set out in *Res. Conf. 9.24 (Rev. CoP17)* are met.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Range

The proposal concerns the population of Salta Province, Argentina.
The species occurs in Argentina, Bolivia (Plurinational States of), Chile, Ecuador (introduced), Peru.

IUCN Global Category

Least Concern (assessed 2018, ver. 3.1).

<u>Biological criteria for inclusion in Appendix I</u>

A) Small wild population

According to a 2018 report, the Vicuña’s population in Salta province in 2018 was estimated at just under 60,000, compared with around 30,000 in 2013. *The national population was estimated in 2006 as between 73,000 and 127,000 individuals.*

B) Restricted area of distribution

According to the supporting statement the Vicuña's extent of occurrence in Salta is around 35,000 km^2, of which around 90,00 km^2 is unsuitable for the species, giving around 26,000 km^2 of suitable habitat, although population densities vary considerably within this area. Wild populations occur in five provinces: Catamarca, Jujuy, La Rioja, Salta and San Juan.

C) Decline in number of wild individuals

According to a 2018 survey, the population Vicuña in Salta Province doubled between 2013 and 2018 *(see small wild population above).*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

There is international demand for Vicuña. In the period 2000 to 2017 according to the CITES Trade Database (2019), Argentina reported the direct export of ca. five tonnes (5,100 kg) of hair, fibre, skins and cloth: around 80% sourced from the wild.

In **Argentina,** *245 kg of fibre were harvested from live animals by local communities in 2016 while a private international company obtained 980 kg, from a total of 2,754 Vicuñas; 143 kg of fibre were obtained from captive Vicuñas and 1,762 kg of fibre exported (Acebes et al., 2018) (it is assumed that these refer to semi-captive populations).*

Precautionary Measures

Species likely to be in demand for trade, but its management is such that the CoP will be satisfied with:

A) Implementation by the range States of the requirements of the Convention, in particular Article IV; and

B) appropriate enforcement controls and compliance with the requirements of the Convention

Export quota or other special measure

The population is intended to be covered by existing annotation number 1 which applies to all other Appendix-II Vicuña populations:

1

For the exclusive purpose of allowing international trade in fibre from vicuñas (Vicugna vicugna) and their derivative products, only if the fibre comes from the shearing of live vicuñas. Trade in products derived from the fibre may only take place in accordance with the following provisions:

a) Any person or entity processing vicuña fibre to manufacture cloth and garments must request authorization from the relevant authorities of the country of origin (Countries of origin: The countries where the species occurs, that is, Argentina, Bolivia, Chile, Ecuador and Peru) to use the "vicuña country of origin" wording, mark or logo adopted by the range States of the species that are signatories to the Convention for the Conservation and Management of the Vicuña.

b) Marketed cloth or garments must be marked or identified in accordance with the following provisions:

i) For international trade in cloth made from live-sheared vicuña fibre, whether the cloth was produced within or outside of the range States of the species, the wording, mark or logo must be used so that the country of origin can be identified. The VICUÑA [COUNTRY OF ORIGIN] wording, mark or logo has the format as detailed below:

This wording, mark or logo must appear on the reverse side of the cloth. In addition, the selvages of the cloth must bear the words VICUÑA [COUNTRY OF ORIGIN].

ii) For international trade in garments made from live-sheared vicuña fibre, whether the garments were produced within or outside of the range States of the species, the wording, mark or logo indicated in paragraph b) i) must be used. This wording, mark or logo must appear on a label on the garment itself. If the garments are produced outside of the country of origin, the name of the country where the garment was produced should also be indicated, in addition to the wording, mark or logo referred to in paragraph b) i).

c) For international trade in handicraft products made from live-sheared vicuña fibre produced within the range States of the species, the VICUÑA [COUNTRY OF ORIGIN] - ARTESANÍA wording, mark or logo must be used as detailed below:

d) If live-sheared vicuña fibre from various countries of origin is used for the production of cloth and garments, the wording, mark or logo of each of the countries of origin of the fibre must be indicated, as detailed in paragraphs b) i) and ii).

e) All other specimens shall be deemed to be specimens of species listed in Appendix I and the trade in them shall be regulated accordingly

In the CITES Trade Database there is generally very good agreement between quantities reported by Argentina and those reported by importing countries, indicating a well-functioning control system. No concerns have been raised within CITES regarding the export of products from Appendix-II listed populations of Vicuña in Argentina. It is not expected that transfer of the population of Salta Province would pose any additional threat to the populations remaining in Appendix I.

Additional Information

Threats

The main threat to the Vicuña is poaching, however, this species also faces threats from mining activities and, to a lesser extent, the introduction of exotic species for livestock, habitat alterations due to increased tourist activity, the extraction of water for mining and water pollution.

The species is affected by competition for grazing by domestic livestock and by some human-wildlife conflict from pastoralists who regard the species as a competitor. There is also some illegal killing for fibre for domestic markets. However, it is not regarded as threatened in Salta Province or in Argentina or in its wider range as a whole (Acebes et al., 2018).

Conservation, management and legislation

There are several conventions and laws, international, national and provincial that protect the Vicuña and regulate the trade in live animals, as well as the products and by-products obtained from this species. Due to the adoption of the federal government system, Argentina has regulated wildlife resource use at two levels, a provincial level and a national level.

Resource use and trade of Vicuña is protected nationally under Article 124 of the National Constitution (modified in 1994), the National Law of the Conservation of Wild Fauna No. 22.421 its regulatory Decree 666/97 and under Article 1947 of the Civil and Commercial Code of the Nation Law 26.994.

Provincially, the Vicuña is protected under the Law of the Province of Salta Nº 6.709/93, a Cooperation Agreement between the former General Directorate of Renewable Natural Resources, the current Ministry of Environment and Sustainable Development and the VII “Salta” Group of the National Gendarmerie and the Law of the Province of Salta 7.070/99 of Environment and 5.513/79 of the Protection of Fauna.

Internationally, the Vicuña is protected under the Agreement for the Conservation and Management of the Vicuña, which Argentina became a signatory of in 1971, other member states include Bolivia, Peru, Chile and Ecuador. Conservation measures include the creation of numerous protected areas in all countries that cover the distribution of the Vicuña, the development of joint strategies among signatory countries to control poaching and prohibit the export of fertile animals or other reproductive material and avoid breeding this species for commercial purposes outside of its range.

The wild population of Jujuy was transferred to Appendix II of CITES in 1997 and that of Catamarca in 2003 and semi-captive populations of all provinces, including the semi-captive population of the Salta Province, are also currently included in Appendix II.

Other comments

The proposal notes that Vicuña management will be performed by local indigenous communities from Andes, La Poma, Rosario de Lerma, Santa Victoria and Iruya departments who initiated an organizational process for the development of local conservation plans (PCL), in territories where land tenure is communal. The technical accompaniment in this process is in charge of an interdisciplinary and inter-institutional team of national and provincial organisms: Undersecretary of Family Agriculture, Secretariat of Environment and Sustainable Development of Salta and the National Institute of Agricultural Technology. The involvement of local communities for Vicuña conservation and management is key in such a vast area as the Puna (G. Lichtenstein, in litt., 2019).

References

Acebes, A., Wheeler, J., Baldo, J., Tuppia, P., Lichtenstein, G., Hoces, D., & Franklin, W. L. (2018). Vicugna vicugna, Vicuna. *The IUCN Red List of Threatened Species*, 2018.

Ianni, J. P., & Bernardos, J. (2018). *Informe técnico sobre abundancia de vicuñas en la provincia de Salta Años analizados : 2013 -2018*.

Lichtenstein, G. (2019). *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

Amend the name of the Vicuña *Vicugna vicugna* population of Chile from "population of the Primera Región" to "populations of the region of Tarapacá and of the region of Arica and Parinacota"

Proponent: Chile

Summary and Analysis: The Vicuña *Vicugna vicugna* is a South American member of the camel family that produces fibre of extremely high quality. Populations were heavily depleted in the mid-20th century mainly by hunting to obtain fibre for export. The species was consequently listed in Appendix I in 1975. In 1987, given the rapid recovery of the species, some populations in Peru and far northern Chile were transferred to Appendix II to allow the export of appropriately labelled fabric woven from fibre sheared from live animals. Other populations have followed suit, including in Argentina and Bolivia and a small introduced population in Ecuador. The current conditions for export, which set out how fibre should be obtained, and fabric labelled for export, are set out in annotation 1. In 2018 the total population was estimated at approximately half a million animals, and the Vicuña is currently classified by IUCN as Least Concern (Acebes *et al.*, 2018).

This proposal concerns the Chilean population of the Vicuña that is already listed in Appendix II. It concerns a technical change to ensure that the geographical description of the population accords with the current official Chilean terminology for the region. Until 2007, the whole of the northernmost part of Chile was referred to under Chilean law as the Primera Región of Tarapacá. All Vicuña in this region are included in Appendix II (under the description "population of the Primera Región"). In 2007 this region was split into two, one called Región de Tarapacá (Tarapacá Region) and the other Región de Arica y Parinacota (Arica and Parinacota Region). Vicuña occur in both these areas. The change in the geographical description ensures that it is clear that both these populations are still in Appendix II under annotation 1.

References

Acebes, A., Wheeler, J., Baldo, J., Tuppia, P., Lichtenstein, G., Hoces, D., & Franklin, W. L. (2018). *Vicugna vicugna*. The IUCN Red List of Threatened Species 2018.

Inclusion of Giraffe *Giraffa camelopardalis* in Appendix II

Proponents: Central African Republic, Chad, Kenya, Mali, Niger and Senegal

Summary: The Giraffe *Giraffa camelopardalis* is the world's tallest land mammal. It remains widespread across Southern and Eastern Africa, with smaller isolated populations in West and Central Africa. Nine subspecies are currently recognised, with each subspecies associated with particular sub-regions and/or range States.

In 2016, based on evidence of declines of 36–40% over three generations (30 years, 1985–2015), the IUCN Red List assessment was revised from Least Concern to Vulnerable. The best available estimates indicate a total population in 1985 of around 152,000–163,000 Giraffes (106,000–114,000 mature individuals), and in 2015 a total population of 98,000 Giraffes (68,000 mature individuals). The main factors responsible for this decline are recognised as habitat loss, illegal hunting (poaching), civil unrest and ecological changes. The presence and severity of these threats, and the conservation strategies used to manage Giraffe populations, show large regional variations.

In Central and Eastern Africa, Giraffes have suffered the greatest declines. Despite national protection, threats including habitat loss and illegal hunting—particularly for meat and some traditional uses—have severely reduced some populations over the last 30–40 years. These include declines of Reticulated Giraffe (*Giraffa camelopardalis reticulata* native to Kenya, Ethiopia, Somalia) of between 56% and 67%, Kordofan Giraffe (*G. c. antiquorum* native to Cameroon, Central African Republic, Chad, Democratic Republic of the Congo (DRC), South Sudan) of 85% and Nubian Giraffe (*G. c. camelopardalis* native to Ethiopia, South Sudan*)* of 97%.

In other regions, however, particularly in Southern Africa, Giraffe populations have undergone large increases in size. These include the Angolan Giraffe (*Giraffa camelopardalis angolensis* native to Botswana and Namibia) of 195%, and the South African Giraffe (*G. c. giraffa* native to Botswana, Mozambique, South Africa, Zambia and Zimbabwe*)* of 167%.

Available international trade data are restricted to USA import data, which along with Europe is considered a major market for trophies. Between 2006 and 2015, around 3,500 Giraffe trophies were imported to the USA, among around 40,000 total Giraffe specimens (largely bone products). Ninety-four percent of these products (and 98% of trophies) were exported by South Africa, Namibia and Zimbabwe, where trophy hunting is legal. There is no evidence to suggest exports from these countries were sourced from Giraffes illegally killed elsewhere. Non-trophy products are generally sourced from the trophy hunting industry, from natural deaths, or from animals culled or hunted for meat.

Conservation measures in both Namibia and South Africa have been associated with an increase in Giraffe populations over the last 30 years. While concerns have been raised over the management of Giraffe populations in Zimbabwe, which declined by 70% from around 26,000 in 1998 to 8,000 in 2016, this appears largely attributable to land reform programmes which have seen the conversion of land to agriculture, and an increase in poaching for local consumption. As the annual offtake for trophy hunting is less than 150 Giraffes (<2% of the population), this is considered unlikely to be negatively affecting Giraffe populations within Zimbabwe.

In some regions of Central and Eastern Africa, the illegal trade in Giraffe meat is known to cross porous borders, particularly where militia are in operation, while a transboundary trade in tail hairs may also occur, following centuries-long traditions. In some regions of Africa, Giraffe products, including Giraffe hair bracelets, have been recorded within tourist markets and may therefore be exported. Giraffe products are also seen for sale online in other markets, including Europe. There is no evidence to suggest that Giraffes are being harvested specifically in order to supply these markets (they are considered likely a “by-product” of the trophy industry, cropping and natural mortality) or that any significant international trade in products made from illegally killed Giraffes is occurring.

The poaching that has contributed to the decline of many Giraffe populations does therefore not appear to be driven by trophy hunting. The current levels of utilisation for trophy hunting in Southern Africa do not appear to be negatively impacting its regional populations of Giraffe, which overall are increasing.

Analysis: Although the Giraffe has experienced population declines of 36–40% over the last three generations, with illegal hunting having contributed to these declines, there is little evidence to suggest that the poaching of Giraffe is driven by international trade, rather it is for local/domestic use. The main populations that are subject to legal offtake for international trade are in Namibia, South Africa and Zimbabwe, where the hunting of Giraffe, mainly for trophies, and export is permitted, and populations are generally increasing, except in Zimbabwe where declines have not been attributed to international trade.

On this basis, it is not clear that regulation of trade is necessary a) to avoid the species becoming eligible for inclusion in Appendix I in the near future or b) to ensure that the harvest of specimens from the wild is not reducing the wild population to a level at which its survival might be threatened by continued harvesting or other influences. Regulation of international trade would also not address the principal threats affecting this species, with habitat loss, illegal hunting for either domestic use or to supply markets across porous borders within Africa, civil unrest and ecological changes, being the main causes of the observed decline in Giraffe.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Wilson and Reeder (2005) (the current CITES Standard Taxonomic Reference for mammals) and the IUCN Red List currently recognise a single species, *Giraffa camelopardalis*. Nine subspecies are also currently recognised, based on morphology *(see Table 1).*

Recently, several authors have proposed more than one species be recognised. For example, based on genetic analysis, Fennessy et al., *(2016a) and Winter* et al., *(2018) proposed four species, indicating that the taxonomic status of Giraffes may change in the near future.*

Range

Native: Angola; Botswana; Cameroon; Central African Republic; Chad; Congo; Democratic Republic of the Congo; Ethiopia; Kenya; Mozambique; Namibia; Niger; Somalia; South Africa; South Sudan; Tanzania, United Republic of; Uganda; Zambia; Zimbabwe.
Regionally extinct: Eritrea; Guinea; Mali; Mauritania; Nigeria; Senegal.
Introduced: *Eswatini; Rwanda (Muller* et al., *2018; J. Fennessy* in litt., *2019)*

IUCN Global Category

Vulnerable A2acd (2018) ver 3.1

<u>Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)</u>

A) Trade regulation needed to prevent future inclusion in Appendix I

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influence

Giraffa camelopardalis was assessed in 2016 as Vulnerable on the IUCN Red List due to an observed, past (and ongoing) population decline of 36–40% over three generations (30 years, 1985–2015). *The best available estimates indicate a total population in 1985 of 151,702–163,452 Giraffes (106,191–114,416 mature individuals), and in 2015 a total population of 97,562 Giraffes (68,293 mature individuals) (Muller* et al., *2018).* In 2010, *Giraffa camelopardalis* was Least Concern.

The pattern of population trends across Africa is not uniform, with some Giraffe populations stable or increasing, while others are in decline (Muller et al., *2018). Each of these populations is subject to threats, conservation management strategies and levels of utilisation that are specific to their range State or region.*

As each subspecies occupies a different geographic area, the population trends of each are also reflective of these regional differences. Table 1 summarises the current conservation status of the nine subspecies.

Table 1. Overview of the status of *Giraffa camelopardalis* subspecies.

Subspecies	Region of Africa	Range States	Status	IUCN Red List Subspecies Classification	Historic population estimate/Yr	Recent Population Estimate – Total Individuals/ (Yr)	Change in Pop. Size (No. of individuals)	% Change
G. c. peralta	West	Niger	Increasing	VU	50 (1990s)	607 (2017)	557	+1114%
G. c. antiquorum	Northern / Central	Cameroon, Chad, Central African Republic, DRC, South Sudan	Decreasing	CR	13,704 (1975-1986)	1,942 (2018)	-11,762	-85%
G. c. camelopardalis	Northern / Eastern	South Sudan, Ethiopia	Decreasing	CR	20,577 (1979-1982)	650 (2018)	-19,927	-97%
G. c. reticulata	Eastern	Kenya, Ethiopia, Somalia	Decreasing	EN	36,000-47,750 (1990s)	15,785 (2018)	-20,215 – 31,965	-56.2%-67%
G. c. rothschildi	Eastern	Uganda, Kenya	Increasing	NT	1,330 (1960s)	2,098 (2018)	768	+57%
G. c. tippelskirchi	Eastern	Kenya, Tanzania	Decreasing	Not assessed	66,449 (1977-1980)	31,611 (2015)	-34,838 (2015)	-52%
G. c. thornicrofti	Eastern	Zambia	Stable	VU	600 (1983)	600 (2018)	0	0%
G. c. angolensis	Southern	Botswana, Namibia	Increasing	LC	5,000 (1970-2004)	14,748 (2018)	9,748	+195%
G. c. giraffa	Southern	Botswana, South Africa, Zambia, Zimbabwe,	Increasing	Not assessed	8,000 (1979)	21,387 (2016)	13,387	+167%

Sources (historic): Fennessy et al., *(2018a), Fennessy and Marais (2018a), Wube* et al., *(*in prep*, 2016), East (1999) Fennessy* et al., *(2018b). Bolger* et al., *(*in prep*, 2015) Bercovitch* et al., *(2018) Marais* et al., *(*in prep*, 2016) Dagg and Foster (1982).*
Sources (recent): Fennessy et al., *(2018), Fennessy and Marais (2018a), Wube* et al., *(2018), Muneza* et al., *(2018), Fennessy* et al., *(2018b). Bolger* et al., *(*in prep, *2015), Bercovitch* et al., *(2018), Marais* et al., *(2018), Muller* et al., *(2018).*

Types of international trade

Giraffe are known to be trophy hunted in four Southern African countries: Namibia, South Africa and Zimbabwe as well as Zambia, and as such the international trade in their body parts (trophies, bones, skeleton, carvings, and hair) from these countries legally occurs (J. Fennessy in litt., *2019). There is also domestic consumption and associated trade within Africa for a variety of purposes.*

As Giraffa camelopardalis *is not listed within CITES, the only available data detailing imports of Giraffe products originates from the U.S. Fish and Wildlife Law Enforcement Management Information System trade database (LEMIS).*

The following section draws on the information in the SS as well as our own rapid assessment of LEMIS data from 2006–2014. Note the time period differs by a year. The USA is probably one of the major destinations for trophies but Europe is also likely to be a key destination.

International trade with the USA:

Between 2006 and 2015, 39,516 Giraffe specimens (Giraffes, dead or alive, and their parts and derivatives) were imported into the USA. Of these, 99.7% were classified as wild sourced (39,397 of 39,516). The proposal states that of these USA imports, around 95% were for hunting trophy purposes. The SS reports that about 95% of individual Giraffes imported to the USA from 2006 to 2015 were for hunting trophy purposes (5,044 Giraffe specimens, representing at least 3,563 individual Giraffes including 3,561 trophies, 1 body, and 1 live animal; comparing the estimated 3,563 individual Giraffes imported for hunting trophy purposes to the estimated 3,751

individual Giraffes imported for all purposes). The principal exporters of these hunting trophies were: South Africa (3,065 or 60.8%), Zimbabwe (1,346 or 26.7%), and Namibia (575 or 11.4%).

Our own analysis of US import data over a shorter period (2006–2014) shows different levels of trophy imports but this may be due to different analysis approaches (see Table 2). This also shows small amounts of trophies exported by Botswana and Zambia during the period.

The SS states that, since 2010, there has been a marked increase in the number of trophies being imported into the USA (based on a low of 276 in 2010, to a high of 457 in 2015). *The total number of trophies compared to the size of the Giraffe populations in the principal exporting range States however, remains very low, (W. Crosmary,* in litt., *2019; J. Fennessy,* in litt., *2019). Populations of the Giraffe subspecies within these trophy exporting range States have also undergone large increases during the last four decades, of 195% (*G. c. angolensis*) and 167% (*G. c. giraffa*) (Muller* et al., *2018).*

Table 2. Source of imports of trophies to the USA between 2006–2014 (all purpose codes included, includes items rejected or seized).

	South Africa	***Zimbabwe***	***Namibia***	***Botswana***	***Zambia***	***Ethiopia***	***Non-Africa Countries***	***Grand Total***
Total quantity of trophies exported	*1,838*	*832*	*441*	*16*	*4*	*1*	*9*	*3,141*
Of which, quantity of trophies originating elsewhere	*30 (18 ZW, 7, NA, 2, BW, 2, MZ, 1 SZ)*	*17 (ZA)*	*3 (2 ZA, 1 MA)*			*1 (ZW)*	*(9 ZA)*	

Between 2006 and 2015, at least 33,321 other (non-hunting trophy) specimens made from Giraffe parts and derivatives were also imported into the USA for commercial purposes. The SS suggests that this is the equivalent of at least 157 individual Giraffes, *although the method of calculating this figure is not provided.* The most common of these items were: bone carvings (20,885), bones (3,768), skin pieces (2,820), bone pieces (1,857), skins (715), jewellery (766), shoes (526), hair (487), small leather products (314), feet (117), large leather products (138), and horn (ossicone) carvings (200).

Table 3 shows the majority of non-hunting exports were from South Africa and Zimbabwe.
The SS also reports 693 specimens as exported from Tanzania, "representing at least one Giraffe". *An additional 1,651 items of jewellery were imported from Taiwan, Province of China.*

Table 3: *Imports of a selection of products into the USA between 2006–2014. Selection based on largest quantities reported (e.g. bone related products (*including bones, carvings and bone carvings, bone pieces and bone products), jewellery, leather products, hair products) and items that might indicate harvest levels (skeletons, bodies, skulls, rugs, tails and live) (all purpose codes included and includes items seized or rejected).*

	Imported from								
Product	**South Africa**	**Zimbabwe**	**Kenya**	**Namibia**	**Zambia**	**Tanzania**	**Botswana**	**Nigeria**	**Non-range States**
Bone related products*	21,288	487	125	86	1				641
Skeletons	55								3
Bodies	4								1
Skulls	127	16		3					10
Skins	120	517		3			1	1	34
Rugs	58	15							1
Jewellery	65	55	3			690			1,651 (all TW)
Leather products large and small	196	190		1					1
Hair and hair products	166	400		5					35
Tails	14	38		3					2
Live									4

Trade within Europe:
It is also likely that Europe is a key importer of trophies. However, no import or export data are available for the levels of trade into Europe.

Between 2016 and 2018, volunteers conducted research into online markets in Europe with the following results. Within European Union (EU) countries, the most numerous Giraffe products available for sale online were: bone scales (170), knives with bone handles (82), bone carvings (10), skins (10), taxidermy busts (9), skulls (6), chef's knives with bone handles (6), and skin pieces (4). *As import data are lacking however, the provenance and age of these items remains unknown:*

- Belgium (search conducted 30th July 2018): 10 products including full body taxidermy, taxidermy busts, bone knife handles.
- France (13th and 14th June 2018): 58 Giraffe products, including 48 bone knife handles, one bone, one pen with a bone part, one set of four Giraffe feet taxidermy, one table featuring four Giraffe legs, two skulls, two revolvers with bone grips, one skin and one pair of bone scales.
- Germany (7th July 2018): 51 Giraffe products, including seven raw bones, seven full skins, four skin pieces, two taxidermy busts, one tail, one hoof, two taxidermy heads, four skulls, three pairs of bone scales, six knives with bone handles, one table featuring four Giraffe legs, 10 bone carvings, and one pen with a Giraffe bone part.
- Greece (30th July 2018): One Giraffe product—a knife with a bone handle.
- Italy (30th July 2018): 18 Giraffe products, including seven knives with bone handles, one taxidermy bust (trophies), two sets of bone scales, two chef's knives with bone handles, and one large hunting knife with a bone handle.
- Spain (11th July 2018): 171 Giraffe products being sold by sellers based in Spain, including seven knives with bone handles, 163 sets of bone scales, and one full skin.
- United Kingdom (23rd July 2018): 21 Giraffe products being sold by sellers based in the United Kingdom, including 11 knives with bone handles, four chef's knives with bone handles, three taxidermy busts (trophies), one pair of bone scales, one full hide, and one hair bracelet.

Transboundary trade within Africa:
Some transboundary trade in Giraffe products, particularly meat, has been recorded in Central Africa, which is likely to have had a significant negative impact on some populations of Giraffe, including the Critically Endangered Kordofan Giraffe Giraffa camelopardalis antiquorum *(Fennessy & Marais, 2018; Ondoua* et al., *2017). This cross-border trade in wild meat is thought frequently to result from civil unrest and the action of militias and highly-militarised poachers (see Nouredine, 2012, in Marais* et al., *2013; Ondoua* et al., *2017), although a small trade in Giraffe tail hairs may also occur, following centuries long traditions (J. Fennessy,* in litt., *2019). Okello* et al., *(2015), also states that poaching of Maasi Giraffe* G. c. tippelskirchi *occurs within borderland regions of Kenya and Tanzania, with Giraffe meat transported on donkeys across borders to market places and other demand areas for sale. Other examples include trade between Central African Republic and South Sudan/Chad (Bouche* et al., *2009); between Cameroon and Chad/Nigeria (see IUCN PACO 2011, as cited in Marias* et al., *2013); and within the Garamba-Bili-Chinko landscape of the Demographic Republic of the Congo and Central African Republic (Ondoua* et al., *2017). However, as this trade in meat operates across highly porous borders, it seems extremely unlikely that a CITES listing would result in any additional enforcement effort being directed at controlling this illegal activity.*

*Crosmary (*in litt., *2019) also notes that in countries where trophy hunting is legal, "meat and other body parts may be distributed or sold to villagers after the trophy hunts. It is possible that some of these body parts are also sold into neighbouring countries, as happens with other trophy hunted species such as lions".*

Provenance of Giraffes in international trade:
The SS suggests that the possibility exists for parts and derivatives from illegally hunted Giraffe to enter established legal trading routes. *However, research into the trade in Giraffe products has been limited (J. Fennessy* in litt., *2019).*

*Many parts and derivatives entering the legal trade are thought to be sourced from the trophy hunting industry itself. Wolfson (2018), found evidence of trophy hunting by-products being sold on to taxidermists and outfitters, and describes how virtually every part of the animal is utilised to make products for sale, which are marketed to sellers in the USA. The Management Authority of Zimbabwe (*in litt., *2019), also confirms that "most parts and derivatives in international sale are sourced from the trophy hunting industry."*

Giraffe parts and derivatives entering legal trade are also from animals acquired via other sources, including natural deaths, and Giraffes killed by collisions with low power lines (Management Authority Zimbabwe, in litt.,

2019). It is also likely that Giraffe parts identified for sale may come from Giraffes that have been culled or killed for meat (J. Fennessy in litt., *2019). This does not however, imply that illegally killed giraffe are entering trade. As Fennessy (*in litt., *2019), notes, "the misinterpretation that it is all about trophy hunting is inaccurate", but "without any additional information, it is assumed that all products come originally (and legally) from Southern Africa*."

A small number of products may enter international trade through tourist markets, with, for example, Giraffe hair bracelets found for sale in Maputo, Mozambique *(TRAFFIC, 2002), and bone products sold at tourist markets in Namibia (Du Raan* et al., *2016). However, Crosmary (*in litt., *2019), reports that "in East Africa, at least in Kenya, Uganda and Tanzania, there is no evidence of any significant international trade, and for the other regions, there is no information."*

Trade in live Giraffes:
A trade in live specimens also exists, between game ranches in Southern Africa*, and also between Africa and Asia (J. Fennessy* in litt.*, 2019), and which may be greater than previously assumed (J. Fennessy* in litt., *2019). Deacon and Parker (2016), note that South African Giraffes (*Giraffa camelopardalis giraffa*) have been moved from South Africa to Zambia and Senegal, while Angolan Giraffes (*G. c. angolensis*) have been transferred from Namibia and Botswana to South Africa. Without accurate data, however, the impact of live trade on Giraffe populations cannot accurately be assessed. Fennessy (*in litt., *2019) notes that live Giraffes are not imported from East, Central or West Africa into Southern Africa, and as such the possibility that these populations are entering the trophy hunting industry is not a concern.*

Impacts of Trade
Trophy prices range from USD 2,500–8000 in South Africa, USD 1,800–3,100 in Namibia, and USD 3,200 in Zimbabwe. The overall cost of a hunt is, however, much higher, and Giraffes are not usually the primary target, and instead are an additionally hunted species (J. Fennessy in litt., *2019). As indicated by the data in Table 1, in the range States where this legal hunting occurs, and which export the vast majority of Giraffe products (Namibia, South Africa and Zimbabwe), Giraffe populations have generally increased during the last 30 years. Further information about the Giraffe populations in countries that permit trophy hunting is presented below:*

South Africa: *In South Africa, total numbers and area of occupancy within South Africa are expanding, due in part to the game ranching industry, where they are highly favoured for their added tourism value (Deacon & Parker, 2016).*
Trading of Giraffe, including for ecotourism, live sales and hunting is controlled by each province's nature conservation office (Deacon & Parker, 2016).

In 2016, populations of the G. c. giraffa *within South Africa were assessed as Least Concern. In 2016, the total population was estimated to be 11,746–15,024 mature individuals living within their natural habitat (on all land types), based on a total population size of between 18,645 and 22,094. The total number inhabiting national parks within their natural distribution (Kruger, Augrabies Falls, Mapungubwe, Marakele and Mokala National Parks) was estimated to be 4,896–7,533 individuals (Deacon and Parker, 2016). Data from a sample of 13 formally protected areas with long-term data over at least two generations show an estimated increase for those sampled populations of 54% (Deacon & Parker, 2016).*

Zimbabwe: *In Zimbabwe, Giraffes are found in State-owned protected areas, private and communal land (ZPWMA, 2019). Giraffe populations in Zimbabwe declined by 70% from around 26,000 in 1998, to 8,000 in 2016 (Fennessy* et al.*, 2016b), but this appears largely attributable to Land Reform programmes which have seen the conversion of wildlife producing land to agriculture, and an increase in poaching for local consumption (Bond* et al., *2004; Degeorges & Reilly, 2007; Williams* et al., *2016).*

The population is reported to be increasing in parts of southern and western Zimbabwe (ZPWMA, 2019) although no recent assessments have been undertaken (Deacon & Parker, 2016; J. Fennessy, in litt., *2019). The Zimbabwe Management Authority states that the current annual trophy quota within Zimbabwe is for 800 Giraffes (10% of the population). These are, however, optional rather than fixed quotas, and around 150 (<2% of the population) permits per year are actually utilised. The <2% offtake is unlikely to be a significant factor affecting the population, and there is no evidence to suggest that the reduction in Zimbabwe's Giraffe population over the last two decades is attributable to international trade (J. Fennessy,* in litt., *2019). Trophy imports to the USA in Table 2 above showed an average of 92 over the years 2006–2014.*

For all hunted species in Zimbabwe, the hunting of mature old males is recommended, and if young males are harvested, the operators are penalised (Zimbabwe Scientific Authority, in litt., *2019). In a 2019 Giraffe status report, the Zimbabwe Parks and Management Authority also states that Giraffe are valued as "an attractive*

*species in photographic safaris" and that efforts are being made to research the conservation status of the species and to ensure long-term monitoring of population trends***.**

Namibia: *In Namibia, Giraffes mainly exist in protected areas, which may be public, private or communal (Du Raan* et al., *2016), and are valued as a popular species on private game farms because of their high tourism value (MET, 2019). While a lack of standardised methods has created challenges in calculating population estimates, Fennessy (2004) estimated the population of Giraffes in Namibia to be around 5,000 individuals (although a national wildlife inventory completed in 2004 estimated it at 10,415 individuals (Barnes* et al., *2009)). The current population of Giraffes in Namibia is estimated to be around 12,000 animals of which 6,500 occur on privately owned land, 2,000 on communal land, and 3,500 within national parks (Du Raan* et al., *2016). The population is thought to be increasing on all land types (J. Fennessy,* in litt., *2019), with above average rainfall and increased conservation awareness and monitoring thought to have contributed to this observed increase in numbers (MET, 2019).*

With regard to trophy hunting, the Namibian Ministry of Environment and Tourism reports that the country has a well-established and strictly controlled wildlife utilisation system, i.e. no person may hunt any species without a permit (MET, 2019). They also report that between 2010 and 2017, a total of 1,115 Giraffes have been trophy hunted in Namibia, at an average of 139 Giraffe per year. Of these 1,115 animals, 96% (n = 1,068) were reportedly hunted on private land, and 4% (n = 47) on State land. (table 4). Some hunting on communal land by conservancies is also known to occur for both trophies and meat for local consumption (J. Fennessy, in litt., *2019).*

Table 4. *Number of Giraffes hunted in Namibia (2010–2017)*

Year	***2010***	***2011***	***2012***	***2013***	***2014***	***2015***	***2016***	***2017***
Number	*85*	*168*	*142*	*143*	*120*	*131*	*162*	*164*

Data indicate that the USA is the destination for some of these trophies. It is also likely that the EU, and in particular Germany, also serves as another export destination (J. Fennessy, in litt., *2019). Within Namibia, Giraffe bones are also sold at local tourist markets (Du Raan* et al., *2016), and Giraffe hair in jewellery stores (J. Fennessy,* in litt., *2019).*

Zambia: *At present, and as reflected by the US Trade data described above, very little trophy hunting of Giraffes takes place in Zambia**,** and Zambia is not thought to supply another international market beyond the USA (Fennessy* in litt., *2019). Chomba and Nyirenda (2015) make no reference to Giraffes when analysing the levels of safari hunting utilisation for 40 other species within the country. The country's population of Thornicroft's Giraffe (*G. c. thornicrofti*) is stable, while its other resident subspecies (*G. c. giraffa*, in the south west of the country), is not threatened by illegal hunting in Zambia (J. Fennessy,* in litt., *2019) and is generally increasing within Southern Africa.*

Botswana: *In the 1990s, Botswana adopted a Community Based Natural Resource Management (CBNRM) programme, which initially focused on safari hunting as the main tourism activity (Mbaiwa, 2018). In 2014, however, Botswana banned safari hunting, which was no longer deemed part of the "national commitment to conserve and preserve local fauna or the long-term growth of the local tourism industry" (Scott, 2013; Mbaiwa, 2018). However, Scott (2013), states that Giraffes acted as a secondary species on Okavango hunts, and that offtake had been minimal, which appears to be corroborated by the low number of Giraffe trophies exported from Botswana to the USA between 2006–2014. Arntzen* et al., *(2003), reported that the population of Giraffes in one CBNRM case study region increased from an estimated 4,248 in 1989, to 7,217 individuals in 2001.*

Summary: *The general increase in Giraffe populations within Southern Africa, in range States where trophy hunting is permitted, suggests that this legal offtake of Giraffe is not adversely affecting these populations or contributing to the overall decline of the species. Equally, there is no evidence to suggest that the legal trade in Giraffe trophies has played a significant role in the population declines in Zimbabwe from 1998 to 2016. These observed increases are due to a range of factors including climatic conditions and increased conservation awareness, although in part, are also likely related to changes in policy regarding the legislation and ownership of wildlife, and subsequent benefit of their use (J. Fennessy* in litt., *2019).*

In areas of Africa where legal hunting is not permitted, however, Giraffe populations have generally declined, due to a range of factors. While illegal hunting (poaching) remains a serious ongoing threat to Giraffes, no evidence currently exists of a connection between this and any significant form of international trade, other than the suspected transfer of illegally killed animals across porous borders within some parts of Africa (see Threats for more detail). Finally, while trophy hunting is predominantly focused on adult male specimens, there does not

*appear to be any evidence to suggest that this selective hunting is negatively impacting the genetic integrity of Giraffe populations. Fennessy (*in litt., *2019), did not consider this to be a research priority, as "trophy hunting is minimal, and most trophy hunts are on small private land which have a limited genetic diversity anyway."*

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

Products that make use of the hide could potentially be distinguished due to the Giraffe's distinctive pelage, although there is currently no method of easily identifying Giraffe bone to species level, in some of the forms in which it is traded (J. Fennessy, in litt., *2019).*

Additional Information

Threats

Four main threats have been identified as being responsible for the decline of some Giraffe populations: habitat loss, civil unrest, illegal hunting and ecological changes. *These threats have not ceased and may not be reversible throughout the species' range. Their presence and severity, however, varies by region (Muller* et al., *2018).*

In West Africa, the main threats are habitat loss due to increasing human populations and human–wildlife conflict. In East and Central Africa, the main threats are habitat loss through rapid conversion of land for farming and increasing human populations, drought, illegal hunting for meat and hide, and armed conflict throughout unstable regions.

In Southern Africa, the main perceived threats are illegal hunting and habitat loss caused by conversion of land for agriculture, human development, and cutting of trees for firewood and construction. The poaching of Giraffes is, however, considered less of a threat in Southern Africa than in Eastern and Central regions.
Two of the most chronic threats—habitat loss and illegal hunting—are considered in more detail below.

Habitat loss:

Expansive habitat is a prerequisite for healthy Giraffe populations, given their relatively large home ranges—which average between 68 km² and 514 km²—and their seasonal migration patterns. Across its range however, *Giraffa camelopardalis* has experienced severe habitat loss and fragmentation, due to: increased human settlement; expansion of agricultural activities; conversion of land to industrial plantations; the uncontrolled harvesting of timber and wood for various uses, including firewood, logging, and charcoal production for both personal and commercial purposes; and poor land use planning. As a result, the Giraffe's range has contracted significantly over the past century. This has resulted in geographical isolation of local populations and some herds surviving at the edge of the species' preferred range. In addition, habitat loss due to aridity may be compounded by climate change in the future. *Lado (2019) reports that in South Sudan, political instability and armed conflict have forced many people to seek refuge and grazing areas within in the preferred habitat of this species. In Ethiopia, the cutting of Acacia trees (an important food source for G. camelopardalis) for charcoal is thought to be a major factor contributing to declines, along with habitat degradation due to expansion of agriculture (Abate & Abate, 2017).*

Illegal Hunting:

Giraffa camelopardalis is targeted by illegal hunters particularly in Central and Eastern regions (Muller et al., *2018). In some areas, such as South Sudan, this is the main threat to Giraffe populations (Lado, 2019). In addition to the actions of militias noted in the trade section of this report, other examples include* the poaching of Giraffe in Botswana's Okavango Delta, *where an estimated 99 animals are killed each year*, representing 12% of illegal meat harvest *(by weight),* sourced from the region *(Rogan* et al., *2017).* Poaching of Masai Giraffe (*G. c. tippelskirchi*) is also common in protected areas in Tanzania and poaching may have caused certain populations in the country to be designated as "population sinks". *In Botswana, meat hunting for local use may be increasing, a trend that many suggest is potentially correlated with the socioeconomic impacts of hunting bans for other animals within the country (J. Fennessy,* in litt., *2019). In Zambia, Giraffes are not thought to be threatened by direct illegal hunting (J. Fennessy*, in litt., *2019), although Thornicroft's Giraffe (*G. c. thornicrofti*) have occasionally been caught in snares that are probably aimed at other animal species (Du Raan* et al., *2015).*

Giraffes targeted by illegal hunters are valued for medicinal, magical *and practical* purposes. For example, Giraffe products may be used as aphrodisiacs or headache cures, *to treat nose bleeds (Abate & Abate, 2017),* and, in Tanzania, as a supposed cure for HIV/AIDS. Giraffe hair and tails *are also* valued for use as *flyswatters, bracelets, necklaces and thread, while their skin is used for shields, sandals and drums (Abate and Abate, 2017). In Ethiopia, Giraffes have been hunted with automatic rifles by local people for their tails (and meat), which are used to make traditional for wedding dowries (Wube, 2013).*

Giraffes are also hunted for meat for human consumption (Muller et al., *2018). As noted above,* s*ome transboundary movement of meat may occur, while in some areas (including within Kenya), local markets have been identified as places of trade for poached Giraffes (Muneza & Fennessy, 2017). In some areas of East Africa, however, illegal hunting appears to be less prevalent. For example, the Management Authority of Uganda (*in litt., *2019) reports that there is no evidence of targeted Giraffe poaching within Uganda, although* Giraffa camelopardalis *are occasionally caught in snares and traps targeting other herbivorous species.*

Conservation, management and legislation
Conservation management strategies for Giraffe and the extent of State protection varies across Africa. Populations may be privately owned and managed or may inhabit protected areas where they are owned by the State (J. Fennessy in litt., *2019).*

Many range States have laws prohibiting the hunting of Giraffes, including Angola, *Botswana,* Cameroon, Central African Republic, Chad, DRC, Kenya, Mozambique, Niger, Rwanda, South Sudan, *Tanzania,* and Uganda. *Despite these protections, however, the success of Giraffe conservation across Africa is highly variable and declines in Giraffe populations have occurred in many of the countries noted above (Muller,* et al., *2018). As noted in the trade section above, Giraffes occur in a number of other range States where hunting is permitted. The principal countries where trophy hunting occurs on this basis are Namibia, South Africa and Zimbabwe. While national Giraffe management plans in these countries have not yet been developed (Deacon & Parker, 2016; Zimbabwe Scientific Authority,* in litt., *2019; Namibian Management Authority, 2019), hunting is only permitted following the issuance of a government permit (J. Fennessy,* in litt., *2019).*

In Namibia, Giraffes are listed under National Legislation as "Specially Protected Game", which prevents the hunting of Giraffes or the trading of Giraffe products, unless a permit is issued (MET, 2019). Due to a steady increase in the Giraffe population since the 1970s, no specific Giraffe management plan has been put in place (Namibian Management Authority, 2019).

In Zimbabwe, there is also currently no management plan in place for Giraffes (Zimbabwe Scientific Authority, in litt., *2019). The Management Authority also states that efforts are being made to consolidate available information, and to research the conservation status of the species to ensure long-term monitoring of population trends***.**

International agreements:
The Giraffe was recently listed on Appendix II of the Convention on Migratory Species (CMS). While 31 countries have ratified the Convention, many with Giraffe populations have not, including Angola, Chad, Ethiopia, Namibia, Somalia, South Africa, South Sudan, and Zimbabwe.

Artificial propagation/captive breeding
Giraffes have been bred in zoos, but there is no evidence of commercial breeding operations, and *captive breeding is not considered to have any relevance to the conservation of this species (J. Fennessy,* in litt., *2019).*

Implementation challenges (including similar species)
While whole Giraffe trophies are readily identifiable, smaller parts and derivatives may be more cryptic. There is currently no method of easily identifying Giraffe bone to species level in some of the forms in which it is traded (J. Fennessy, in litt., *2019).*

Potential risk(s) of a listing
A listing could conceivably result in a fall in demand for Giraffe trophies, due to potential difficulties in obtaining a CITES permit, or the additional stigma that this may attach to the sport (J. Fennessy in litt., *2019). Given that Giraffe numbers have generally increased in several Southern Africa range States in the last 30 years where hunting is permitted, it could be argued that the value attached to Giraffes through trophy hunting has contributed to this increase in population numbers. A fall in demand and consequent decrease in the value of Giraffes to landowners might conceivably contribute to a decrease in Giraffe ownership, and a subsequent decrease in the populations of Giraffe within these countries.*

Potential benefit(s) of listing for trade regulation
An Appendix II listing would produce data that would provide a clearer picture of a). the full extent of international trade, and b). the provenance of some Giraffe products in trade. It would also provide a more detailed picture of the extent of trade in live specimens.

However, as there is currently no scientific basis to suggest that the current levels of international trade pose a threat to Giraffe populations, this may not ultimately prove useful to the future conservation of the species.

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Transfer of the Small-clawed Otter *Aonyx cinereus* from Appendix II to Appendix I

Proponents: India, Nepal and the Philippines

Summary: The Small-clawed Otter *Aonyx cinereus* is the smallest of the otter species. The species has a broad range extending from India eastwards through South-east Asia to southern China. It is dependent on aquatic habitats for foraging and sheltered terrestrial areas for resting and denning. It occurs in a range of aquatic habitats from coastal wetlands to mountain streams, and in at least some human-modified habitats such as rice fields and coffee/tea plantations wherever there is prey and adequate shelter.

This species was assessed as Vulnerable in 2014 on the IUCN Red List. The assessment states that although quantitative data on population sizes or trends are lacking, it is inferred that the global population of *Aonyx cinereus* has declined by greater than 30% over the past 30 years (three generations). However, as *A. cinereus* was assessed as Vulnerable and not Endangered, declines of greater than 50% were not indicated. There are no current population estimates available for *A. cinereus*. Although populations and habitat are believed to be stable in parts of the range, its distribution in the west is believed to be contracting, and it is now considered to be very rare in southern China and Myanmar. Its population status is reportedly unknown in a number of countries (Bhutan, Cambodia, Lao People's Democratic Republic (PDR), Thailand and Viet Nam). Where national Red List assessments have been undertaken, the status varies from data deficient in Nepal to near threatened in Malaysia to endangered in Bangladesh. However, in other parts of its range there are healthy populations and habitat.

The subfamily Lutrinae, which includes *Aonyx cinereus,* has been included in Appendix II since 1977. Poaching is considered a significant threat; *A. cinereus* has a history of exploitation for its fur and for body parts used in Traditional Asian Medicine, which has been identified as one of the main causes of historical population declines. Live trade for the pet trade and otter-petting cafes is said to be an emerging use with Japan and Thailand identified as destinations. CITES records of legal international trade show relatively low volumes, mostly of live animals (*ca*. 600 between 1980–2017) reported mainly as from captive sources. Many reported consumer countries are also range States so some trade is likely domestic. While online advertisements of live *A. cinereus* often describe the otters as captive-bred, it is believed that that many animals in trade are wild-caught. There are concerns that illegal pet trade in otters in general is a growing threat and there is evidence to suggest that this species is the most in demand. Live *A. cinereus* have been offered for sale online in Indonesia, Thailand and elsewhere. The total level of trade in this species for pelts, pets and medicine is unclear as much of the trade is apparently illegal and unreported.

The species is protected in all range States except Brunei Darussalam, Cambodia, Indonesia and Nepal, although protection may vary in form and enforcement. For example, in Thailand whilst the possession of otters is prohibited, and all native otters are protected, online advertisements for *A. cinereus* and other otter species can still be found.

A number of otter seizures have been reported in range States, with some apparently destined for export; enforcement staff are reported to have difficulty in identifying pelts and products in trade to the species level, therefore seizures are often not reported to the species level.

There are reportedly otter farms in China, Pakistan and Indonesia, and although species can be bred in captivity, it is not clear how much of the trade is being met from these sources.

In addition to harvest, *Aonyx cinereus* are believed to be affected to some extent by widespread human development and activities such as habitat loss and degradation, pollution, and reduced prey base, in addition to climate change.

Analysis: Information on the status of *Aonyx cinereus* is scarce although the population is considered unlikely to be small or to have a restricted range. There is anecdotal information that the species is scarcer than it was, and it has been extirpated in parts of its range, but in other areas populations are reported to be stable. There are no baseline population data on which to measure trends, but a recent (over three generations) decline of greater than 30%, but less than 50%, has been inferred from rates of habitat loss and exploitation resulting in an IUCN Red List assessment of Vulnerable (2014). Legal international trade in the species has been low, but there are concerns over the impacts of illegal harvest for pelts and more recently in an apparently growing demand for the pet market. It is not clear what proportion of harvest is for domestic versus international trade. While there is some evidence that *A. cinereus* can be successfully bred in captivity, it is unclear if any of the international or domestic trade, including the pet trade, is being met from captive sources. Given the available information, it is not possible to determine the overall level of harvest from the wild or its impact on the species.

On the basis of a population decline greater than 30% but less than 50%, inferred from a decline in habitat and exploitation in the Red List assessment, it seems uncertain that this species meets the guideline for a marked recent population decline as described in *Res. Conf. 9.24 (Rev. CoP17)* for inclusion in Appendix I at the present time. However, here are significant levels of uncertainty regarding status in some parts of the species' range and levels of trade, and if further information were to become available it may help to determine if the species is closer to the 50% decline guideline for inclusion in Appendix I.

Other Considerations: Levels of legal international trade appear low, and therefore it is assumed that most harvest is for domestic and/or illegal trade. Any additional benefits of an Appendix I listing are not clear unless enforcement efforts are increased.

Res. Conf. 12.10 (*Rev. CoP15*) outlines that inclusion in Appendix I would mean commercial captive breeding operations would need to meet the provisions of *Res. Conf. 10.16* (*Rev.)* to be registered with the CITES Secretariat, and that registered operations should ensure an appropriate and secure marking system to identify all breeding stock and specimens in trade. This enhanced oversight could help allay concerns over fraudulent claims of captive breeding and wild offtake for breeding stock.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Range

The species is native to Bangladesh, Bhutan, Brunei Darussalam, Cambodia, China, India, Indonesia, Lao People's Democratic Republic, Malaysia, Myanmar, Nepal; Philippines (Palawan), Singapore, Taiwan Province of China, Thailand and Viet Nam, and is introduced to the United Kingdom.

IUCN Global Category

Vulnerable *A2cde (assessed 2014) ver 3.1.*

Biological criteria for inclusion in Appendix I

A) Small wild population

The current population of *Aonyx cinereus* is unknown but is likely not to be considered small given the species' wide range.

In the Sundarbans, Aziz (2018) estimated approximately 1 otter per 30 km of river surveyed which, the author suggests, should be considered preliminary and an underestimate of the actual otter population in the area. *Although this survey was preliminary initial findings it could indicate the presence of a healthy population of* Aonyx cinereus *in the Sundarbans and that this mangrove forest could signify a stronghold for this population (Aziz, 2018).*

It is estimated that there are other areas within its range that can also support healthy populations of *Aonyx cinereus*, including Nakai-Nam Theun National Protected Area in Lao PDR and Puerto Princesa City in Palawan, Philippines.

B) Restricted area of distribution

Aonyx cinereus has a broad distribution range, extending from India in South Asia eastwards through South-east Asia. *According to Wright* et al., *(2015) the species is native to 16 countries/territories.* However, they have disappeared or declined in many parts of their range.

C) Decline in number of wild individuals

This species is classified as Vulnerable due to an inferred past population decline because of habitat loss and exploitation. Although quantitative data on population sizes or trends are lacking, it is inferred that the global population of the Small-clawed Otter has declined by >30% over the past 30 years (three generations) (Wright *et al.,* 2015). *There is no generally accepted/agreed scientific method for most population counts and/or estimates in the wild for most species (although there are exceptions e.g. Lutra lutra see Reuther* et al., *2000), therefore population changes of most Asian otter species are inferred from habitat trends or presence/absence surveys. Wright* et al., *(2015) note range constrictions in the western part of the range (India) and the eastern part (Indochina). Country specific accounts are provided below*:

Bangladesh: Aziz (2018) suggests that a large population of *Aonyx cinereus* is probably present in the Sundarbans, Bangladesh, and that the mangrove forests probably provide a stronghold for the species' long-term survival.

In the National Red List assessment (2015), Aonyx cinereus *is categorised as endangered and the range for this species in the wild has decreased, particularly in the western portion of its range and its Area of Occurrence is estimated to be 4,114 km^2 (Begum, 2015).*

Bhutan: *A survey conducted along one of the major rivers in central Bhutan also confirmed the presence of otters and that it is likely* Aonyx cinereus*,* Lutrogale perspicillata *and* Lutra lutra *are all present in this country. Only* Lutrogale perspicillata *was directly observed during this study. A total of six 5.5 km transects were sampled and five of them were positive for otter signs and one site had the highest rate (95%) of positive plots. Over 40% of land in Bhutan has been designated as a natural protected area and hunting is considered a serious offence and is prohibited (Chettri & Savage, 2014). This study represented the first survey of otters in Bhutan and further research is required to assess their status in this country.*

Cambodia: Aonyx cinereus *has been reported in Virachey National Park in north-eastern Cambodia (McCann & Pawlowski, 2017). The species has also been confirmed at Chhlong, Kratie Province (Gray* et al., *2012); Seima Protected Forest, Mondulkiri Province (WCS Cambodia unpublished, 2015) and along the Prek Kasap in Stung Treng and Ratanakiri Provinces (Hon* et al., *2010). A rapid camera-trap survey conducted in Prek Toal Core Area, Tonle Sap Lake, Cambodia found presence of only Smooth-coated Otter* Lutrogale perspicillata *and the Endangered Hairy-nosed Otter* Lutra sumatrana*.*

Fishermen around Tonle Sap area often keep otter cubs as pets, however according to interviews in the early 2000s they often die after a short time in captivity (Poole, 2003), or they are eventually killed and skinned (Wilcox et al., *2016). In north-eastern Cambodia interviews with local people state that local wildlife traders buy otters skins for prices varying between USD 2 for a small skin and over USD 35 for a large skin, and otters have been used in traditional medicine for pregnant women (Hon* et al., *2010, Ashwell & Watson, 2008).*

China: Believed to be extirpated or extremely rare throughout much of their range in southern China *(according to Wright* et al., *(2015) its range does not reach other parts of China)*, this is due to large scale commercial hunting of all otter species prior to the 1980s. *Historical records of otter presence in China indicate that* Aonyx cinereus *was previously limited to the tropical and subtropical regions of the country, with records of their presence in Brahmaputra, the Irrawaddy, the Mekong, Red River and the Pearl River Basins as well as Hainan Island (Li & Chan, 2017). Based on pelt trade statistics it seems that before the 1980s, otters were abundant in many provinces, particularly the Yangtze and Pearl River Basins (more than 10,000 were killed annually, in 1957 alone more than 40,000 pelts were officially traded in China (species not specified)). By 1980–1985 the annual harvest had decreased dramatically and* recent surveys (field surveys, camera trapping, interviews with locals and otter experts) found A. cinereus at only two sites in China, these authors suggest that all three-otter species (*Aonyx cinereus, Lutra lutra* and *Lutrogale perspicillata*) found in China are on the verge of extinction (Li & Chan, 2017).

India: In India, *Aonyx cinereus* distribution range has decreased, for example in the Sundarbans. *Evidence to suggest that otters have disappeared from the western Himalayas is lacking, there have been studies to suggest that otter species, including* Aonyx cinereus *are still present in Arunachal Pradesh in the Himalayas region (Hussain* et al., *2018).*

A study (Prakash et al., *2012) of* Aonyx cinereus *presence in human modified environments in the Western Ghats attempted to identify key habitat features that influenced otter occupancy in the Anamalai Tiger Reserve (958 km^2) and adjoining tea and coffee plantations (220 km^2). Otter occupancy was recorded in all three habitat types indicating widespread use of riparian vegetation in human modified habitat. However, intensity of habitat use was much lower in tea plantations and coffee plantations than in protected areas, and potential refuge was considered a factor influencing habitat use (Prakash* et al., *2012).*
It is also reported to occur in Odisha, Eastern Ghats and direct observations of Aonyx cinereus *during 2005–2012 were recorded in six different locations including protected and non-protected areas (Mohapatra* et al., *2014).*

Otters are hunted in India for their pelts. According to researchers, Chinese and Western tourists in Tibet Autonomous Region (TAR) buy skins for home decor and possibly good luck but Tibetans are a large consumer base for pelts from otters poached in India and Nepal. At least 50% of the otter skins in China originate in India, with Lhasa, TAR, identified as a hub for trade in pelts *(species not specific)*.

Indonesia: *In West Java, Indonesia,* Aonyx cinereus *have been recorded in anthropogenic habitats such as rice paddy fields as well as fish and crab pond aquaculture areas. Wetland conversion as well as pollution were seen as the main threats towards the species. They are often perceived as pests and seen as competitors in fish pond and crab pond aquaculture areas leading to human–otter conflicts and possible retaliatory killings (Melisch* et al., *1996).* Aonyx cinereus *have been recorded to utilise rice paddy fields along the Batang Anai River, West Sumatra and these were found to serve as important potential habitats for long-term survival of otter species in human altered environments (Aadrean & Usio, 2017).*

Massive destruction of wetland forests in Indonesia has reduced the species habitat, as has habitat conversion to oil palm plantations in Sabah. *In the early 19th and 20th centuries* Aonyx cinereus *was relatively common in agricultural and urban or semi-urban environments on the island of Java and was considered to be a pest by local residents. It is still found to persist in heavily polluted waterways of Jakarta (Meijaard, 2014) and known to utilise rice paddies in Padang Pariaman, West Sumatra where pesticides in agriculture are heavily used (Aadrean* et al., *2011). Between 2000 and 2012 over 6.02 million ha of primary forest was lost in Indonesia and by 2012 proportional loss of primary forest in wetland areas increased (Margono* et al., *2014). Rudyanto and Melisch (1994) confirm very low-scale incidental hunting and trade from Sumatra from the early 1990s.*

Lao PDR: A healthy population was reported in Nakai-Nam Thuen National Protected Area in Lao PDR in 2016. *Camera trap surveys were conducted along one of the main rivers in Nakai-Nam Thuen National Protected Area, the Nam River, where locals reported they regularly observe otters. Between November to December 2015, 11 camera traps were placed and four recorded the presence of otters, all of which were* Aonyx cinereus *(Coudrat, 2016).*

The Beung Kiat Ngong Ramsar Site in Champasak Province of Lao PDR represents suitable habitat for otter populations and otters were recorded in this area in the early 1990s (Duckworth & Timmins, 2015). Further surveys in this area between 2007–2008 (Duckworth, 2008) and a short survey conducted in 2013 and in 2014 (Duckworth & Timmins, 2015) found no evidence of otters in the area. A study of small carnivore status in Nam Et Phou Louey National Protected Area on the Lao PDR–Viet Nam border reported the presence of Aonyx cinereus *(Johnson,* et al., *2009). Otter presence was also recorded in the Xe Sap National Protected Area in 2012, however this was detected from otter signs and these were not identified at the species-level (Gray* et al., *2013).*

Otters are reportedly hunted by Vietnamese poachers crossing into Lao PDR at the Nakai-Nam Theun National Protected Area (IUCN SSC Otter Specialist Group, in litt., *2019).*

Malaysia: The species is present throughout Peninsular Malaysia, but it is not known whether they are in decline. The Lower Kinabatangan Wildlife Sanctuary in Sabah is a key conservation area for *Aonyx cinereus,* and Borneo's Yayasan Sabah Conservation Area also hosts viable populations of this species. *A recent study of otters conducted by Abdul-Patah* et al., *(2014), in various sites in Peninsular Malaysia between April 2010 and March 2011, identified 126 spraints (dung) as from otter species; 83% of otter samples were identified to be* Lutrogale perspicillata *and 17% were* A. cinereus. Aonyx cinereus *was found to utilise four different habitat types with the highest occurrence recorded in paddy fields and casuarina forest. The species has also been reported to occur in the mangrove forests of Sarawak, Borneo (Mohd-Azlan* et al., *2016).*

The National Red List of Mammals for Peninsular Malaysia (2017) assessed Aonyx cinereus *as near threatened with an estimated area of occupancy of over 3.3 million ha (Perhilitan, 2017).*

Myanmar: In Myanmar, *Aonyx cinereus* was historically considered to occur throughout the country but is now considered to be extremely rare *across Myanmar's lowlands* (Zaw *et al.,* 2008), *however, there is little available information on the current distribution and status of A. cinereus in Myanmar. Multiple surveys conducted at the Mong La wildlife market have found specimens of otters; in 2006 an otter tail was observed for sale (species not specific) (Shepherd & Nijman, 2006) and in 2014, two carcasses of* A. cinereus *were observed, which had been sourced by local hunters (Shepherd & Nijman, 2014).*

Otters were poached in Myanmar for their fur and body parts and were listed among the highest value wildlife commodities for sale in Putao in the late 1990s (Zaw et al., *2008).*

Nepal: *In the Nepal National Red List assessment (2011),* Aonyx cinereus *has been categorised as data deficient. The species is found in Makalu Barun National Park and the districts of Kailali and Kapilbastu and documented to be found up to an elevation of 1,300 m (Jnawali* et al., *2011).*

Otters are hunted in Nepal for their pelts. *As noted above, otter pelts from Nepal are said to be sold in Tibet Autonomous Region.* In Nepal, a total of 756 otter pelts were seized between 1989 and 2017, either in Kathmandu or near an international border *(species not specific).*

Philippines: *In the Philippines* Aonyx cinereus *is only known to occur in Palawan. In 2010 a study focused on the distribution and trade dynamics of* A. cinereus *confirmed that the species was widely distributed in Palawan with its presence confirmed in all 12 municipalities of the mainland (Gonzalez, 2010).*

Singapore: Aonyx cinereus *populations in Singapore are highly localised in comparison to the wider distribution of* Lutrogale perspicillata *(W. Duckworth,* in litt., *2019).* Aonyx cinereus *was present in Singapore up until the 1960s and by the 1970s they were considered a rare species and threatened by habitat loss due to reclamation of rivers and pollution; they are now localised to offshore islands of Pulau Tekong (Lim* et al., *2016) and Ubin (K. Krishnasamy,* in litt., *2019).*

Thailand: Aonyx cinereus *has been reported to occur in Pru Toa Daeng Peat Swamp Forest in the Narathiwat Province of southern Thailand, otter signs were also found in rice paddies and secondary forest outside of the protected area (Kanchanasaka & Duplaix, 2011). A camera trap survey of small carnivores which was conducted in Thailand between 1996–2013 surveyed 21 sites; 19 based in protected areas and resulted in no camera-trap records for* A. cinereus *and there were few substantiated records in the 2000s for this species (Chutipong* et al., *2014).*

Bangkhuntien fishermen reported that otters threaten their fish farms, so they are persecuted but not typically for trade purposes (Chutipong et al., *2014).*

Viet Nam: *The population of* Aonyx cinereus *in Viet Nam has been reported to have a relatively wide distribution and is reported to occur from the northern highland's limestone, Hoang Lien mountain range, northern and central Annamites, central Indochina limestone, Ke Go/Khe New lowlands, the eastern plains dry forest, lowland Dong Nai watershed and the Mekong delta (Hussain* et al., *2011).*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Poaching and illegal trade for use as pets, for the fur trade and for the trade in parts for traditional medicine is said to pose a significant and growing threat to all four tropical Asian otter species. Commercial exploitation of otters is taking place both domestically, and internationally in violation of CITES. Information on the overall scale of illegal trade in tropical Asian otter species is scarce, due in part at least to relatively little attention paid to the enforcement of this species.

Most of the trade reflected in the CITES Trade Database is in live, reportedly captive-bred specimens for non-commercial purposes (primarily zoological purposes).

Analysis of CITES Trade Data between 1980–2017 shows relatively small levels of trade since Aonyx cinereus *was included on Appendix II in 1977.*

- *Only one skin (importer-reported) was exported from Viet Nam to the Czech Republic in 2010 which was a confiscation ("I").*
- *A total of 537 (exporter-reported) live specimens were exported during this period, of which 477 were reported to be captive-bred, 18 captive-born and 15 wild-sourced live specimens.*
- *The last reported exports of live wild-sourced specimens were in 2015 when four live specimens were exported from Lao PDR to the Republic of Korea for zoological purposes.*

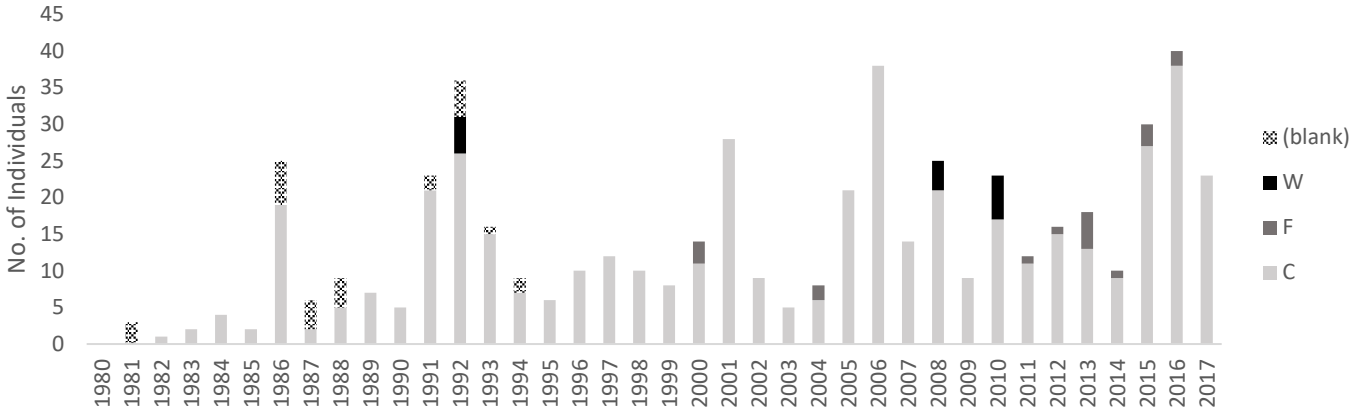

Figure 1: CITES Trade Data between 1980–2017 of live specimens of *Aonyx cinereus* (exporter-reported figures).

Live: Most seizures of live otters have occurred in Thailand, followed by Indonesia, Viet Nam and Malaysia and claims have been made that otters are being bred in captivity for the pet trade in Indonesia and Thailand. *Analysis of CITES Trade Data reported involving those countries showed under 30 reported in trade, the majority reported as captive-bred and the majority for the purposes of education, circus or travelling exhibition and zoo. However, all these countries are range States and therefore any seizures may not have been of specimens destined for international trade. Authorities in Malaysia have confirmed that captive breeding of otters occurs in Malaysia (K. Krishnasamy* in litt., *2019).*

In Indonesia and Thailand claims have been made that otters are being bred in captivity. *These reports that claim to be of captive-bred specimens need to be corroborated by both importing and exporting countries (K. Krishnasamy* in litt., *2019). Japan has been suggested as a key destination market for* Aonyx cinereus *in the international pet trade (Kitade & Naruse, 2018). A total of 99 live individuals were reportedly exported to Japan between 1977 and 2017 (CITES Trade Database). The recent emergence of otter cafes in Japan has the potential to motivate the otter pet trade (McMillan, 2018).*

Otters have been observed for sale online in a number of countries including China, Indonesia, Japan, Malaysia and in Thailand, where the possession of otters is prohibited, and all native otter species are protected (Siriwat & Nijman, 2018). Newborn pups and juvenile otters are in most demand within the pet trade and often offered for sale as an entire litter rather than individuals (Phassaraudomsak & Krishnasamy, 2018; Siriwat & Nijman, 2018). They appear to have high mortality rates when they are extracted from their natural habitats (Gonzalez, 2010) but are preferred when they are younger as they are easier to tame (Feeroz et al., *2011) and more suitable for pets.*

Similarly, there is also a demand for otters as pets in China, and it appears that keeping otters as pets is reported to occur in some south China provinces such as Yunnan (Y. Wang in litt., *2019). Interest from people wanting to buy otters in Sichuan, Shanghai, Beijing, Guangzhou and Yunnan can be found online (L. Hsun-Yen* in litt., *2019). There is evidence for demand in Europe and between 2010 and 2014 20 live specimens of* Aonyx cinereus *were observed for sale online (USD 600–1,600 per individual) (Fischer* et al., *2015).*

Skins: Most of the demand for skins appears to be in China, where populations have declined and are now likely only found in protected areas. Seizures of otter skins have declined since 2005; however, the reason for this decline is not clear. *In China it appears that otters are in demand for both their fur, body parts for Traditional Chinese Medicine and for the pet trade. Evidence can be found online of advertisements for otter fur on various platforms, although information on the species, source or authenticity of these furs is often lacking (L. Hsun-Yen* in litt., *2019 & Y. Wang* in litt., *2019). Some posts, in particular for hats, claim that furs are imported from Canada or USA (Y. Wang* in litt., *2019). There has only been one reported export of an* Aonyx cinereus *skin between Viet Nam and the Czech Republic, although this was reported to be a confiscation ("I") (CITES Trade Database).*

Medicine: Otters are also used for a range of medicinal purposes. *There appears to be little evidence that there is much demand for otters for medicinal purposes in China and that this utilisation was more historical, and that quantitative data are lacking to determine if this trade poses a significant threat (L. Hsun-Yen* in litt., *2019). A search conducted on Chinese markets online for medicinal products derived from otters found products from otter liver offered for sale, however these advertisements were not at the species-level (Y. Wang,* in litt., *2019),*

therefore making it difficult to determine the level of trade in these commodities. Otters were poached in India for skins and oil which is extracted from the animal's fat and used for medicine destined for markets in northern India and also trade routes through Nepal and Bangladesh (Meena, 2002). In Cambodia, otters were reportedly used as medicine for pregnant women (Hon et al., *2010).*

More detailed information on seizures and online/physical markets of otter commodities across both range and non-range States can be found below (Table 1):

Table 1: Information on seizures and online/physical markets across both range and non-range States (source: SS unless referenced).

Country	Seizures	Online & Physical Markets
Cambodia (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015, Aadrean* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980-2015 a total of 46 individuals were seized (nine live and 37 dead, not species specific) (Gomez* et al., *2016).*	• *Market surveys between 2016–2017 observed one* Aonyx cinereus *skin (Gomez & Bouhuys, 2018).*
China (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980–2015 a total of 16 seizures reported 2,403 individuals (not species specific), with no reported seizures of live individuals (Gomez* et al., *2016).*	• *Searches conducted online found otter products for sale with prices varying from USD 60–120 for a complete otter pelt and USD 2,600 for a live specimen (not species specific) (Y. Wang* in litt., *2019).* • *Provincial pelt trade statistics for 1953–1985 reports >10,000 individuals killed annually in Yangtze and Pearl River Basins (not species specific). In 1957 alone >40,000 otter pelts were officially traded (not species specific) (Li & Chan, 2017).* • *Searches conducted on Chinese online platforms found otter commodities for sale varying from USD 60–120 for a whole pelt and USD 2,600 (maximum price) for one specimen (Y. Wang* in litt., *2019).*
India (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• Between 1980–2015 a total of 2,949 otter pelts were seized (not species specific) and many had been marked with Tibetan script suggesting international trade. • *Between 1994 and August 2006 978 reported seizures (involving tiger, leopard and otter) amounting to 777 otter individuals (not species specific) (Banks* et al., *2006).*	

Indonesia (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015, Aadrean* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980–2015 one live individual was seized (not species specific) (Gomez* et al., *2016).* • *A total of three seizures between 2015–2017 amounting to seven live individuals (four Lutrogale perspicillata and three unidentified) (Gomez & Bouhuys, 2018).*	• *449 online adverts with an estimated minimum of 503 individuals and maximum of 917* Aonyx cinereus *offered for sale between January–May 2017 (Gomez & Bouhuys, 2018).* • *Market surveys between 2016–2017 observed two live* A. cinereus *juveniles for sale (USD 90–115), one of the traders claimed the otter was captive-bred and the other trader claimed it had been wild-caught (Gomez & Bouhuys, 2018).* • *Five-month study online between January and May 2012 found a total 63 posts from 46 sellers, 84% of posts were identified as* A. cinereus *(Aadrean, 2013).*
Japan (non-range State) Otter species present: *Enhydra lutris* *(Doroff & Burdin, 2015)*	• *Between 1980–2015 one seizure was reported in Japan involving one individual (not reported if live or dead and not species specific) (Gomez* et al., *2016).* • *A total of three seizures between 2015–2017 amounting to 32 live otters (not species specific) (Gomez & Bouhuys, 2018).*	• *A study of online markets identified 12 sellers were selling/sold a minimum of 85 individuals since 2011 of which most (87%) were* Aonyx cinereus*, prices range from USD 7,200–14,580 (11 adverts) (Kitade & Naruse, 2018).* • *A total of 20% were reportedly imported and 46% captive-bred and only vague information was provided on domestic captive breeding sources, a domestic zoo was reported to be the origin of three individuals (Kitade & Naruse, 2018).* • *This study also suggested the possibility of specimens imported from Malaysia to Japan for the purpose "zoo" were subsequently being sold into the pet market (19 were imported with source code "Z" between 1977–2018 (CITES Trade Database, 2019).* • *A total of 10 otter cafes were identified across Japan, all established after 2015, with a total of 32 otters, all claimed to be A. cinereus (Kitade & Naruse, 2018).*
Lao PDR (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• Between 1980–2015 all specimens seized were dead, mostly skins, *involving 23 individuals (not species specific) (Gomez et al., 2016),* suggesting that poaching is largely motivated by the demand for pelts.	• *Market surveys between 2016–2017 observed two* Aonyx cinereus *skins (Gomez & Bouhuys, 2018).* • *One dead* A. cinereus *was observed drying on clothes line in the small town of Boten in Lao PDR, which is about 1 km away from the Chinese border; the purpose of trade for this specimen was unknown (Krishnasamy* et al., *2018).*
Malaysia (range State) Otter species present: *Aonyx cinereus* *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980–2015 a total of 14 live individuals were seized (not species specific) (Gomez et al., 2016).* • *A total of two seizures between 20152017 amounting to two live* Aonyx cinereus *(Gomez & Bouhuys, 2018).*	• *10 online adverts with an estimated minimum of 14 and maximum 16 live* Aonyx cinereus *offered for sale (Gomez & Bouhuys, 2018).*

Myanmar (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980–2015 a total of two individuals seized (not species specific) and no seizures of live individuals (Gomez* et al., *2016).*	
Nepal (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• Between 1989–2017 a total of 756 otter pelts were seized (not species specific). *Between 1980–2015 a total of nine seizures involving 383 individuals (not species specific), of which there were no reports of live individuals seized (Gomez* et al., *2016).*	
Philippines (range State) Otter species present: *Aonyx cinereus* (*Wright* et al., *2015)*	• *Between 1980–2016 a total of six live individuals were seized (one live and five dead, not species specific) (Gomez* et al., *2016).* • *A total of eight seizures between 2002–2014 involving 20* Aonyx cinereus *(not reported if specimens were live or dead) (Gomez & Bouhuys, 2018).*	• *A study of* Aonyx cinereus *in Philippines also found that otters were being extracted from the wild in Palawan and transported to "stocking centres" and subsequently transported to cities such as Manila for the pet trade or for the purpose of zoos (Gonzalez, 2010).*
South Korea (non-range State) Otter species present: *Lutra lutra* *(Roos* et al., *2015)*	• *Between 1980–2015 there was one reported seizure involving one individual (not reported if live or dead and not species specific) (Gomez* et al., *2016).*	
Tibet Autonomous Region (non-range State) Otter species present: *Lutra lutra (plateau region)* *(Roos* et al., *2015)*		• *In 2005 and 2006 over 1,800 otter skins were observed at Linxia market, including* Lutrogale perspicillata *and* Lutra lutra *documented (Banks* et al., *2006).*

Thailand (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015, Aadrean* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980–2015 a total of 14 individuals were seized (12 live and two dead, not species specific) (Gomez* et al., *2016).* • *A total of five seizures between 2015–2017 involving 35 live individuals (24* Aonyx cinereus *and 11 unidentified), of these seizures four indicated international trafficking and at least three were destined for Japan (Gomez & Bouhuys, 2018).* • *In 2018 five* A. cinereus *were seized in Thailand (TRAFFIC's wildlife trade information system).* • *In 2013, six live* Aonyx cinereus *and five* Lutrogale perspicillata *were seized at Suvarnabhumi International Airport (Kitade & Naruse, 2018).*	• *80 online adverts with an estimated minimum of 182 and maximum of 221 live* Aonyx cinereus *offered for sale (Gomez & Bouhuys, 2018).* • *160 online adverts totalling 337 individuals (between March 2017–April 2018) of which 57 posts could be identified as* A. cinereus *(127 individuals), this suggested international trade occurring between Thailand and Malaysia, however this was not verified (Siriwat & Nijman, 2018).* • *Over a 23-day period in 2016 at least 13* A. cinereus *were offered for sale on Facebook (Phassaraudomsak & Krishnasamy, 2018).*
Viet Nam (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015, Aadrean* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980–2015 a total of 13 live individuals were seized (not species specific) (Gomez* et al., *2016).* • *A total of three seizures between 2015–2017 amounting to 15 live otters (six* Aonyx cinereus *and nine unidentified) and at least one incident indicated international trafficking as the otters were claimed to have been sourced from Thailand (Gomez & Bouhuys, 2018).* • *In 2018 10 live* A. cinereus *were seized in Nam Dinh (TRAFFIC's wildlife trade information system).*	• *A total of 21 online adverts with an estimated 12 live* Aonyx cinereus *offered for sale and 15 unidentified between January and May 2017 (Gomez & Bouhuys, 2018).*

Additional Information

Threats

In northern South-east Asia, international trade has probably been a substantial part of the problem causing declines, but cannot be considered the only, nor even the main driver of illegal offtake (W. Duckworth, in litt., *2019).* In addition to international and domestic trade, throughout its range *Aonyx cinereus* faces threats including human development and natural wetland conversion for agriculture including aquaculture, plantations (e.g. coffee, oil palm, tree plantations and rice fields), wetland reclamation; siltation due to deforestation; pollution by pesticides; mining; quarrying; slash-and-burn agriculture. South-east Asia has the highest rate of forest loss in the tropics. *The global extent of wetlands is now estimated to have declined between 64–71% since 1900 and habitat loss and degradation continues worldwide (Ramsar, 2015). Evidence suggests that although available otter habitat is decreasing and wetlands continued to be converted into agriculture throughout much of* A. cinereus *range, otters are adapted to survive in moderately disturbed habitats where there is some level of human presence (Mohapatra* et al., *2014,).*

A further threat to otters is a reduced prey base from overfishing and the species preference for crabs and other invertebrates makes water contamination from inappropriate or illegal pesticide use in wetlands, as well as organochlorides, heavy metals and other pollutants an important concern. The common practice of dumping garbage in wetlands is also a threat to otters, whose scat has been observed to contain plastics. Fishermen are known to kill otters as a competitor for fish. Destructive fish-kill practices are common in the Western Ghats where dynamite fishing, bleach fishing and electric-rod based fishing and the use of pesticides in banana plantations and rice fields.

Recent modelling research suggests that climate change will significantly impact the *Aonyx cinereus* habitat, forecasting up to 40% loss of suitable areas by 2070. This scenario is made worse for *A. cinereus* by the marginality of its climatic niche.

Conservation, management and legislation

Aonyx cinereus is generally protected in its range States (although some needs reviewing *(Duplaix & Savage, 2018)) e*xcept in Cambodia, which removed protections for *A. cinereus* and *Lutrogale perspicillata* in 2007; Indonesia where *A. cinereus* is not protected by domestic legislation and Nepal and Brunei Darussalam. No management plans or species-specific conservation measures are in place for *A. cinereus* in range States. *A. cinereus* was listed in Appendix II of CITES (1977) and is not protected by any other international agreement.

Artificial Propagation/captive breeding
In captivity, successful breeding occurs at about two years old, but sometimes younger (Wright *et al.,* 2015); they breed year-round and gestation is believed to be 60–86 days. Litter size (in captivity) is 1–7 (up to 16; with a mean of 3 to 4) (Hussain, *et al.,* 2011) and longevity (in captivity) is around 11 years (Wright *et al.,* 2015).

There have been reports of otter farms in China (Y. Wang, in litt., *2019) and Pakistan (ISOF, 2014). In both Indonesia and Thailand there have been unconfirmed/unverified reports of otters being bred for the purpose of trade (Gomez & Bouhuys, 2018). It is possible that juveniles and newborn pups which are extracted from the wild are then sold as captive-bred or used as breeding stock.*

Small-clawed Otters breed well in captivity and there are over 226 zoos that maintain captive Aonyx cinereus *individuals and the studbook lists over 977 individuals living from 48 founders (Duplaix & Savage, 2018). The species occurs widely in zoo and animal exhibitions throughout South-east Asia (R. Melisch,* in litt., *2019).*

Implementation challenges (including similar species)
Unprocessed fur of otters allows for distinction of specimens at the species-level (Kuhn, 2009). Although, mammalian hair microscopy for species identification is widely used for ecological, forensic and/or economic purposes, few such guidelines are publicly available and cover only some otter species. There is very little published material that would allow the simple distinguishing of processed pelts, therefore, it would be challenging for untrained officials to distinguish between processed pelts (Gomez et al., *2016).*

In some consumer countries such as Japan (where trade in otters is evident), there are apparent loopholes that allow Appendix II (and III) species to be traded once smuggled into the country and therefore CITES-listing does not fully protect the animal from illegal trade (Duplaix & Savage, 2018).

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Transfer of Smooth-coated Otter *Lutrogale perspicillata* from Appendix II to Appendix I

Proponent: Bangladesh, India and Nepal

Summary: The Smooth-coated Otter *Lutrogale perspicillata* is an otter of lowlands and floodplains. It forages in a wide variety of habitats including large rivers and lakes, peat swamp forests, mangroves and estuaries, as well as rice-fields. It has a broad distribution range, from Java, Sumatra and Borneo, northward to south-western China, west through Nepal, Bhutan and India to Pakistan, with an outlying, and taxonomically distinct population in Iraq. There are some indications that *L. perspicillata* can adapt to live in human-modified environments.

Lutrogale perspicillata was assessed in the IUCN Red List as Vulnerable in 2014 on the basis that the population was inferred to have declined by more than 30% in the last 30 years (three generations) due to habitat loss and exploitation. However, as *L. perspicillata* was assessed as Vulnerable and not Endangered, declines of greater than 50% were not indicated. There is evidence that there have been declines in a number of national populations: in China, Viet Nam and parts of Bangladesh it appears to have been extirpated, and declines are noted elsewhere (e.g. Pakistan). While some national populations appear healthy (Singapore, Iraq) there is uncertainty for other countries (e.g. India, Indonesia). Its status in National Red Lists varies from least concern in Malaysia (2017) to vulnerable in Thailand (2005), to endangered in Nepal (2011) and critically endangered in Bangladesh (2014).

The subfamily Lutrinae was listed in CITES Appendix II in 1977. Historically exploited for the fur trade and for use in Traditional Asian Medicine this exploitation was considered one of the main causes for past population declines. The pet trade and otter-petting cafes have been identified as emerging uses for otters, with Japan and Thailand identified as destinations. *Lutrogale perspicillata* does not appear to be one of the favored species for this trade, perhaps due to its larger size, although a limited number of online advertisements were found in Thailand and elsewhere.

According to the CITES Trade Database, international legal trade has been limited to small quantities in recent years: there have been no reported direct exports of skins since 1983 and only 41 live individuals were reported to have been exported between 1977 and 2016 (most live individuals were reported to be captive-bred). Some countries from where otters have been observed as offered for sale are also range States for the species, which suggests a degree of domestic trade. The total current level of demand for this species for pelts, pets and medicine is unclear as much of the trade is apparently illegal and unreported.

The species is protected in all range States except for Cambodia and Brunei Darussalam and the status in Bangladesh is unclear. Protection may vary in form and enforcement, for example: in Thailand the possession of otters is prohibited and all native otters are protected, but online adverts of *L. perspicillata* and other otter species can still be found.

Lutrogale perspicillata was historically in high demand for its pelts and a trade in pelts continues illegally. A number of otter seizures have been reported in range States, with some apparently destined for export; enforcement staff are reported to have difficulty in identifying pelts and products in trade to the species level, therefore seizures are often not reported to the species level. Whilst consumers and tourists in the Tibet Autonomous Region (TAR), China, are thought to provide the largest consumer base for otter pelts (particularly sourced from India and Nepal) the impact on *L. perspicillata* is unclear as very little species-specific information is available.

There are reportedly otter farms in China, Pakistan and Indonesia, and although species can be bred in captivity, it is not clear how much of the trade is being met from these sources.

In addition to harvest, *Lutrogale perspicillata* is believed to be affected by habitat loss and

degradation, as well as pollution, decline in prey biomass, persecution and climate change.

Analysis: Information on the status of *L. perspicillata* in large parts of its range (e.g. India, Lao PDR, Thailand, Myanmar etc.) is scarce although the population is unlikely to be considered small globally. The species does not have a restricted range, occurring from Iraq in the west to Indonesia in the east. There are no quantitative baseline data on which to base population trends, although there is some information that the species has decreased or been extirpated in some parts of its range. The IUCN Red List assessment notes a decline of more than 30% over three generations. While habitat loss is a serious threat, the species is known to occur in human-modified environments. There has been limited reported legal trade in *L. perspicillata* since 1977. The current level of demand for this species is unclear, as is the amount of international or domestic trade met by captive-bred sources. The size of the illegal and/or domestic trade is also largely unknown.

On the basis of a greater than 30% (but less than 50%) population decline over three generations inferred from a decline in habitat and exploitation in the Red List assessment, it seems uncertain that this species meets the guidelines for a marked recent population decline as described in *Res. Conf. 9.24 (Rev. CoP17)* for inclusion in Appendix I at the present time. There are significant levels of uncertainty regarding status in some parts of the species' range and levels of trade, and if further information were to become available on this it may help determine of the species lies closer to the 50% decline guideline for inclusion in Appendix I.

Other Considerations: Levels of legal international trade appear low, and therefore it is assumed that most harvest is for domestic and/or illegal trade. Any additional benefits of an Appendix I listing are not clear unless enforcement efforts are increased.

Res. Conf. 12.10 (*Rev. CoP15*) outlines that inclusion in Appendix I would mean commercial captive breeding operations would need to meet the provisions of *Res. Conf. 10.16* (Rev.) to be registered with the CITES Secretariat, and that registered operations shall ensure an appropriate and secure marking system to identify all breeding stock and specimens in trade. This enhanced oversight could help allay concerns over fraudulent claims of captive breeding and wild offtake for breeding stock.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Range

Bangladesh, Bhutan, Brunei Darussalam, Cambodia, China, India, Indonesia, Iraq, Lao People's Democratic, Republic, Malaysia, Myanmar, Nepal, Pakistan, Singapore, Thailand, Viet Nam (de Silva et al.*, 2015).*

IUCN Global Category

Vulnerable *A2cde* *ver 3.1* *(assessed 2014) (de Silva* et al.*, 2015).*

Biological criteria for inclusion in Appendix I

A) Small wild population

No current population estimates for L. perspicillata *are available. One study noted that otters live at low densities and are shy and often nocturnal or crepuscular, and hence are difficult to track and to make direct estimates of population size (Khan* et al.*, 2014). With such a widespread range the population is unlikely to be small.*

B) Restricted area of distribution

Lutrogale perspicillata is found in Java, Sumatra and Borneo, northward to south-western China, west through Nepal and Bhutan and India to Pakistan, excluding the Indus Valley. There is an isolated population in the marshes of Iraq (*L. p. maxwelli*) indicating the range of the species must have once been wider. *Lutrogale perspicillata* remains distributed throughout much of south Asia and southern South-east Asia, though now in northern South-east Asia it is restricted and occurs mainly in protected areas, Lutrogale perspicillata *populations in South-east Asia are present in human-modified environments, most notably Thailand and Singapore (Theng & Sivasothi, 2016 & Kamjing* et al.*, 2017). Habitat loss and landscape modification for agriculture and urbanisation is threatening* L. perspicillata *habitat, although in Asia it is largely unknown to the extent this impact is having on*

wild populations (Kamjing et al.*, 2017). Otter presence in human-modified environments in some parts of its range suggests that wider use of anthropogenic habitat is prevented by offtake (W. Duckworth, in litt., 2019).*

C) Decline in number of wild individuals

The population of *L. perspicillata* is inferred to have declined by >30% in the last 30 years. *Although there is evidence that some* L. perspicillata *populations have experienced decline (see country profiles below), the lack of baseline data on the distribution and ecology of* L. perspicillata *makes it difficult to determine the overall current rate of decline. There is no generally accepted/agreed scientific method for most population counts and/or estimates in the wild for most species (although there are exceptions e.g.* Lutra lutra *see Reuther* et al., *2000), therefore population changes of most Asian otter species are inferred from habitat trends or presence/absence surveys. Multiple studies have reported minimal or no sightings of this species or its signs in locations it has been historically recorded (Hussain & Choudhary, 1997, Li & Chan, 2017, Acharya & Rajbhandari, 2012).*

Bangladesh: *Lutrogale perspicillata* were formerly widely distributed and found in all major wetlands in Bangladesh, they are now locally extinct from much natural habitat in the country due to rapid habitat degradation and food scarcity. *A study by Aziz (2018) investigated the distribution and status of the Asian Small-clawed Otter* A. cinereus *in the Sundarbans mangrove forest recorded a total of 53 individuals of* A. cinereus *but no wild encounter was recorded for* L. perspicillata*.* Fishermen in the Sundarbans still maintain a semi-captive population of *L. perspicillata.*

Lutrogale perspicillata are said to be hunted in Bangladesh for their pelts.

The National Red List Assessment (2014) for Bangladesh has classified Lutrogale perspicillata *as critically endangered and its area of occurrence is estimated to be 48,147 km^2 (Feeroz, 2015). The species is restricted to hilly areas in the north-east and south-east and that the largest population possibly still thrives in the Sundarbans mangrove forest, however the population is said to have declined by 90% due to hunting, poaching, loss of habitat and human-wildlife conflict as a result of commercial fish farming (Feeroz, 2015). There is a small semi-captive population in Norail District used for traditional fishing which has also declined by 80% in the last two decades.*

Bhutan: *A survey of otter distribution in Central Bhutan also stated that it is likely that* Lutrogale perspicillata*,* Lutra lutra *and* Aonyx cinereus *all inhabit the rivers of Bhutan and only* Lutrogale perspicillata *was directly sighted during the study. A total of five of the six 5.5 km transects were positive for otter signs and one site had the highest rate (95%) of otter plots that were positive for otter signs. Over 40% of land in Bhutan has been designated as a natural protected area and hunting is considered a serious offence and is prohibited (Chettri & Savage, 2014). This study represented the first survey of otters in Bhutan and further research is required to assess their status in this country.*

Cambodia: *Smooth-coated Otter is one of the more widespread otter species in Cambodia with reliable field records from a number of sites, including Tonle Sap Biosphere Reserve, where a relatively large number of unconfirmed and many confirmed Smooth-coated Otters have been reported. The site is likely to be regionally significant for Smooth-coated Otter; the species is probably extinct in Viet Nam and is in decline in other ranges countries as well as Cambodia (Willcox* et al.*, 2016).*

In Cambodia, big pelts in particular are said to be sold to middlemen who take the skins to Viet Nam. *Local trade in otters has reportedly decreased in both quantity and price however, otters are still reportedly hunted in the Tonle Sap Lake area (IUCN SSC Otter Specialist Group in litt 2019).*

China: *Historic records of* L. perspicillata *presence in China report its occurrence in the Pearl River Delta of Guangdong Province and the Red River and the Irrawaddy in Yunnan Province. Based on pelt trade statistics it seems that before the 1980s, otters were abundant in many provinces, particularly the Yangtze and Pearl River Basins (more than 10,000 were killed annually, in 1957 alone more than 40,000 pelts were officially traded in China (species not specified)). By 1980–1985 the annual harvest had decreased dramatically, and no evidence of presence of* L. perspicillata *in China based on surveys, expert consultation and literature review was found between 2006-2016, indicating that the species had been extirpated (Li & Chan, 2017).*

India: In the Indian subcontinent the species is adapted to live in the semi-arid region of northwestern India and the Deccan Plateau, as well as more mesic areas such as the southern plains. In the Punjab plains of India, it occurs along some stretches of the Beas, Sutlej and Ravi Rivers and the Harike wetlands. *L. perspicillata occurrence has been recorded in a number of protected areas, including but not limited to National Chambal Wildlife Sanctuary (Hussain, 1993), Corbett Tiger Reserve (Nawab, 2007), Dudhwa Tiger Reserve and Katerniaghat Wildlife Sanctuary (Hussain, 2002), Hastinapur Wildlife Sanctuary (Khan* et al.*, 2014), Periyar Tiger*

Reserve (Anoop & Hussain, 2004), Similipal Tiger Reserve (Mishra, et al.*, 2018), Sundarbans Tiger Reserve (Manjrekar & Prabu, 2014) and the Tungabhadra Otter Reserve Sanctuary created in 2016 (Duplaix & Savage, 2018).A study in the National Chambal Sanctuary found that in 1988, 36% of sites were found with* L. perspicillata *signs compared to 24% in 1992 (in total 21 fewer sites showed signs of otters in 1992) (Hussain & Choudhury, 1997).*

Lutrogale perspicillata *occupancy in Hastinapur Wildlife Sanctuary was much lower (7%) compared to 29% at the Alaknanda-Ganga Basin site, which may be because Hastinapur Wildlife Sanctuary is one of the most human populated and disturbed protected areas in Uttar Pradesh and very accessible to humans. In the Alaknanda-Ganga Basin otters were recorded in less disturbed habitats which were inaccessible to humans. Populations at both sites were fragmented with areas of available habitat restricted, although showed some adaptability (Khan* et al.*, 2014).*

Otters are hunted in India for their pelts. According to researchers, Chinese and Western tourists in Tibet Autonomous Region (TAR) buy skins for home decor and possibly good luck but Tibetans are a large consumer base for pelts from otters poached in India and Nepal. At least 50% of the otter skins in China originate in India (Ghosh 2005; Duckworth 2013) with Lhasa, TAR, identified as a hub for trade in pelts *(species not specific)*.

Indonesia: Lutrogale perspicillata *has been historically reported to occur in Sumatra (Aadrean,* et al.*, 2011), however there is little available information on the current distribution and status of* L. perspicillata *in Indonesia.*

Between 2000 and 2012 over 6.02 Mha of primary forest was lost in Indonesia and by 2012 and proportional loss of primary forest in wetland areas increased (Margono et al.*, 2014). Rudyanto and Melisch (1994) confirm very low-scale incidental hunting and trade from Sumatra from the early 1990s.*

Iraq: There is an isolated population in the marshes of Iraq (*L. p. maxwelli*), indicating that the range of *L. perspicillata* must have once been wider. The Iraq subspecies has recently (surveys during 2005 to 2012) been found to be thriving in the southern marshes of Iraq. *A study confirmed the persistence of* L. p. maxwelli *in Iraq and in Arbil Province in Kurdistan, extending the known range of this species within the Middle East (Omer* et al.*, 2012).* In Iraq, *L. perspicillata* are hunted for their pelts and sold to smugglers who operate along Iraq's borders, fetching between USD100–300 per pelt.

A survey between 2005-2012 conducted intensive in-situ field research to determine the status of L. p. maxwelli *and Eurasian Otter* Lutra lutra *in Iraq and surveyed 21 different sites in nine Iraqi Provinces (Al-Sheikhly & Nader, 2013).* Lutrogale perspicillata *maxwelli was thriving in the marshes of southern Iraq where its presence had previously been reported and DNA analysis confirmed the presence of* L. p. maxwelli *in TaqTaq, Arbil province in 2007, however apart from this isolated population all other northern sites were found to contain the Eurasian Otter* Lutra lutra *(Al-Sheikhly & Nader, 2013).*

Lao PDR: *The Beung Kiat Ngong Ramsar Site in the Champasak province of Lao PDR represents suitable habitat for otter populations and otters were recorded in this area in the early 1990s (Duckworth & Timmins, 2015). Further surveys in this area between 2007-2008 (Duckworth, 2008) and a short survey conducted in 2013 and in 2014 (Duckworth & Timmins, 2015) found no evidence of otters in the area. Otter presence was also recorded in the Xe Sap National Protected Area in 2012, however this was detected from otter signs and these were not identified at the species-level (Gray* et al.*, 2013).*

Malaysia: L. perspicillata *is present throughout the Malay Peninsula. In 1994 the species was considered common and widely distributed (Sivasothi & Nor, 1994). A recent study of otters conducted by Abdul-Patah* et al.*, (2014) in various sites in Peninsular Malaysia between April 2010 and March 2011 identified 126 spraints as from otter species; 83% of otter samples were identified to be* L. perspicillata *and 17% were A. cinereus. A total of eight different habitats were utilised by* L. perspicillata *and its highest occurrence was recorded in paddy fields. The National Red List Assessment of Peninsular Malaysia (2017) has categorized* Lutrogale perspicillata *as least concern with an estimated area of occupancy of 3.2 million ha (Perhilitan, 2017).*

Myanmar: *In Myanmar* L. perspicillata *has been recorded to occur at 1000 m above sea level (U Tun Yin, 1967).* Previously the species occurred throughout Myanmar, and was considered common in the Chindwin, it is now considered to be rare in the lowlands *(Zaw* et al.*, 2008), however there is little available information on the current distribution and status of* L. perspicillata *in Myanmar. Multiple surveys conducted at the Mong La wildlife market have found specimens of otters, in 2006 an otter tail was observed for sale (species not specific) (Shepherd & Nijman, 2006).*

Otters have been poached in Myanmar for their fur and body parts and were listed among the highest value wildlife commodities for sale in Putao in the late 1990's (Zaw et al.*, 2008).*

Nepal: In Nepal, *L. perspicillata* is found along the braided channels of Narayani River, with its slow current and shallow depth. *A study in Narayani River conducted during the months of June 2011 and January 2012 did not record any occurrence of* Lutrogale perspicillata*, although some of the areas surveyed had recorded signs in 2009 and had been identified as key priority sites for otter conservation (Acharya & Rajbhandari, 2012). In the National Red List for Nepal,* Lutrogale perspicillata *has been categorised as endangered and its population is estimated to be <1,000, direct observations are said to be rare and only from Bardia National Park, Chitwan National Park and Shukla Phanta Wildlife Reserve and absent from other previous localities (Jnawali* et al*. 2011).*

Otters are hunted in Nepal for their pelts. *As noted above, otter pelts from Nepal are said to be sold in Tibet Autonomous Region.* In Nepal, a total of 756 otter pelts were seized between 1989 and 2017, either in Kathmandu or near an international border *(species not specific)*).

Lutrogale perspicillata *is said to occur at two Ramsar sites in Nepal; Beeshazar and Associated Lakes and Ghodaghodi Lake Area Ramsar Site, however no population studies appear to have been conducted in these areas. Both these sites are under threat from construction, over grazing, illegal logging and eutrophication accelerated by agricultural activities in addition to invasive species (Kafle & Savillo, 2009).*

Pakistan: It has been confirmed in several sites in Pakistan in 2008/9 but thought to be declining at all sites where it was detected (and only at five of the 25 sites year-round) and currently existing in scattered populations in fragmented habitats. There is evidence it was once widely distributed through the Sindh region in Pakistan and now only exists in isolated and fragmented populations, otters are said to have disappeared from some lakes due to pollution. In Pakistan, fishermen are said to target *L. perspicillata* for their pelts as they fetch high prices from middlemen who move the pelts into Russia and China, and populations are said to be under extreme pressure from illegal hunting.

In Pakistan, L. perspicillata *presence has been recorded at both Haleji Lake and Keenjhar Lake (one of the largest lakes in Pakistan), both designated Ramsar sites situated near Karachi. At Haleji Lake the Sindh Wildlife Department maintains a captive breeding centre which includes* L. perspicillata *(Khan* et al.*, 2012). Biodiversity at both sites are said to be under threat from disturbance, habitat degradation, pollution and eutrophication.* Lutrogale p. sindica *was also recorded at Chotiari Reservoir in 2008, which was the first record of a direct sighting of otters in this area in over a decade (Rais,* et al.*, 2009).*

Singapore: *Theng & Sivasothi (2016) compiled 370 sightings of this species between 1998 and 2014, which showed generally an increase in the number of* L. perspicillata *sightings from 1998 (peaking in 2014). The population was found to be doing particularly well in areas including Sungei Buloh Wetland Reserve, Pulau Ubin and Serangoon Reservoir. They are often encountered along the northern coastline of Singapore Island and Pulau Ubin (Lim* et al.*, 2016) and found in urban areas such as the tourist hotspot Gardens by the Bay (a 101 hectares nature park adjacent to the Marina Reservoir) (Khoo & Sivasothi, 2018). Between August 2016 and January 2017 field surveys were conducted at 15 sites (selected based on site availability) in Singapore, and reported 11 distinct groups of* L. perspicillata*, with an estimated total of at least 79 individuals (Khoo & Sivasothi, 2018)*

Thailand: L. perspicillata *has been classified as vulnerable (2005) in the National Red List assessment (Nabhitabhata & Chan-ard, 2005).* Remaining natural patches seem to be critical refuges for otter, allowing them to persist in an otherwise heavily transformed landscape. Recent studies in Thailand suggest that *L. perspicillata* can persist in highly modified fragmented if patches of natural habitats (e.g. mangroves) are still available. *A camera trap study between 1996-2013 surveyed 21 sites (19 based in protected areas) recorded* L. perspicillata *presence in four surveys areas (19%) (Chutipong* et al.*, 2014).*

In the inner Gulf of Thailand, otters use cover in traditional aquaculture ponds, but not agriculture and urban cover. In Thailand, rapid economic development and expansion of Bangkok has caused extensive destruction and fragmentation of mangrove forests along the coast: between 1961 and 2009 the coverage of mangrove forest decreased by half.

Pet otters are popular in Thailand, most of the seizures of live otters reported occurred in Thailand (some at international airports (species not specific)). Pelts from Thailand are said to end up in China.

Viet Nam: L. perspicillata *presence in Viet Nam has been reported from Dak Lak and Kon Tum provinces (Duckworth & Canh, 1998). It has been suggested that* L. perspicillata *could possibly be extinct in Viet Nam (Willcox* et al.*, 2016).*

A survey conducted in the U Minh Ha National Park in the Mekong Delta identified a seized otter skin in 2007 as L. perspicillata *and it was noted that the skin had unlikely been sourced from outside of U Minh district (Willcox* et al.*, 2017).*

The population of *L. perspicillata* is inferred to have declined by more than 30% in the last 30 years. A reduction in the population size of *L. perspicillata* has been observed in many parts of its range due to intense poaching and the extent of loss of habitat in south and southeast Asia.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Poaching and illegal trade for use as pets, for the fur trade and for the trade in parts for traditional medicine is said to pose a significant and growing threat to all four tropical Asian otter species. Commercial exploitation of otters is taking place both domestically, and internationally in violation of CITES. Information on the overall scale of illegal trade in tropical Asian otter species is scarce, due in part at least to relatively little attention paid to the enforcement of this species.

The CITES Trade Database includes very little legal trade since 1983. Exports since 2000 have been limited to live specimens for scientific or zoological purposes. *Analysis of CITES Trade Data between 1977 and 2016 shows relatively small levels of trade since* L. perspicillata *was included on Appendix II in 1977. According to importers:*

- *No reported direct imports of skins for the purpose of commercial trade since 1983, Germany did report a re-export of 3,058 skins in 1993 from China back to China (pre-convention specimens).*
- *A total of 41 live animals were imported during this period, of these 13 were exported for the purpose of commercial trade and 17 for zoos.*
- *Japan imported 13 live specimens (from China and Thailand) and a further two live animals were imported into Malaysia (from Indonesia)*
- *Two live specimens were reported to have been wild-sourced ("W") and imported for the purpose of commercial trade (a further seven had no source code), all other exports were from captive-bred or captive-born sources exported from China.*

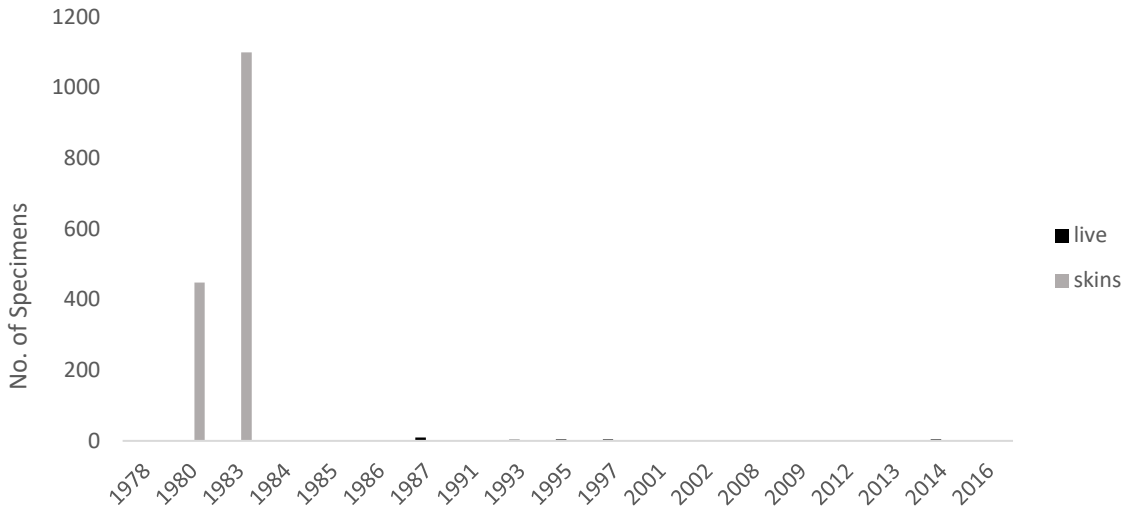

Figure 1: CITES Trade Data between 1977-2016 of live specimens and skins of *Lutrogale perspicillata* (importer-reported figures)

Lutrogale perspicillata *occurs in the range States which represent the key destination countries, and although international trade between range States is occurring, a large proportion of this trade could be domestic (Siriwat & Nijman, 2018). Recent studies that have observed otters illegally advertised for sale online (Gomez & Bouhuys, 2018 and Siriwat & Nijman, 2018, Aadrean, 2013).*

Live Trade: *It appears from recent studies monitoring illegal trade of otters online that newborn pups and juvenile otters are in most demand within the prices were significantly higher for juvenile pups that were given a better chance of survival (Siriwat & Nijman, 2018). They appear to have high mortality rates when they are extracted from their natural habitats (Gonzalez, 2010) but are preferred when they are younger as they are easier to tame (Feeroz* et al.*, 2011a) and more suitable for pets. Otters have been observed for sale online in a number of countries including China, Indonesia, Japan, Malaysia and in Thailand, where the possession of otters is prohibited and all native otter species are protected (Siriwat & Nijman, 2018, Gomez & Bouhuys, Aadrean, 2013). Although there is evidence that* L. perspicillata *is also targeted for the international pet trade, evidence suggests that they are less in demand than the smaller Asian Small-clawed Otter Aonyx cinereus (Gomez & Bouhuys, 2018).*

Skins: Seizures of otter skins have declined since 2005; however, the reason for this decline is not clear. Between 1980 and 2015; 2,949 otter pelts were seized in India, although few can be identified down to the species-level, given that *L. perspicillata* pelts are desirable, a significant number are likely *L. perspicillata.* Of these 787 were seized in Delhi, a northern Indian city that serves as a hub for wildlife traders moving goods into Nepal, some pelts had been marked with Tibetan script, suggesting they were likely destined for the Tibetan Autonomous Region. In Cambodia and Lao PDR, all of the specimens seized (species not specific) between 1980 and 2015 were dead specimens, mostly skins, suggesting that poaching is largely motivated by the demand for pelts.

Analysis of CITES trade data show that there have been no reported exports of L. perspicillata *skins since 1983.All exports of skins prior to 1983 were from unreported sources, except for 3,058 skins exported from Germany from pre-convention specimens. Observations of skin trade online in China found posts advertising otter fur, especially for hats but advertisements were not down to the species-level, some advertisements stated that the furs were imported from Canada or United States (Y. Wang, in litt., 2019). As there have been no legal exports of skins since 1983 as evident from CITES trade data, the origin of these skins is questionable.*

The main market for the trade in otter fur is China and more specifically in the Tibetan Autonomous Region (Yoxon & Yoxon, 2015). In the Tibetan Autonomous Region, otter pelts are used to decorate chupas, a traditional garment and according to one garment-maker, it takes three otters to decorate one chupa. *In China otters were widely harvested between the 1950s and 1980s, in 1955 a total of 25,733 otter pelts were harvested from Hubei Province alone, however due to over-exploitation the production of otter pelts declined by over 90% in provinces such as Julin, Anhui, Fujian, Guangdong and Guangxi from the 1950s to 1980s (Zhang* et al.*, 2018). In 2008 it was reported that one otter fur would sell for USD200 and therefore a high economic incentive to fuel this illegal trade (Yoxon & Yoxon, 2015), although this information is not provided at the species-level.*

A study by Gomez et al. *(2016) analysed 161 otter seizures across 15 countries in Asia between 1980 and 2015 and estimated to have involved 5,881 individuals. Of these seizures the majority (99%) consisted of otter skins in China, India and Nepal. A large number of skins (82%) were unable to be identified down to the species-level which makes it difficult to determine the level of trade for Asian otter species (Gomez* et al*, 2016).*

Medicine: *A search conducted on Chinese markets online for medicinal products derived from otters found products from otter liver advertised on Alibaba and Jingdong, however these advertisements were not at the species-level (Y. Wang, in litt., 2019), therefore making it difficult to determine the level of trade in these commodities. Otters are poached in India for skins and oil, the oil can be extracted from the fat and used for medicine with main destination markets in northern India and through trade routes in Nepal and Bangladesh (Meena, 2002). In Cambodia, otters have been reportedly used as medicine for pregnant women (Hon* et al.*, 2010).*

More detailed information on seizures and online/physical markets of otter commodities across both range and non-range States can be found below (Table 1):

Table 1: Information on seizures and online/physical markets across both range and non-range States (source SS unless referenced).

Country	Seizures	Online & Physical Markets
Cambodia (range State) Otter species present: *Aonyx cinereus*	• *Between 1980-2015 a total of 46 individuals were seized (nine live and 37 dead, not species specific) (Gomez* et al.*, 2016).*	• *In Phnom Penh, a key hub for wildlife trade in Cambodia, L. perspicillata tails were observed for sale at USD 45 per kg in 1994 to help women with labour pains (Ashwell & Walston, 2008).*

Lutra lutra *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015, Aadrean* et al., *2015 & de Silva* et al., *2015)*		
China (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980-2015 a total of 16 seizures reported 2,403 individuals (not species specific), with no reported seizures of live individuals (Gomez* et al., *2016).*	• *Searches conducted online found otter products for sale with prices varying from USD 60-120 for a complete otter pelt and USD 2,600 for a live specimen (not species specific) (Y. Wang in litt., 2019).* • *Provincial pelt trade statistics for 1953-1985 reports >10,000 individuals killed annually in Yangtze and Pearl River Basins (not species specific). In 1957 alone >40,000 otter pelts were officially traded (not species specific) (Li & Chan, 2017).* • *Searches conducted on Chinese online platforms found otter commodities for sale varying from USD 60-120 for a whole pelt and USD 2,600 (maximum price) for one specimen (Y. Wang in litt., 2019).*
India (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• Between 1980-2015 a total of 2,949 otter pelts were seized (not species specific) and many had been marked with Tibetan script suggesting international trade. • In 1993 40 skins were identified as *L. perspicillata* amongst 234 total skins seized that year in India. • *Between 1994 and August 2006 978 reported seizures (involving tiger, leopard and otter) amounting to 777 otter individuals (not species specific) (Banks* et al., *2006).*	
Indonesia (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015, Aadrean* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980-2015 one live individual was seized (not species specific) (Gomez* et al., *2016).* • *A total of three seizures between 2015-2017 amounting to seven live individuals (four Lutrogale perspicillata and three unidentified), this shipment reportedly originated from East Kalimantan, however the intended destination for domestic or international markets was unknown (Gomez & Bouhuys, 2018).*	• *Five-month study online between January and May 2012 found a total 63 posts from 46 sellers, only one post was identified as L. perspicillata (Aadrean, 2013).*
Japan (non-range State) Otter species present: *Enhydra lutris* *(Doroff & Burdin, 2015)*	• *Between 1980-2015 one seizure was reported in Japan involving one individual (not reported if live or dead and not species specific) (Gomez* et al., *2016).* • *A total of three seizures between 2015-2017 amounting to 32 live otters (not species specific) (Gomez & Bouhuys, 2018).*	• *A study of online markets identified 12 sellers were selling/sold a minimum of 85 individuals since 2011 of which most (87%) were A. cinereus and the remainder were unidentified (Kitade & Naruse, 2018).* • *A total of 20% were reportedly imported and 46% captive-bred and only vague information was provided on domestic captive breeding sources, a domestic zoo was reported to be the origin of three individuals (Kitade & Naruse, 2018).*

		• *This study also suggested the possibility of specimens imported from Malaysia to Japan for the purpose 'zoo' were subsequently being sold into the pet market (19 were imported with source code "Z" between 1977-2018 (CITES Trade Database, 2019).*
Lao PDR (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• Between 1980 and 2015 all specimens seized were dead, mostly skins, *involving 23 individuals (not species specific) (Gomez* et al., *2016),* suggesting that poaching is largely motivated by the demand for pelts.	
Malaysia (range State) Otter species present: *Aonyx cinereus* *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980-2015 a total of 14 live individuals were seized (not species specific) (Gomez* et al., *2016).*	• *Between January and May 2017, a total of 10 online advertisements recorded including four live L. perspicillata for sale (Gomez & Bouhuys, 2018).*
Myanmar (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• *Between 1980-2015 a total of two individuals seized (not species specific) and no seizures of live individuals (Gomez* et al., *2016).*	
Nepal (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutrogale perspicillata* (*Wright* et al., *2015, Roos* et al., *2015 & de Silva* et al., *2015)*	• Between 1989-2017 a total of 756 otter pelts were seized (not species specific). • *Between 1980-2015 a total of nine seizures involving 383 individuals (not species specific), of which there were no reports of live individuals seized (Gomez* et al., *2016).*	
Philippines (non-range State) Otter species present: *Aonyx cinereus* (*Wright* et al., *2015)*	• *Between 1980-2016 a total of six live individuals were seized (one live and five dead, not species specific) (Gomez* et al., *2016).*	
South Korea (non-range State) Otter species present:	• *Between 1980-2015 there was one reported seizure involving one individual (not reported if live or dead and not species specific) (Gomez* et al., *2016).*	

Lutra lutra *(Roos* et al.*, 2015)*		
Tibet Autonomous Region (not present) Otter species present: *Lutra lutra (plateau region)* *(Roos* et al.*, 2015)*		• *In 2005 and 2006 over 1,800 otter skins were observed at Linxia market, including L. perspicillata and Lutra lutra documented (Banks* et al.*, 2006).*
Thailand (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al.*, 2015, Roos* et al.*, 2015, Aadrean* et al.*, 2015 & de Silva* et al.*, 2015)*	• *Between 1980-2015 a total of 14 individuals were seized (12 live and two dead, not species specific) (Gomez* et al.*, 2016).* • *A total of five seizures between 2015-2017 involving 35 live individuals (24 A. cinereus and 11 unidentified), of these seizures four indicated international trafficking and at least three were destined for Japan (Gomez & Bouhuys, 2018).* • *In 2013, six live A. cinereus and five L. perspicillata were seized at Suvarnabhumi International Airport (Kitade & Naruse, 2018).*	• *160 online adverts totalling 337 live individuals (between March 2017 – April 2018) of which 17 posts included L. perspicillata (29 individuals), this study also suggested international trade between Thailand and Malaysia, however this was not verified (Siriwat & Nijman, 2018). Lutrogale perspicillata was found to be offered at a higher price than A. cinereus.* • *Between January and May 2017, a total of 80 online advertisements recorded three L. perspicillata for sale (Gomez & Bouhuys, 2018).*
Viet Nam (range State) Otter species present: *Aonyx cinereus* *Lutra lutra* *Lutra sumatrana* *Lutrogale perspicillata* (*Wright* et al.*, 2015, Roos* et al.*, 2015, Aadrean* et al.*, 2015 & de Silva* et al.*, 2015)*	• *Between 1980-2015 a total of 13 live individuals were seized (not species specific) (Gomez* et al.*, 2016).* • *A total of three seizures between 2015-2017 amounting to 15 live otters (six A. cinereus and nine unidentified) and at least one incident indicated international trafficking as the otters were claimed to have been sourced from Thailand (Gomez & Bouhuys, 2018).*	• *Between January and May 2017, a total of 21 online advertisements recorded 12 A. cinereus and 15 unidentified otters for sale (Gomez & Bouhuys, 2018).*

Additional Information

Threats

In northern South-east Asia, international trade has probably been a substantial part of the problem causing declines, but cannot be considered the only, nor even the main driver of illegal offtake (W. Duckworth, in litt., 2019). In addition to international and domestic trade, throughout its range *L. perspicillata* faces threats including human development and natural wetland conversion for agriculture including aquaculture, plantations (e.g. coffee, palm oil, tree plantations and rice fields), wetland reclamation; siltation due to deforestation; pollution by pesticides; mining; quarrying; slash-and-burn agriculture. *The global extent of wetlands is now estimated to have declined between 64-71% since 1900 and habitat loss and degradation continues worldwide (Ramsar, 2015). Evidence suggests that although available otter habitat is decreasing and wetlands continued to be converted into agriculture throughout much of* L. perspicillata *range, otters are adapted to survive in moderately disturbed habitats where there is some level of human presence (Mohapatra* et al.*, 2014,).*

A further threat to otters is a reduced prey base from overfishing and the species' preference for crabs and other invertebrates makes water contamination from inappropriate or illegal pesticide use in wetlands, as well as organochlorides, heavy metals and other pollutants an important concern. The common practice of dumping garbage in the wetlands is also a threat to otters, whose scat has been observed to contain plastics. Fishermen are known to kill otters as a competitor for fish. Destructive fish killing practices are common in the Western Ghats where dynamite fishing, bleach fishing and electric-rod based fishing and the use of pesticides in banana plantations and rice fields.

Climate change is predicted to seriously impact otter populations worldwide. *L. perspicillata,* like all otter, depend on rivers, lakes and streams, which face dramatic alteration under a warming scenario, reducing water levels in long-term droughts and affecting prey densities.

Conservation, management and legislation
Lutrogale perspicillata has legal protection in range States except Cambodia, Brunei Darussalam (status in Bangladesh unclear). No management plants or species-specific conservation measure are known to be in place for *L. perspicillata* in range States, although development of a management plan is suggested under the Pakistan 2018 National Biodiversity Strategy and Action Plan. *Lutrogale perspicillata* has been included in CITES Appendix II since the subfamily Lutrinae was listed in 1977. *Lutrogale perspicillata* inhabits protected areas in numerous countries and some, including India and Nepal, provide enhanced protection for otters. *In Karnataka, India, the first-ever dedicated otter conservation reserve has been developed in 2016 and reserved a 34km stretch along the Tungabhadra river (Duplaix & Savage, 2018).*

Artificial propagation/captive breeding
There have been reports of otter farms, in China (Y. Wang, in litt., 2019), and Pakistan (IOSF, 2014). In both Indonesia and Thailand there have been unconfirmed/unverified reports of otters being bred for the purpose of trade (Gomez & Bouhuys, 2018). It is possible that juveniles and newborn pups which are extracted from the wild are then sold as captive-bred or used as breeding stock. In the wild it has been observed that the average litter size is 1-5 pups (Hwang & Larivière, 2005) and in captivity L. perspicillata *was observed to produce litter sizes between 2-5 (mean =3.25) (Desai, 1974).*

Lutrogale perspicillata are bred in many zoos across South and South-east Asia (including in Thailand, Cambodia, Viet Nam, Malaysia and Singapore). Few European or American Zoos hold or breed *L. perspicillata*. The Twycross Zoo in the United Kingdom was the first in the Western Hemisphere to breed *L. perspicillata* in 1972. Breeding for reintroduction purposes has not been attempted.

In Bangladesh, fishing with otters has been practiced for the last two hundred years (Feeroz et al.*, 2011b). A small semi-captive population exists in Noraile district which is used for traditional fishing, however this population has also been found to have declined by more than 80% in the last two decades (Feeroz, 2014).*
A study on the breeding activities of L. perspicillata *in Bangladesh (Feeroz* et al.*, 2011b) in different captive conditions. The study found that 90% of adult females gave birth at 3-4 years of age in semi-captive conditions and overall breeding success recorded for the study was 90%. This study proposed the development of an ex-situ management plan for* L. perspicillata *in Bangladesh. The generation length is said to be 10 years (de Silva* et al.*, 2015).*

Implementation challenges (including similar species)
Unprocessed fur of otters allows for distinction of specimens at the species-level (Kuhn, 2009). Although, mammalian hair microscopy for species identification is widely used for ecological, forensic and/or economic purposes, few such guidelines are publicly availably and cover only some otter species. There is very little published material that would allow the simple distinguishing of processed pelts, therefore, it would be challenging for untrained officials to distinguish between processed pelts (Gomez et al.*, 2016).*

Analysis of CITES Trade Data indicates that legal trade in this species appears to be relatively small. Much of the trade of L. perspicillata *is illegal (and some domestic) and therefore inclusion in Appendix I may not have any significant impact on the threat from international trade of this species.*

In some consumer countries such as Japan, there are loopholes that allow Appendix II (and III) species to be traded once smuggled into the country and therefore CITES does not fully protect the animal from illegal trade (Duplaix & Savage, 2018).

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Remove the existing annotation for the population of Eswatini of Southern White Rhinoceros *Ceratotherium simum simum* listed in Appendix II

Proponent: Eswatini

Summary: The Southern White Rhinoceros *Ceratotherium simum simum* is one of two subspecies of White Rhinoceros (the other being the Northern White Rhinoceros *C. s. cottoni,* now believed extinct in the wild). In 2012 the global wild population was estimated at around 21,300, having increased from a few hundred at most in the 1920s. Owing to a combination of increased poaching since 2008 (particularly in Kruger National Park, South Africa), and drought in southern Africa (which has now eased in parts), numbers declined to around 18,000 in 2017. From 2015-2018, the number of rhinos known to have been poached in Africa is estimated to have declined by a third. Although poaching remains at a high level particularly in Mozambique, South Africa and Zimbabwe, provisional data for 2018 indicate that the numbers of rhino killed by poachers per day (2.6) declined to its lowest level since 2012. *Ceratotherium simum simum* was assessed as Near Threatened on the IUCN Red List in 2011. Around 86% of the population is in South Africa.

The Rhinocerotidae family was included in Appendix I in 1977. The South African population of *C. s. simum* was transferred to Appendix II in 1994 with 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 a proposal was accepted to transfer Eswatini's population to Appendix II using the same annotation.

Having become extinct in Eswatini in the mid-20th century, *C. s. simum* was reintroduced to the country from South Africa in 1965. The population reached a peak of around 120 in the late 1980s but was reduced to around 20-30 animals in the early 1990s by poaching.

The population is confined to secure sites in two protected areas. Improved protection, including through a change to national legislation, led to an increase in the population to 60 individuals in 2004. In 2015 the population numbered 90 individuals, which then reduced to 66 in 2017 due to drought, and is currently estimated at 79. Three rhinos have reportedly been poached in the country since 1992, although it is possible that not all poached carcasses have been discovered so this may be an underestimate.

According to the proponent, no trophy hunting of *C. s. simum* has taken place because all rhinos occur in reserves where sport and trophy hunting are not permitted. All reported trade from Eswatini has been to South Africa; since 2004 Eswatini exported 19 live individuals to South Africa (and imported 28 animals).

On a continental scale, the cost and risk of keeping rhinos has risen and many private owners are now reported to be leaving the market. In Eswatini, the recent drought meant the rhinos were fed fodder at great cost.

This proposal is to delete the existing annotation as it applies to Eswatini's population, with the intention of allowing limited and regulated trade in stockpiles of *C. s. simum* horn which has been legally collected in the past or recovered from poached Eswatini rhino (totalling 330 kg), as well as horn to be harvested annually in a non-lethal way in the future (amounting to up to 20 kg per year). The proponent notes that it would reserve the right to adjust prices and amounts adaptively once sales commence.

The Supporting Statement provides the following details on implementation: the CITES Management Authority of Eswatini will be the sole seller, and will sell to a small number of licensed retailers (likely including Traditional Chinese Medicine (TCM) hospitals in the Far East). Horn will be “properly documented, certificated and recorded on a DNA database, a national register and with the CITES

Secretariat to safeguard its integrity". The CITES Secretariat will be requested to closely monitor consignments, and trade will be open to inspection and verification by the CITES Secretariat. If legal trade is ultimately proven to pose a renewed threat to the subspecies, then further trade would be prohibited by Eswatini. The Proponent states that its intention is to use proceeds from the horn sales to fund conservation, including security and improved park employee pay.

Analysis: Removal of the annotation would mean that all specimens of *C. s. simum* exported from Eswatini would be subject to Appendix II regulation. There are no specific guidelines for assessing proposals to change annotations of this nature, but it seems appropriate to ensure that satisfactory precautionary measures, as detailed in Annex 4 of *Res. Conf. 9.24 (Rev. CoP17)*, remain in place:

Annex 4 2 a) i): the subspecies is in demand, and the proposed amendment has the potential to stimulate trade (it is unlikely that 20 kg per year will meet global demand). It is not possible to predict if legalising trade in rhino horn from one population will stimulate trade in other populations. While legal trade could replace some of the demand currently being met by illegally obtained horn, raise funds for conservation and/or reduce the "exclusive" status of horn to certain consumers, legalisation could also lead to new consumers entering the market who had previously been put off by its illegality. The proponent states that if the trade were judged to be having a negative impact on the subspecies it would be stopped, although no clear mechanism is proposed for how such an assessment would be undertaken.

Annex 4 2 a) ii): management measures in place since 2004 have seen the Eswatini population increase, despite a recent drought-induced decline. Few details are provided as to how the proposed legal trade will be carried out and controlled; for example it is not specified which importing countries would permit a legal trade (China recently reaffirmed its 25-year ban on the use of rhino horn for TCM), how retailers (including international) would be selected, how and by whom these would be licensed, or how trade would be monitored throughout the trade chain (including in end-user markets) to avoid laundering, and who would fund this. While the CITES Secretariat is identified as playing a significant role, it is not clear how it would undertake this work, similarly it is not clear if authorities in importing countries have been consulted.

Eswatini has provided some detail on precautionary measured that they would implement, but it is not clear what safeguards would be implemented by any anticipated trade partners or even which countries would be able legally to import the horn. In summary, this proposal does not provide sufficient information to address the precautionary measures in Annex 4 to *Res. Conf. 9.24 (Rev. CoP17).*

Removal of the annotation would also remove the constraint that live animals be exported only to "appropriate and acceptable destinations" (*Res. Conf. 11.20 (Rev. CoP17)*). In the period that this annotation has applied, Eswatini has only exported live individuals to South Africa (whose own population of this subspecies would remain covered by this annotation) and it is not known if Eswatini would begin exporting to other countries.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Ceratotherium simum simum is one of two subspecies of *Ceratotherium simum* (the other being the Northern White Rhinoceros *C. s. cottoni,* now believed extinct in the wild).

Population Size and Range

Table 1. Estimated number of C. s. simum by country as of the end of 2017, and including global totals as of the end of 2017, 2015 and 2012 (adapted from Emslie et al., 2019).

Country	Population
South Africa	*15,625*
Namibia	*975*
Kenya	*510*
Botswana	*452*
Zimbabwe	*367*
Eswatini	*66*
Mozambique	*29*
Uganda	*22*
Zambia	*14*
*Senegal**	*3*
*Cote d'Ivoire**	*1*
End 2017 Total	***18,064***
End 2015 Total	***20,053***
End 2012 Total	***21,316***

** Outside of natural range*

From 2015-2018, poaching of all rhinoceros species in Africa is thought to have declined by a third, but in 2018 still remained at a high level particularly in Mozambique, South Africa and Zimbabwe with a provisional reported poaching rate of 2.6 rhinos per day at the continental level in 2018 (Emslie et al. 2019).

IUCN Global Category

Globally assessed as Near Threatened (assessed 2011, ver. 3.1).

Summary of Population, Distribution and Rhino Horn in Eswatini

Eswatini's rhino population occurs in two parks, Hlane Royal National Park (est. 1967) *(217 km^2 (Protected Planet, 2019))* and Mkhaya Game Reserve (est. 1980) *(100 km^2 (Protected Planet, 2019)).* White rhino are also likely to be placed in the Mlilwane Wildlife Sanctuary (est. 1961) *(46 km^2 (Protected Planet, 2019)* in the future.

Ceratotherium simum simum became extinct in Eswatini due to hunting under colonial rule, but was re-introduced to Mlilwane Wildlife Sanctuary in 1965 using animals from South Africa. More animals were sent to Hlane Royal National Park in the late 1960s and to Mkhaya Game Reserve when secure habitat became available in the 1980s. The population increased in Hlane and Mkhaya Parks to approximately 120 animals by 1988, but poaching during the "rhino war" of 1988-1992 reduced the number to 24 (*33 in 1993 - with rhino poaching in the country peaking in 1991 (R. Emslie, in litt., 2019)*). Three rhino have been poached in Eswatini since (two in 2011 and one in 2014), and the population in Eswatini as of the end of 2017 was 66 after recent drought mortalities. *The population has since increased to 79 following good calving rates in response to better rains, emergency supplementary feeding and lower post-drought densities of competing grazers (R. Emslie, in litt., 2019).*

Legal rhino horn is kept in stockpiles in various places within Eswatini. This horn has been legally collected from natural deaths, horn knock-offs and legitimate management actions (including dehorning and horn-tipping for translocation) over many years, or has been recovered from illegally hunted Eswatini rhino.

The mean weight of adult horns is 5.16 + 2.0kg for the front horn, and 1.86 + 1.0kg for the posterior horn. Their horns grow continuously throughout their life at approximately one kg per year.

Since 2004 Eswatini has sold or exchanged, and exported white rhino bulls to South Africa, and imported white rhino cows and bulls for genetic and sex ratio purposes. *Eswatini reported exporting 19 live C. s. simum between 2004 and 2016 (all of which went to South Africa), and importing 28 live animals from South Africa during the same period (CITES Trade Database, 2019).*

Precautionary Measures

Species likely to be in demand for trade, but its management is such that the CoP will be satisfied with:

A) Implementation by the range States of the requirements of the Convention, in particular Article IV; and

B) appropriate enforcement controls and compliance with the requirements of the Convention

Export quota or other special measure

This proposal is for Eswatini to sell from existing stock 330 kg of rhino horn and also up to 20 kg per year, including harvested horn. Eswatini gives estimated average weights of front horn per individual of 5.2kg. *Dehorning every 18 months is said to generate approximately 0.75 kg and 1.5 kg per adult female and male*

white rhino respectively (J. Hume in litt. in Emslie et al., 2019). Research of stress levels shows little impact of regular dehorning every 18 months in a South African white rhino population that is breeding well (R. Emslie. in litt., 2016). Other than indicating that it will be non-lethal, the Supporting Statement (SS) 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 its breeding performance (Emslie, R. in litt., 2016).

The SS provides the following detail on how the horn trade will be controlled:

Big Game Parks, the CITES Management Authority of Eswatini, will be the sole seller and horn will be sold directly to a small number of licenced retailers, which is likely to include Traditional Chinese Medicine (TCM) hospitals in the Far East. All horn offered for sale will be properly documented, certificated and recorded on a DNA database, a national register and with the CITES Secretariat to safeguard its integrity *(the DNA database will be RhoDIS (Rhino DNA Index System) (R. Emslie, in litt.. 2019).* All traded specimens will carry DNA certificates and the Secretariat will be requested to closely monitor consignments. The retailers will be licensed and will qualify by undertaking not to trade horn from illegal sources. The breach of such will disqualify such traders. The trading operation will be open to inspection and verification by the CITES Secretariat. If, for some unexpected reason, a legal trade is ultimately proven to pose a renewed threat to the species, then the trade will be closed down by Eswatini.

The proceeds from the sale of stocks should raise approximately USD 9.9 million, and will be placed in a conservation endowment fund. The proceeds of the annual sale of up to 20 kg of horn will raise a further USD 600,000 per annum bringing total recurrent annual income from horn to USD 1.2 million. Eswatini would reserve the right to adjust prices and amounts adaptively once sales commence. The SS states that the proceeds from the sale of horn are needed by Eswatini's rhino parks to cover security costs, improved pay of park employees, additional infrastructure and equipment, range expansion and supplementary food during periods of drought, and that proceeds will also be used to provide for sustainable long-term developments, all of which will strengthen species protection and other nature conservation initiatives, while also benefitting neighbouring rural communities and the country as a whole.

Additional Information

Threats

Despite losing only one Southern White Rhinoceros to poaching in the period 2012-2017, Eswatini's population declined by 21% over the same period due to severe drought (Emslie et al., 2019), which affected southeast Africa between 2015 and 2017 and reduced the white rhino population to 66 animals. It was necessary to provide supplementary fodder, which was said to be difficult to find and very expensive to buy and store. Some rhino were lost as not all animals adapted readily to being artificially fed. In addition to the loss of adult stock, calves either perished or were orphaned, and some were hand-reared at a cost of at least USD 12,000 per calf to raise.

Conservation, management and legislation

The entire rhinoceros family, the Rhinocerotidae, was included in Appendix I of CITES in 1977. The South African population of C. s. simum 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, a proposal to transfer the population of Eswatini (then Swaziland) to Appendix II using the same annotation was accepted. In agreeing to this transfer, the CoP agreed that the precautionary measures in Annex 4 of Res. Conf. 9.24 (Rev. CoP17) were met based on management measures described in the Supporting Statement.

Rhino poaching and trafficking offences are punishable in Eswatini by mandatory custodial sentences of 5 – 15 years without the option of a fine, plus replacement of the poached rhino. While undoubtedly occurring, levels of illegal trafficking through Eswatini remain low—one trafficking syndicate was disrupted when two individuals from Taiwan Province of China were arrested with 36 kg of white rhino horn (the horn originated in South Africa). Both accused were sentenced to 29 years imprisonment without the option of a fine and ordered to replace the rhinos poached or compensate the owners, failing which they will each serve a further four years imprisonment.

There is no trophy hunting of *C. s. simum* in Eswatini because all rhinos occur in reserves where sport and trophy hunting is not permitted. Despite the provisions of Eswatini's annotation, no trophy hunting has taken place since its formal approval by CITES 15 years ago. *According to data in the CITES Trade Database, one live rhino was exported to South Africa in 2015 from Eswatini for the purpose of hunting (CITES Trade Database, 2019).*

In China, on 29th October 2018, a Notice by the General Office of the State Council on Strict Rules of Activities of Operation and Utilization of Rhinoceros and Tiger and their Products repealed a longstanding prohibition on domestic use of rhino horn spanning two and half decades since adoption in 1993. Among the provisions of the Notice was a measure that could allow licensed medicinal use of rhino horn from captive-bred animals by approved TCM doctors in accredited hospitals. However, on 12th November 2018, a statement by State Council Executive Deputy Secretary-General Ding Xuedong announced that the detailed regulations for implementation of the October Notice had been "postponed after study", and that the strict bans on domestic rhino horn use and trade would continue to be enforced (China.org, 2018).

Implementation challenges (including similar species)

The only similar species is the Black Rhinoceros *Diceros bicornis* of which there are approximately 5,500, including 21 in Eswatini. This proposal is restricted to trade in *C. s. simum* only.

References

China.org (2018). Full transcript: *State Council Executive Deputy Secretary-General Ding Xuedong Answers Media Questions*. Retrieved on 4th March 2019 from http://www.china.org.cn/china/Off_the_Wire/2018-11/12/content_71941870.htm.

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

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Emslie, R. Milliken, T., Talukdar, B., Burgess, G., Adcock, K., Balfour, D.& Knight, M. H. (compilers) (2019). *African and Asian Rhinoceroses – Status, Conservation and Trade. A report from the IUCN Species Survival Commission (IUCN SSC) African and Asian Rhino Specialist Groups and TRAFFIC to the CITES Secretariat pursuant to Resolution Conf. 9.14 (Rev. CoP17).*

Transfer of the population of Namibia of Southern White Rhinoceros *Ceratotherium simum simum* from Appendix I to Appendix II with an annotation

Proponent: Namibia

Summary: The Southern White Rhinoceros *Ceratotherium simum simum* is one of two subspecies of White Rhinoceros (the other being the Northern White Rhinoceros *C. s. cottoni,* now believed extinct in the wild). In 2012 the global wild population was estimated at around 21,300, having increased from a few hundred at most in the 1920s. Owing to a combination of increased poaching since 2008 (particularly in Kruger National Park, South Africa), and drought in southern Africa (which has now eased in parts), numbers declined to around 18,000 in 2017. From 2015-2018, the number of rhinos known to have been poached in Africa is estimated to have declined by a third. Although poaching remains at a high level particularly in Mozambique, South Africa and Zimbabwe, provisional data for 2018 indicate that the numbers of rhino killed by poachers per day (2.6) declined to its lowest level since 2012. *Ceratotherium simum simum* was categorised on the IUCN Red List as Near Threatened in 2011. Around 86% of the population is in South Africa.

The Rhinocerotidae family was included in Appendix I in 1977. This proposal is to transfer Namibia's population of *C. s. simum* to Appendix II with 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." The populations of South Africa and Eswatini are already included in Appendix II with this annotation (since 1995 and 2005 respectively).

Having become extinct in Namibia before the end of the 19th century, *C. s. simum* was first reintroduced to Namibia in 1975 when 16 animals were imported from South Africa. The population was estimated at 293 in 2005, and the most recent population estimate (2017-2018) is nearly 1,100, almost 800 of which are reported to be in private ownership across 70 populations, with the remainder in national protected areas.

This increase is due to both an intrinsic population increase and imports of live animals from South Africa: between 2002 and 2017 South Africa recorded the export of nearly 400 *C. s. simum* to Namibia, 80% of these from 2012 onwards. In the same time period less than 50 rhino were imported from Namibia (the largest importer being the Democratic Republic of the Congo), all of which were reported after 2010.

From 2008 to 2018 a total of 57 *C. s. simum* were legally hunted in Namibia, indicating an average annual offtake of 0.5% of the population. Virtually all resulting trophies appear to have been exported.

Reported poaching in Namibia has until recently been at a very low level (three animals poached in total for the years 2008-2013). Poaching has increased but is still at a relatively low level (average of nine animals per year for 2015-2018) and is lower than the intrinsic population growth rate. However, poaching of Black Rhinoceros *Diceros bicornis* in Namibia has been much higher: averaging approximately 50 animals per year for the period 2014-2018 (2.4% of the current population per year), although for both taxa not all poached carcasses will have been discovered so this may be an underestimate. Due to increasing security costs which are reported not to be offset by available means of utilisation, a future reduction in private ownership is considered a significant threat.

Ceratotherium simum simum is classified as a "Specially Protected" species under Namibian legislation. Permits are needed for possession of live animals or their parts, and for utilisation, movement, imports and exports. Transport or hunting permits are only issued if the rhino in question has been microchipped and DNA profiled with samples sent to the RhODIS database. Only Namibia-registered game dealers are allowed to capture and trade wild animals and only Namibia-registered professional hunters and operators are allowed to conduct hunting.

Analysis: The Namibian population of *Ceratotherium simum simum* does not have a restricted distribution. Its population is relatively small, but is increasing owing to a combination of intrinsic population growth and imports. Nearly 80% of the population is in around 70 privately-owned subpopulations. Although the poaching rate has increased, it is currently less than 1% of the population annually, which is lower than the intrinsic population growth rate. Overall, the Namibian population does not meet the biological criteria for retention in Appendix I.

The species is in demand for international trade. The proposed annotation, which restricts the kinds of specimens and type of export trade to be permitted, can be considered a special measure under the terms of the precautionary measures in Annex 4 of *Res. Conf. 9.24 (Rev. CoP17).* Namibia already undertakes such trade under the Appendix I listing and has a system in place to licence and track specimens in trade.

The annotation in question has been used for export of this subspecies from South Africa and Eswatini for several years with no apparent problems.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Ceratotherium simum simum is one of two subspecies of *Ceratotherium simum* (the other being the Northern White Rhinoceros *C. s. cottoni,* now believed extinct in the wild**).**

Population Size and Range

Namibia had the second largest population of *C. s. simum* (Table 1).

Table 1. *Estimated number of C. s. simum by country as of the end of 2017, including global totals as of the end of 2017, 2015 and 2012 (adapted from Emslie et al., 2019).*

Country	***Population***
South Africa	*15,625*
Namibia	*975*
Kenya	*510*
Botswana	*452*
Zimbabwe	*367*
Eswatini	*66*
Mozambique	*29*
Uganda	*22*
Zambia	*14*
*Senegal**	*3*
*Cote d'Ivoire**	*1*
End 2017 Total	***18,064***
End 2015 Total	***20,053***
End 2012 Total	***21,316***

** Outside of natural range*

From 2015-2018, poaching of all rhinoceros species in Africa is thought to have declined by a third but in 2018 still remained at a high level particularly in Mozambique, South Africa and Zimbabwe with a provisional reported poaching rate of 2.6 rhinos per day at the continental level in 2018 (Emslie et al. 2019).

IUCN Global Category

Globally assessed as Near Threatened (assessed 2011, ver. 3.1).

Biological criteria for inclusion in Appendix I

A) Small wild population

See C) below.

B) Restricted area of distribution

Ceratotherium simum simum habitat in Namibia is limited by the minimum suitable rainfall per annum. Apart from more than 1.5 million ha of white rhino habitat in three national parks currently occupied by white rhinos, an estimated additional 0.5 million to 1 million ha of habitat is available in national parks currently without white

rhinos and could be restocked in future. Suitable habitat in the country is said to be able to support a white rhino population of around 14,000.

The distribution of the species is precisely known in Namibia as it was reintroduced to specific national parks and private land units. This species is being reintroduced to new areas in its former range.

C) Decline in number of wild individuals

Having become extinct in Namibia before the end of the 19th century, *C. s. simum* was first reintroduced to Namibia in 1975 when 16 animals were imported from South Africa. The population was estimated at 1,037 in 2017–2018 *and more recently at around 1,080 (R. Emslie, in litt., 2019)* (Table 2). Nearly 800 rhino were reported to be in private ownership across 70 populations, with the remainder in national protected areas.

Between 2002 and 2018 the population grew at an average annual rate of 6.7%, including both intrinsic population increase and imports of live animals from South Africa: between 2002 and 2016 South Africa reported the export of just over 300 white rhino to Namibia, 80% of these since 2012.

***Table 2**: Estimates of Southern White Rhino population size in Namibia.*

Year	Population size	Source
1975	*16 (imported from South Africa)*	*SS*
2005	*293*	*CoP14 Doc. 54*
2007	*370*	*CoP15 Doc. 45.1 Rev. 1 Annex*
2010	*469*	*CoP16 Doc 54.2 Rev.1*
2015	*822*	*CoP17 Doc. 68 Annex 5*
2017-2018	*1,037*	*SS*
Current	*1,080*	*R. Emslie, in litt., 2019*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

The proposal is for trade in live animals to appropriate and acceptable destinations and hunting trophies only. Both live animals and trophies are currently in trade from Namibia under the Appendix I-listing. The proponent states that transferring the population to Appendix II will enable Namibia to export live animals and hunting trophies to more countries and will increase revenue through sustainable use.

From 2008 to 2018 a total of 57 white rhino were legally hunted in Namibia, indicating an average annual offtake of 0.4-0.5% of the population, considerably below the rate of recruitment. *Between 2008 and 2017, Namibia reported exporting 47 trophies (reported as Ceratotherium simum and Ceratotherium simum simum), in addition to a number of other commodities exported for hunting trophy purposes including 10 horns (CITES Trade Database, 2019).*

Between 2008 and 2018, under the current Appendix-I listing, Namibia exported 27 live white rhinos to Angola, Cuba, the Democratic Republic of the Congo and South Africa. *According to the CITES Trade Database, importers reported importing 46 live white rhino from Namibia: the Democratic Republic of the Congo (32), China (eight), Cuba (five) and South Africa (one). In the same time period, Namibia only reported exports of seven live white rhino (CITES Trade Database, 2019). Between 2002 and 2016 South Africa reported the export of 371 live white rhino to Namibia, 80% of these from 2012 onwards (CITES Trade Database, 2019).*

Reported incidents of illegal killing of *C. s. simum* in Namibia have until recently been at a very low level (three animals poached in total for the years 2008-2013) (Figure 1). Poaching has increased since then to an average of nine animals per year for 2015-2018 (see graph). *Emslie et. al.* (*2019) report total poaching of 291 Black and Southern White Rhinoceros in Namibia in the period 2011 to 21 November 2018. Poaching of Black Rhinoceros in Namibia has been much higher than Southern White Rhinoceros, averaging approximately 50 animals per year for the years 2014-2018 which represents around 2.4% of the current Black Rhinoceros population per year (based on information in the SS and Emslie, et al. 2019).*

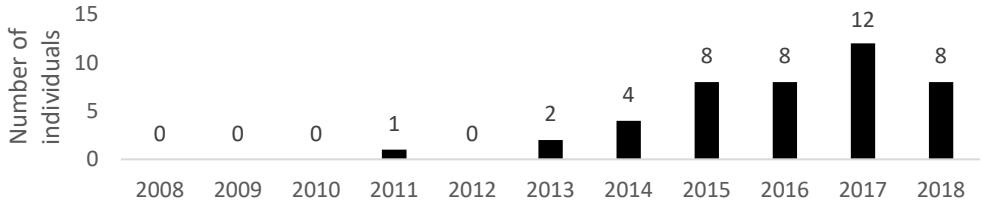

Figure 1: Numbers of *Ceratotherium simum simum* known to have been poached in Namibia (from SS).

In their report to CoP17, the IUCN African and Asian Rhinoceros Specialist Groups and TRAFFIC suggested Parties consider adding Namibia (as well as China) to the list of countries for priority attention by the CITES Rhino Working Group because of recent escalation of poaching at that time in the country (CoP17 Doc. 68). Namibia provided an update on its implementation of Res. Conf. 9.14 (Rev. CoP17) Conservation of and trade in African and Asian rhinoceroses to SC70 (SC70 Doc. 56 A9). The CITES Rhino Working Group recommended that Namibia continue to be considered a candidate priority country and be encouraged to take action to ensure prosecutions are concluded swiftly and appropriate penalties are imposed on offenders and to report convictions achieved to the CITES Secretariat (SC70 Doc. 56). This recommendation was supported by SC70 (SC70 Sum. 12 (Rev. 1)).

Precautionary Measures

Species likely to be in demand for trade, but its management is such that the CoP will be satisfied with:

A) Implementation by the range States of the requirements of the Convention, in particular Article IV; and

B) appropriate enforcement controls and compliance with the requirements of the Convention

In Namibia, *C. s. simum* is classified as Specially Protected Game (wild animals) under the Nature Conservation Ordinance 4 of 1975 as amended, and any rhino (or any portion thereof), as well as any product derived from a white rhinoceros is classified as a Controlled Wildlife Product under the Controlled Wildlife Products and Trade Act (Act 9 of 2008) as amended. This means that permits are required for possession of white rhinos or their parts, and for utilisation, movement, imports and exports. The Ministry of Environment and Tourism permit office at Windhoek issues all permits relating to white rhinoceros and their parts or derivatives.

Private owners only receive transport or hunting permits if the relevant individuals have been microchipped and DNA profiled with samples send to the RhODIS database housed at Onderstepoort, South Africa. Private owners have their own monitoring systems and stud books. Only Namibian registered game dealers are allowed to capture and trade wild animals. In the case of hunting, only Namibian registered professional hunters and operators/outfitters are allowed to conduct hunting. Domestic consumptive use of white rhinoceros and trade in rhinoceros horn and other products is currently not permitted in Namibia.

Export quota or other special measure

The proposal is for trade in live animals to appropriate and acceptable destinations and hunting trophies only.

Additional Information

Threats

As noted above, ongoing poaching of *C. s. simum* for international trade is of concern. The proponents state that another serious threat is that private owners will dispose of their rhinos, which constitute the largest part of the Namibian population, if the costs of rhino protection cannot be offset by the available means of utilisation of and trade in this species. The costs of protection have been and can be expected to continue escalating, increasingly making rhino “a liability to conservation authorities, private and communal landowners”. Existing benefit streams from tourism, limited trophy hunting and live sales of rhinoceros are reported not to be sufficient to offset these security costs.

Conservation, management and legislation

See “precautionary measures” above. Almost 17% of the land surface of Namibia has been designated as protected areas, including approximately 30% of the potential habitat for *C. s. simum*.

Implementation challenges (including similar species)

The only similar species is the Black Rhinoceros *Diceros bicornis* of which there are approximately 5,500, including 1,857 in Namibia. This proposal is restricted to trade in *C. s. simum* only.

References

Emslie, R. Milliken, T., Talukdar, B., Burgess, G., Adcock, K., Balfour, D.& Knight, M. H. (compilers) (2019). *African and Asian Rhinoceroses – Status, Conservation and Trade A report from the IUCN Species Survival Commission (IUCN SSC) African and Asian Rhino Specialist Groups and TRAFFIC to the CITES Secretariat pursuant to Resolution Conf. 9.14 (Rev. CoP17).*

Emslie, R. (2019). *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

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, including an annotation agreed at CoP14. The annotation allowed 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 allowed 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 was 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 (the sale of ivory in question took place in November 2008). It also specified that such further proposals should be dealt with in accordance with Decisions 14.77 and 14.78.

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 and the CoP agreed Decision 16.55 which again directed the Standing Committee, with the assistance of the Secretariat, to propose for approval 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. This Decision was also not implemented as no such decision-making mechanism was submitted to CoP17. Parties at CoP17 did not agree to any extension of the work on the development of the decision-making mechanism.

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 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 was instructed, 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.

At CoP17, Parties agreed to incorporate the provisions of the Decision into *Res. Conf. 10.10 (Rev. CoP17) Trade in elephant specimens* and these are now paragraph 11 of that Resolution.

A further issue of note in recent years has been the issue of domestic ivory markets, with many countries significantly increasing level of restrictions imposed on the sale of ivory nationally, such as China's landmark closure of its domestic ivory market. Amendments to *Res. Conf. 10.10* were adopted at CoP17, recommending that Parties in whose jurisdiction there is a legal domestic market for ivory that is contributing to poaching or illegal trade take all necessary legislative, regulatory and enforcement measures to close their domestic markets for commercial trade in raw and worked ivory.

In response to growing concerns over levels of illegal ivory trade, a process to address this through development of National Ivory Action Plan's (NIAPs) was initiated within CITES. This involves key Parties implicated in the global illicit ivory trade developing country-specific action plans that outline urgent actions or activities that need to be implemented against specified time frames and milestones for implementation. Various amendments made to *Res. Conf. 10.10* at CoP17 further streamlined NIAP processes, enhancing the level of consultation with the Parties involved in the making of decisions, as well as providing them with guidelines in implementing their NIAPs. The process has resulted in many very positive actions taken by a wide range of players, with the 70th meeting CITES Standing Committee agreeing to China, Kenya, Philippines, Tanzania and Thailand and Uganda exiting the oversight process due to progress made.

Three proposals concerning the African Elephant have been proposed for consideration at CoP18. Proposal 10, submitted by Zambia, seeks to transfer its population from Appendix I to Appendix II, subject to a number of conditions. Proposal 11 from Botswana, Namibia and Zimbabwe, seeks amendments to Annotation 2 that would remove references to the conditions that were imposed for the earlier one-off sale that took place following CoP12, allowing for normalised trade in ivory from all four Appendix II-listed African Elephant populations. Proposal 12, submitted by ten Parties, is to transfer from Appendix II to Appendix I the African Elephant populations of Botswana, Namibia, South Africa and Zimbabwe.

Transfer of the population of African Elephant *Loxodonta africana* in Zambia from Appendix I to Appendix II

Proponent: Zambia

Summary: This proposal, which only applies to the African Elephant *Loxodonta africana* population of Zambia, is to transfer that population from Appendix I to Appendix II subject to:

- Trade in registered raw ivory (tusks and pieces) for commercial purposes only to CITES approved trading partners who will not re-export;
- Trade in hunting trophies for non-commercial purposes;
- Trade in hides and leather goods;
- All other specimens shall be deemed to be specimens of species in Appendix I and the trade in them shall be regulated accordingly.

Zambia submitted proposals to transfer its population of *Loxodonta africana* to Appendix II at CoP12 in 2002 and at CoP15 in 2010, both of which were rejected. For the most recent proposal submitted at CoP15 a panel of experts was convened in conformity with *Res. Conf. 10.9 Consideration of proposals for the transfer of African elephant populations from Appendix I to Appendix II.* The Panel of Experts made a generally favourable response having visited Zambia and reviewed the status and management of its elephant populations and Zambia's ability to control trade in ivory. No Panel of Experts has been convened to assess these factors in detail this time and we have been constrained to the assessment of the information contained within the proposal and its Supporting Statement (SS). We present here an assessment of this information against *Res. Conf. 9.24 (Rev. CoP17)*. However, the proponent may be able to provide further detail on factors relating to control of ivory not included within the SS, which would help Parties in their consideration of this proposal.

The SS states that the number of *Loxodonta africana* in Zambia declined significantly due to poaching in the 1970s and 1980s with estimated populations declining from 200,000 in 1972 to ca. 18,000 by 1989. The most comprehensive and reliable information on distribution and population of the species is contained in the African Elephant Database (AED), maintained by the IUCN SSC African Elephant Specialist Group, and presented in the African Elephant Status Reports (AESR), the latest of which was published in 2016. The 2016 report estimates a range of approximately 170,000 km^2 for Zambia and a total population estimate of ca. 22,000. Data for Zambia's population of elephants from the African Elephant Database are:

> 2002 – 12,457 definite, 6,961 probable, 7,631 possible and 235 speculative;
> 2006 – 16,562 definite, 5,948 probable, 5,908 possible and 813 speculative;
> 2015 – 21,967 ± 4,703 (based on systematic survey data). There may be an additional 214 to 314 in areas not systematically surveyed.

CoP18 Doc. 69.2 (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 2017. It reports on the proportion of illegally killed elephants (PIKE) at more than 60 sites in 30 countries in Africa and 28 sites in 13 countries in Asia. A PIKE level of 0.5 has been used as a threshold above which elephant populations are very likely to be in net decline, although the report suggests that the use of the 0.5 PIKE "threshold" should be treated with some caution. The southern African subregion (Angola, Botswana, Eswatini, Malawi, Mozambique, Namibia, South Africa, Zambia and Zimbabwe) was assessed as having a PIKE level of 0.48 in the most recent assessment, having increased from 0.41 in 2016. 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. In Zambia's only MIKE site—South Luangwa National Park—the PIKE estimate increased from 0.59 in 2016 to 0.66 in 2017 based on 85 and 126 detected carcasses in 2016 and 2017, respectively. In a 2015 aerial survey a carcass ratio of 4.5% was reported for Zambia, suggesting a stable population.

Although the proposal seeks to allow "trade in registered raw ivory (tusks and pieces) for commercial purposes only to CITES approved trading partners who will not re-export", the SS is somewhat ambiguous as to whether it is the proponent's intention to do so or not. If the intention is to export ivory, it is not clear whether this would be from stockpiles already registered, and if so whether only tusks in those stockpiles derived from natural mortality or those from poached elephants as well, and whether the intention is also to harvest new ivory for export. The proponent argues that the proposed annotation is in conformity with the precautionary measures as defined in Annex 4 of the above-mentioned Resolution. However, the SS gives little information, nor does it detail any regulatory or enforcement controls as appropriate measures to ensure it complies with the requirements of the Convention, although the proponents do state that in general "Zambia has demonstrated its capacity to comply with the requirements of CITES both by the implementation of the Convention and by further enacting legislation to domesticate the Convention" and that "Controlled legal trade shall provide the required funding for enforcement and management". Population monitoring measures are described. No details are given for measures on controlling trade in ivory, such as stockpile management and law enforcement measures. A quota system has been in place for trophy hunting, which would presumably continue to be employed.

Analysis: The *Loxodonta africana* population of Zambia is not small, nor does it have a restricted distribution. Although it underwent a marked decline since the 1970s, the population size appears to have been relatively stable in the last decade. This population appears therefore not to meet the biological criteria for inclusion in Appendix I. There is a lack of clarity over the intention of the proposal with regard to trade in ivory. Little detail is given on proposed management to ensure that Article IV requirements would be met or of any appropriate enforcement controls in place. On the basis of the information provided it is not possible to determine that the precautionary safeguards are met.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Range

This proposal relates to the population of Zambia.

IUCN Global Category

Vulnerable A2a (2008) ver 3.1

Biological criteria for inclusion in Appendix I

A) Small wild population

The African Elephant Status Report 2016 concluded the estimated population in Zambia is 21,967 ± 4,703 in 2015 (Thouless et al., 2016), noting that there may be an additional 214 to 314 elephants in areas not systematically surveyed. These guesses likely represented a minimum number, and actual numbers could have been higher than those reported. Together, this estimate and guess applied to 105,461 km^2, which was 62% of the estimated known range and possible elephant range. There remains an additional 38% of the estimated range for which no elephant population estimates were available. According to the SS, 13,898 *Loxodonta africana* were estimated in the Luangwa Valley, followed by 6,688 individuals in the Kafue, 1,125 in the Lower Zambezi and 48 estimated in the Sioma-Ngwezi system.

B) Restricted area of distribution

The SS states that *Loxodonta africana* is distributed in seven subregions in Zambia which exceeds 170,000 km^2 in total; this includes Luangwa valley, Mid Zambezi Valley, Kafue areas, Sioma Ngwezi and West Zambezi, and Bangweulu and Nsumbu.

C) Decline in number of wild individuals

In Zambia, Loxodonta africana numbers, range and connectivity have been much reduced over the past 40 years (CITES, 2010). The number of L. africana populations declined significantly due to poaching in the 1970s and 1980s, with Zambia's estimated population declining from 200,000 in 1972 to ca. 18,000 by 1989 (Chomba et al., 2012; Ministry of Tourism, 2003). In recent years, its elephant population appears to have been relatively stable. The main changes have been a reduction in "guesses" from isolated small populations such as West Lunga and Bangweulu. There has been a substantial reduction in numbers in Sioma Ngwezi National Park but this makes up a small proportion (2%) of the national population (Thouless et al., 2016).

The carcass ratio for Kafue National Park was only 1% in 2008 and increased to 7% by 2015 (DNPW, 2016) giving some cause for concern, despite the apparent stability of the population. In Sioma Ngwezi National Park the carcass ratio increased dramatically from 3% to 85% between 2008 and 2015 (DNPW, 2016) indicating either high levels of poaching and/or a seasonally varying population. The Great Elephant Census in 2014 reported carcass ratios of 4.5% for all carcasses and 0.1% for fresh carcasses (Chase et al., 2016). These values are within the sustainable limits and may indicate the population is stable.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Loxodonta africana has not been exploited for commercial trade or domestic consumption in Zambia since 1989 when the species was listed in CITES Appendix I.

Trophy hunting of *Loxodonta africana* has been legal in Zambia since 2005. Zambia increased its annual export quota of hunting trophies from 40 (tusks and other hunting trophies from 20 animals) to 160 (tusks and other trophies from 80 animals) in 2011 (CITES, 2019) after the USA allowed imports of tusks and other trophies for non-commercial purposes from Zambia in 2010. *Zambia declared a temporary suspension of elephant hunting in 2013 (Thouless et al., 2016)*, *and the quota for tusks and other trophies was not reported to CITES in 2013 and 2014*. The *annual export quota was 160 (tusks and other trophies from 80 animals) from 2015 to 2017 but those for 2018 and 2019 are not available on the CITES website (accessed on 9th January 2019).* The SS suggests the quota level is still conservative and below the standing population guideline; *it is also within the limit of hunting for trophies suggested by the Panel of Experts in 2010 (120 animals with close monitoring of trophy quality) (CITES, 2010)*.

According to the CITES Trade Database, Zambia exported 31 trophies and 166 tusks for non-commercial purposes between 2005 and 2016 (Zambia had not submitted annual reports for 2013, 2017 and 2018 as of 10th January 2019). Based on importers' reports, these countries imported 72 trophies and 135 tusks between 2005 and 2017. These data suggest that the EU and the USA are the main importers of tusks and trophies from Zambia. According to exporter's reports, these were exported mainly to Denmark, Spain, France especially during 2006–2012 whereas importers' reports also show Cyprus to be a key importer in the EU. The EU's Scientific Review Group formed a positive opinion on hunting trophies of Loxodonta africana from Zambia at its 73rd meeting in September 2015 after Zambia provided necessary information (SRG, 2015).

According to the SS, Zambia reported 157 seizures to the Elephant Trade Information System (ETIS) between June 2015 and July 2018; there were 37 seizures of raw ivory weighing a total of ca. 730 kg in 2017. *ETIS data report there being 161 seizures reported by Zambia between 2014–2018, totalling ca. 3,773 kg (T. Milliken, in litt., 2019).*

Precautionary Measures

Species likely to be in demand for trade, but its management is such that the CoP will be satisfied with:

A) Implementation by the range States of the requirements of the Convention, in particular Article IV; and

B) appropriate enforcement controls and compliance with the requirements of the Convention

The transfer of Zambia's population to Appendix II would be subject to:

- Trade in registered raw ivory (tusks and pieces) for commercial purposes only to CITES approved trading partners who will not re-export;
- Trade in hunting trophies for non-commercial purposes;
- Trade in hides and leather goods;
- All other specimens shall be deemed to be specimens of species in Appendix I and the trade in them shall be regulated accordingly.

The SS provides information on Zambia's general implementation of CITES, however it does not offer details on how international trade of Loxodonta africana specimens would be monitored and controlled, and therefore it is uncertain that this can be interpreted as a special measure.

The proposal and SS are not consistent: while the proposed annotation includes trade in raw ivory (tusks and pieces) for commercial purposes, the SS noted that Zambia proposes to down-list its population of Loxodonta africana to allow for sustainable use of the species through trophy hunting for non-commercial purposes, commercial trade in hides and leather goods, with no mention of ivory. It is therefore unclear from the SS whether Zambia intends to resume ivory trade for commercial purposes, let alone how any such trade will be carried out and controlled.

The SS reports that there are estimated to be ca. 55,000 kg of ivory held in the strong room as of 31st December 2017. *However, the SS makes no reference to the volume of ivory it plans to trade.*

Zambia seeks to collect hides from elephants killed in protection of property or in other management actions, the proposal does not explain the number of elephants killed for this reason or the amounts that might be traded. There is no indication of the volume of trade in hides and leather goods or how this trade would be managed. The SS refers to elephant trophy hunting being based on a quota that accommodates problem animals. It is not clear if the trade in elephant specimens would fall under the same quota system and therefore would be regulated under that system.

In the previous proposal submitted at CoP15, precautionary measures were detailed in relation to the specific elements of proposed trade in registered raw ivory including the volume, the type of sale and the origin of the ivory to be traded. Likewise, information was presented on the potential trading partners, with the registered government-owned stocks being verified by the Secretariat and the proceeds of the trade being used exclusively for elephant and community conservation, and development programmes within or adjacent to the elephant range in Zambia. Processes would have been put in place in the event of non-compliance. No such measures are outlined in this proposal.

Export quota or other special measure
No export quota is provided in the SS.

Additional Information

Threats

According to the SS, the major threats are human–elephant conflicts and poaching. *When Zambia's proposal to CoP15 was evaluated by the Panel of Experts convened under Res. Conf. 10.9 (see CoP15, Doc. 68, Annex 6 b), it was concluded that both legal and illegal offtake appeared sustainable except for in the Lower Zambezi.*

Additionally, the MIKE programme data for South Luangwa National Park in Zambia show the proportion of illegally killed elephants (PIKE) estimate, which is calculated as the number of illegally killed elephants found divided by the total number of elephant carcasses encountered by patrols and other means, increased from 0.59 in 2016 to 0.66 in 2017 (CITES, 2018), suggesting the ratio of the elephant deaths attributed to illegal killing was higher than those attributed to natural causes. However, it must be noted that this level is not representative of the country, only the site (ca. 15% of total national population).

Conservation, management and legislation

The Zambia Wildlife Act No. 14 of 2015 places responsibility for wildlife management under the National Parks and Wildlife Department. Penalties include 5–20 years of imprisonment for illegal hunting of elephants (first offender) and 7–20 years of imprisonment for illegal hunting and trade or trafficking (first offender). *The act also stipulates that the hunting revenues (concession and hunting fees) should be divided equally between the Zambia Wildlife Authority (ZAWA) and the community concerned. Fees for elephant hunts are subject to a separate Statutory Instrument which stipulates payment of USD 10,000 per elephant (CITES, 2010).*

International trade in wildlife is regulated by the Statutory Instrument for International Trade in Endangered Species of Wild Fauna and Flora No. 61 of 2007. Zambia also published a national policy and action plan for Elephant management in 2003. According to the supporting statement, the Zambia National Strategy for Elephant Management and Conservation is currently under review.

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Amend the existing annotation for the populations of African Elephant *Loxodonta africana* in Botswana, Namibia, South Africa and Zimbabwe

Proponents: Botswana, Namibia and Zimbabwe

Summary: The African Elephant *Loxodonta africana* populations of Botswana, Namibia and Zimbabwe were transferred from Appendix I 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 2. The annotation allows for trade in various non-ivory specimens and products of *L. africana* under a range of conditions, somewhat different for each of the four range States in question. Regarding 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 conditions. One of these conditions was 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 which occurred in 2008, during which time a decision-making mechanism for a process of trade in ivory would be developed. There is currently no agreed decision-making mechanism for allowing trade in ivory under the auspices of the Conference of the Parties.

The proposal is to amend the existing annotation for the Appendix II populations of *Loxodonta africana* in Botswana, Namibia, South Africa and Zimbabwe, on the grounds that the proponents believe some elements of the current annotation "are no longer relevant or not appropriate."

The amendments proposed are as follows:

"For the exclusive purpose of allowing:

a. trade in hunting trophies for non-commercial purposes
b. trade in live animals to appropriate and acceptable destinations, as defined in Resolution Conf. 11.20 (Rev. CoP17), for Botswana and Zimbabwe and for in situ conservation programmes for Namibia and South Africa;
c. trade in hides;
d. trade in hair;
e. trade in leather goods for commercial or non-commercial purposes for Botswana, Namibia and South Africa and for non-commercial purposes for Zimbabwe;
f. 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;
g. trade in registered raw ivory (for Botswana, Namibia, South Africa and Zimbabwe, whole tusks and pieces) subject to the following:
    i. only registered government-owned stocks, originating in the State (excluding seized ivory and ivory of unknown origin);
    ii. only to trading partners that have been verified by the Secretariat, in consultation with the Standing Committee, to have sufficient national legislation and domestic trade controls to ensure that the imported ivory will not be re-exported and will be managed in accordance with all requirements of *Resolution Conf. 10.10 (Rev. CoP17*) concerning domestic manufacturing and trade;
    iii. not before the Secretariat has verified the prospective importing countries and the registered government-owned stocks;
    iv. ~~raw ivory pursuant to the conditional sale of registered government-owned ivory stocks agreed at CoP12, which are 20,000 kg (Botswana), 10,000 kg (Namibia) and 30,000 kg (South Africa);~~
    v. ~~in addition to the quantities agreed at CoP12, government-owned ivory from Botswana, Namibia, South Africa and Zimbabwe registered by 31 January 2007 and~~

~~verified by the Secretariat may be traded and despatched, with the ivory in paragraph (g) iv) above, in a single sale per destination under strict supervision of the Secretariat;~~

vi. the proceeds of the trade are used exclusively for elephant conservation and community conservation and development programmes within or adjacent to the elephant range; and

vii. ~~the additional quantities specified in paragraph g) v) above shall be traded only after the Standing Committee has agreed that the above conditions have been met; and~~

h. ~~no further proposals to allow trade in elephant ivory from populations already in Appendix II shall be submitted to the Conference of the Parties for the period from CoP14 and ending nine years from the date of the single sale of ivory that is to take place in accordance with provisions in paragraphs g) i), g) ii), g) iii), g) vi) and g) vii). In addition such further proposals shall be dealt with in accordance with Decisions 16.55 and 14.78 (Rev. CoP16).~~

On a proposal from the Secretariat, the Standing Committee can decide to cause this trade to cease partially or completely in the event of non-compliance by exporting or importing countries, or in the case of proven detrimental impacts of the trade on other elephant populations.

All other specimens shall be deemed to be specimens of species included in Appendix I and the trade in them shall be regulated accordingly."

If accepted, the proposal's main effect would be to allow exports of registered raw ivory. Although trading partners would need to be verified by the Secretariat, in consultation with the Standing Committee, no formal and specific mechanisms are proposed to oversee any trade, except that the Standing Committee (based on a proposal from the Secretariat) would be able to decide to cause this trade to cease partially or completely in the event of non-compliance by exporting or importing countries, or in the case of proven detrimental impacts of the trade on other elephant populations. The Parties therefore need to be satisfied that the Precautionary Measures in *Res. Conf. 9.24 (Rev. CoP17)* Annex 4 are met.

The proponents state that "Robust control measures are already in place within the legal framework of the proponents, at national level. The comprehensive commitments under various SADC regional initiatives and agreements ensure accountability and safeguards for compliance". Legal instruments are noted. The SS states that elephant populations are managed according to elephant management plans and strategies at national level and spatially-explicit management plans that are responsive to local dynamics. Zimbabwe is one such country with an up-to-date elephant management plan. However, for all the countries, details of the precautionary measures are lacking in the Supporting Statement (SS).

The only safeguards for any future exports of raw ivory would be the basic requirements of Article IV of the Convention for trade in Appendix II species (i.e. non-detriment findings and legal acquisition findings). The SS does not provide details as to how the proposed trade would be assessed for sustainability and controlled.

This proposal applies only to the *Loxodonta africana* population of four contiguous southern African countries: Botswana, Namibia, South Africa and Zimbabwe. The most comprehensive and reliable information on distribution and population of the species is contained in the African Elephant Database (AED), maintained by the IUCN SSC African Elephant Specialist Group, and presented in the African Elephant Status Reports (AESR) the latest of which was published in 2016. The 2016 AESR estimates a combined range in the four countries considered here as approximately 500,000 km^2 and a total population estimate of at least 255,000. This amounts to ca. 50–60% of the species as a whole (415,428 ± 20,112 with possibly an additional 117,128 to 135,385 in areas not systematically surveyed). A detailed breakdown of these figures is as follows:

Botswana: 2002 – 100,629 definite, 21,237 probable and 21,237 possible;
2006 – 133,829 definite, 20,829 probable and 20,829 possible;
2015 – 131,626 ± 12,508 (based on systematic survey data);

Namibia: 2002 – 7,769 definite, 1,872 probable and 1,872 possible;
2006 – 12,531 definite, 3,276 probable and 3,296 possible;
2015 – 22,754 ± 4,305 (based on systematic survey data). There may be an additional 90 in areas not systematically surveyed;

South Africa: 2002 – 14,071 definite and 855 possible;
2006 – 17,847 definite, 638 possible and 22 speculative;
2015 – 18,841 (based on systematic survey data). There may be an additional 8,425 to 8,435 in areas not systematically surveyed;

Zimbabwe: 2002 – 81,555 definite, 7,039 probable, 7,373 possible;
2006 – 84,416 definite, 7,033 probable, 7,367 possible and 291 speculative;
2015 – 82,630 ± 8,589 (based on systematic survey data). There may be an additional 1,635 to 1,805 in areas not systematically surveyed;

Further discussion of the populations of *Loxodonta africana* in Botswana, Namibia, South Africa and Zimbabwe can be found in the IUCN/TRAFFIC Analyses for CoP18 Prop. 12.

CoP18 Doc. 69.2 (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 2017. It reports on the proportion of illegally killed elephants (PIKE) at more than 60 sites in 30 countries in Africa and 28 sites in 13 countries in Asia. A PIKE level of 0.5 or lower is generally considered sustainable, although the report suggests that the use of the 0.5 PIKE “threshold” should be treated with some caution. The southern African sub-region (Angola, Botswana, Malawi, Mozambique, Namibia, South Africa, Swaziland, Zambia and Zimbabwe) was assessed as having a PIKE level of 0.48 in the most recent assessment, having increased from 0.41 in 2016. This is the second highest level ever recorded in this subregion. 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.

The ETIS analysis of illegal ivory trade for CoP18 has identified Zimbabwe as a Category C country for the first time; a party affected by the illegal trade in ivory. South Africa has also been reported as a country with an involvement in illegal ivory trade for several years. Namibia exhibits mid-range variables in terms of the mean number of seizures and the mean weight value, while for Botswana, frequency and scale measures point to a rather small number of mostly medium weight seizures, with no involvement in the large-scale ivory movements.

Analysis: The *Loxodonta africana* population of Botswana, Namibia, South Africa and Zimbabwe is not small, nor does it have a restricted distribution or undergoing a marked decline. Therefore, this population does not meet the biological criteria for inclusion in Appendix I (See Analyses for CoP18 Prop. 12). There are no explicit guidelines in *Res. Conf. 9.24 (Rev. CoP17)* as to how to deal with a proposal to amend an annotation for an Appendix II-listed species. However, these proposed amendments can be interpreted as special measures under the terms of the precautionary measures in Annex 4 of *Res. Conf. 9.24 (Rev. CoP17)*. Adoption of the proposed changes would remove some which are no longer valid, with timeframes having passed and decisions no longer in effect. However, if accepted, the main effect would be to allow exports of registered raw ivory but without the oversight of previous mechanisms by the Standing Committee and the Conference of the Parties. Parties would need to be satisfied that Botswana, Namibia, South Africa and Zimbabwe are implementing the requirements of the Convention, particularly Article IV, and that the appropriate enforcement controls and compliance with the requirements of the Convention are in place. Insufficient detail of such measures is provided in the SS to determine whether or not this would be the case.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Range
The proposal relates to the populations of Botswana, Namibia, South Africa and Zimbabwe

IUCN Global Category
Vulnerable A2a (2008) ver 3.1

Biological criteria for inclusion in Appendix I
See IUCN/TRAFFIC Analyses for CoP18 Prop. 12.

Trade criteria for inclusion in Appendix I
The species is or may be affected by trade
Loxodonta africana is subject to both domestic and international trade. Within Africa, derivatives including skin and hair are made into a variety of products: elephant meat is consumed in parts of Western, Central and Southern Africa; elephants are sport-hunted; and live elephants are sometimes caught and traded. While Botswana has no legal domestic ivory market, legislation in Namibia, South Africa and Zimbabwe allows domestic sales of ivory subject to permit. *The SS reports that* robust monitoring systems controlled by permits and licences for domestic trade are in place, regular inspections are also done to check on compliance to set standards and security arrangements. Botswana is the only one of the four countries that currently has a suspension on trophy hunting, including that of elephants, *although this is currently under review*. *Namibia, South Africa and Zimbabwe set annual export quotas for tusks and elephant trophies whereas Botswana placed a ban on trophy hunting in 2014 and a zero-export quota for exports of hunting trophies has been set since 2015 (CITES, 2019b; Thouless* et al., *2016).*

Under the Appendix II listing, Botswana, Namibia and Zimbabwe were permitted two "one-off" sales of registered raw ivory from the government-owned stocks; the first to Japan in 1999 *(*ca. 50 t*)*, and the second to Japan and China in 2008 (*ca*. 102 t) (South Africa was also permitted to sell its ivory in this sale). This resulted in a nine-year moratorium until 2017 specified in the current annotation, in accordance with the agreement *reached at CoP14. According to the CITES Trade Database, reported exports in* Loxodonta africana *directly from these four range States over the last five years has principally involved wild-sourced hunting trophies and tusks (Table 1). All trade reported by weight was by Zimbabwe whose exports include* ca. *29,000 kg of tusks and almost 11,500 kg of ivory carvings.*

***Table 1**. Exporter reported quantities from Botswana, Namibia, South Africa and Zimbabwe by weight (kg) and "blank" in the selected products terms, between 2013 and 2017 (equivalent quantities were not reported by importers) (Source: CITES Trade Database).*

Item	*Botswana*	*Namibia*	*South Africa*	*Zimbabwe*
Ivory carvings				
kg				*11,484*
Number of items (Unit not specified)			*1,242*	*2,271*
Tusks				
kg				*28,632**
Number of items (Unit not specified)	*1,031*	*175*	*590*	*90*
Trophies				
kg				
Number of items (Unit not specified)	*37*	*362*	*659*	*317*

**Based on an average tusk weighing 5.5 kg (SC62 Doc. 46.4 Annex, 2012), these total weights of ivory would equate to approximately 2,603 elephants.*

Trade criteria for inclusion in Appendix I

Precautionary Measures
Species likely to be in demand for trade, but its management is such that the CoP will be satisfied with:

A) Implementation by the range States of the requirements of the Convention, in particular Article IV; and
B) appropriate enforcement controls and compliance with the requirements of the Convention
Namibia, South Africa and Zimbabwe set annual export quotas for tusks and elephant trophies whereas Botswana placed a ban on trophy hunting for elephants in 2014 and a zero-export quota for exports of hunting trophies has been set since 2015 (CITES, 2019b, Thouless et al., *2016), although lifting this is currently being considered.*

Although the SS outlines penalties for illegal hunting and/or trade in Botswana, Namibia, South Africa and Zimbabwe and enforcement controls and compliance with CITES, the SS does not provide details as to how the proposed trade will be carried out and controlled beyond indicating that ivory collected through routine conservation management, which has been officially registered to ensure traceability of every ivory piece and secured in government storage facilities, and whose records are monitored by responsible authorities will be disposed to responsible markets. The income generated will be used to fund implementation of national elephant management plans and anti-poaching strategies, as well as supporting community-based initiatives for securing elephant habitat, dispersal areas and movement corridors.

Analysis of the Elephant Trade Information System (ETIS) prepared for CoP18 (CITES, 2019a) and the report submitted to SC69 in 2017 (CITES, 2017) indicates that enforcement controls and compliance may be problematic in some countries. Zimbabwe was identified as one of the most important countries of origin or export of illegal commercial shipments of worked ivory products (10+ kg); this trade is important as it suggests ivory processing in Africa for Asian markets. Zimbabwe's seizure data also point to considerable movements of raw ivory during 2015–2017 as well. Zimbabwe's domestic ivory market is a concern. Although South Africa was not included in the National Ivory Action Plan (NIAP) process after CoP17, the country was identified as a Category A country in the ETIS analysis to CoP16, a Category B country in the ETIS report to CoP17 and a Category C country in the report to CoP18 pursuant to Annex 3 of Res. Conf. 10.10 (Rev. CoP17). These assessments continue to suggest that significant quantities of ivory have entered international trade from South Africa and large-scale movements of ivory suggest a considerable level of criminal activity operating here. It seems that enforcement and controls in these countries are not sufficient at present to ensure that no significant amounts of ivory taken or traded illegally from other countries are traded within or through these countries.

For Botswana, frequency and scale measures point to a rather small number of mostly medium weight seizures, with no involvement in the large-scale ivory movements which are the hallmark of organised crime. Most seizures are made by Botswana showing generally good law enforcement. Namibia exhibits mid-range variables in terms of the mean number of seizures and the mean weight value, but most seizures take place within Namibia rather than further along the trade chain. There are some seizures involving large-scale movements, though a notable degree of organised criminal activity is not apparent (CITES, 2019a).

Independent assessments of stockpile management have not been conducted in recent years (T. Milliken, in litt., *2019).*

The SS reports that robust control measures are already in place within the legal framework of the proponents, at national level. The comprehensive commitments under various Southern African Development Community (SADC) regional initiatives and agreements ensure accountability and safeguards for compliance. *No information is provided in the SS on the implementation of these commitments by the countries in question.*

Export quota or other special measure
No export quota is reported. There is no information provided in the SS on the quantity of ivory currently stockpiled by the proponents.

Additional Information

Threats
See IUCN/TRAFFIC Analyses for CoP18 Prop. 12.

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

Proponents: Burkina Faso, Côte d'Ivoire, Gabon, Kenya, Liberia, Niger, Nigeria, Sudan, Syrian Arab Republic and Togo

Summary: This proposal applies only to the African Elephant *Loxodonta africana* population of four contiguous southern African countries: Botswana, Namibia, South Africa and Zimbabwe. The most comprehensive and reliable information on distribution and population of the species is contained in the African Elephant Database (AED), maintained by the IUCN/SSC African Elephant Specialist Group, and presented in the African Elephant Status Reports (AESR), the latest of which was published in 2016. The 2016 AESR estimates a combined range in the four countries considered here as approximately 500,000 km^2 and a total population estimate of at least 255,000. This amounts to ca. 50–60% of the species as a whole (global population is 415,428 ± 20,112 with possibly an additional 117,128–135,385 in areas not systematically surveyed). A detailed breakdown of these figures is as follows:

Botswana: 2002 – 100,629 definite, 21,237 probable and 21,237 possible;
2006 – 133,829 definite, 20,829 probable and 20,829 possible;
2015 – 131,626 ± 12,508 (based on systematic survey data);

Namibia: 2002 – 7,769 definite, 1,872 probable and 1,872 possible;
2006 – 12,531 definite, 3,276 probable and 3,296 possible;
2015 – 22,754 ± 4,305 (based on systematic survey data). There may be an additional 90 in areas not systematically surveyed;

South Africa: 2002 – 14,071 definite and 855 possible;
2006 – 17,847 definite, 638 possible and 22 speculative;
2015 – 18,841 (based on systematic survey data). There may be an additional 8,425 to 8,435 in areas not systematically surveyed;

Zimbabwe: 2002 – 81,555 definite, 7,039 probable, 7,373 possible;
2006 – 84,416 definite, 7,033 probable, 7,367 possible and 291 speculative;
2015 – 82,630 ± 8,589 (based on systematic survey data). There may be an additional 1,635 to 1,805 in areas not systematically surveyed.

CoP18 Doc. 69.2 (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 2017. It reports on the proportion of illegally killed elephants (PIKE) at more than 60 sites in 30 countries in Africa and 28 sites in 13 countries in Asia. A PIKE level of 0.5 or lower is generally considered sustainable, although the report suggests that the use of the 0.5 PIKE “threshold” should be treated with some caution. The southern African subregion (Angola, Botswana, Eswatini, Malawi, Mozambique, Namibia, South Africa, Zambia and Zimbabwe) was assessed as having a PIKE level of 0.48 in the most recent assessment, having increased from 0.41 in 2016. This is the second highest level ever recorded in this subregion. 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.

The Supporting Statement (SS) of the proposal deals extensively with the wider *Loxodonta africana* 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 CoP18 Doc. 69.2), 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 CoP18 Doc. 69.3). The proponents argue that transferring the Appendix-II listed *L. africana* 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: The *Loxodonta africana* population of Botswana, Namibia, South Africa and Zimbabwe is not small, nor does it have a restricted range and it is not undergoing a marked decline. Therefore, this population does not meet the biological criteria for inclusion in Appendix I.

Regarding the potential impact of this proposed listing amendment on elephant populations elsewhere, there is no provision to address this question in any guidelines or criteria under the Convention. There is a wide and divergent range of views on the subject, as can be seen in the SS of the current proposal and of proposals CoP18 Prop. 10, submitted by Zambia, and CoP18 Prop. 11 submitted by Botswana, Namibia and Zimbabwe.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS

Range

This proposal relates to the population of Botswana, Namibia, South Africa and Zimbabwe

IUCN Global Category

Vulnerable A2a (2008) ver 3.1

Biological criteria for inclusion in Appendix I

A) Small wild population

According to the latest African Elephant Status Report (AESR) released in 2016, the 2015 total continental population based on "estimates" was 415,428 ± 20,111. *There may be an additional 117,127 to 135,384* Loxodonta africana *in areas not systematically surveyed. Together, these figures apply to an area of 1,932,732 km², which is 62% of the estimated known and possible elephant range. There remains an additional 38% of range for which no elephant population estimates are available, although it is likely that average population densities across this range are much lower than in the surveyed areas (Thouless* et al., *2016).*

Holding over 70% of the estimated elephants in Africa, Southern Africa (Angola, Botswana, Eswatini, Malawi, Mozambique, Namibia, South Africa, Zambia and Zimbabwe) has by far the largest population, with an estimated 293,447 ± 16,682 Loxodonta africana*, with nearly 75% of the population occurring in the Kavango Zambezi Transfrontier Conservation Area (KAZA TFCA) (Thouless* et al., *2016).*

The four countries with Appendix-II listed populations had in 2015 a total of 255,851 *Loxodonta africana* (see table 1) and Botswana remains the stronghold of the species.

Table 1. *Populations of* Loxodonta africana *in 2002, 2006 and 2015 in Botswana, Namibia, South Africa and Zimbabwe.*

Country	*Population level 2002*	*Population level 2006*	*Population level 2015*	*Population status*
Botswana	*100,629 definite; 21,237 probable; 21,237 possible.*	*133,829 definite; 20,829 probable; 20,829 possible.*	*131,626 ± 12,508*	*Decreased/Stable*
Namibia	*7,769 definite; 1,872 probable; 1,872 possible.*	*12,531 definite; 3,276 probable; 3,296 possible.*	*22,754 ± 4,305 90 min. guess 90 max. guess*	*Increased*
South Africa	*14,071 definite; 855 possible.*	*17,847 definite; 638 possible; 22 speculative.*	*18,841 8,425 min. guess 8,435 max. guess*	*Increased*
Zimbabwe	*81,555 definite; 7,039 probable; 7,373 possible.*	*84,416 definite; 7,033 probable; 7,367 possible; 291 speculative*	*82,630 ± 8,589 1,635 min. guess 1,805 max. guess*	*Decreased*
Source	*Blanc* et al., *2003*	*Blanc* et al., *2007*	*Thouless* et al., *2016*	

B) Restricted area of distribution
The total range area for the species as a whole (defined as "Known" and "Possible") across Africa was approximately 3.1 million km^2 in 2015.

The range of the species within Botswana, Namibia, South Africa and Zimbabwe covers approximately 504,000 km^2.

C) Decline in number of wild individuals
Botswana
The AESR 2016 gave an estimate of 131,626 ± 12,508 (95% confidence limits), based largely on the Great Elephant Census of 2014. Aerial sample count surveys in 2014 produced an estimate of 129,939 ± 12,501 while equivalent surveys in 2010 gave 128,340 ± 9,938. These differ considerably from estimates made by the Botswana Department of Wildlife and National Parks (154,658 ± 21,253 in 2006 and 207,545 ± 21,771 in 2012, the former being the figure used in the 2007 AESR). However, these differences are believed to result from differences in survey techniques or calculations rather than real population change or elephant movements across national boundaries. There is little evidence from other indicators of major changes in the Botswana elephant population from excess natural mortality, serious poaching or emigration. The 2014 survey report gave a carcass ratio of 7% for northern Botswana, which is not considered exceptionally high, as carcass ratios of up to 8% are considered typical of a stable or increasing elephant population. However, it showed a notable increase over the earlier carcass ratio of 2% in 2010 and 2012.

Namibia
The AESR 2016 noted that the estimated number of elephants in areas surveyed in the last ten years was 22,754 ± 4,305 at the time of the last survey for each area. There may be an additional 90 elephants in areas not systematically surveyed. These guesses were likely to represent a minimum number, with actual numbers possibly higher than those reported. Together, this estimate and guess applied to 84,283 km^2, which was just over half (52%) of the estimated known and possible elephant range. No population estimates were available for the remainder. The report noted that the elephant population in Namibia had continued to increase, although with wide confidence limits in aerial surveys and elephants moving across international borders, it was not possible to be precise about how great the increase in the national population had been.

South Africa
The AESR 2016 reported that the estimated number of elephants in areas surveyed in the last ten years in South Africa was 18,841 at the time of the last survey for each area. It noted that there may be an additional 8,425 to 8,435 elephants in areas not systematically surveyed. These guesses were likely to represent a minimum number, with actual numbers possibly higher than those reported. Together, this estimate and guess applied to 28,203 km^2, which is 92% of the known and possible elephant range. No population estimates were available for the remaining 8%.

There had been a reported increase of about 1,000 elephants in the form of estimates, and about 8,000 in guesses since the AESR 2007. While the major population in Kruger National Park had increased substantially, the quality of information for other areas was lower in the AESR 2016 than in the AESR 2007, and many areas that previously had documented surveys now had unsupported population figures, meaning that elephant numbers were recorded as guesses.

Zimbabwe
The AESR 2016 reported that the estimated number of elephants in areas surveyed in the last ten years in Zimbabwe was 82,630 ± 8,589 at the time of the last survey for each area. It noted there may be an additional 1,635 to 1,805 elephants in areas not systematically surveyed. These guesses likely represented a minimum number, and actual numbers could be higher than those reported. Together, this estimate and guess apply to 78,839 km^2, which is 97% of the estimated known and possible elephant range. No population estimates were available for the remaining 3%.

There had been a decline of just over 10,000 elephants from surveyed populations since the AESR 2007 and an increase of about 1,000 in guesses, mostly from previously unsurveyed areas in North West Matabeleland. Although there had been large losses from the Sebungwe and Lower Zambezi populations, these have been partially compensated by increases in Gonarezhou and North West Matabeland.

Trade criteria for inclusion in Appendix I
The species is or may be affected by trade
Loxodonta africana is subject to both domestic and international trade. Within Africa, derivatives including skin and hair are made into a variety of products: elephant meat is consumed in parts of Western, Central and

Southern Africa; elephants are trophy hunted; and live elephants are sometimes caught and traded. While Botswana has no legal domestic ivory market, legislation in Namibia, South Africa and Zimbabwe allows domestic sales of ivory subject to permit. Botswana is the only one of the four countries that currently has a suspension on trophy hunting, including that of elephants, *although this is currently under review*. *Namibia, South Africa and Zimbabwe set annual export quotas for tusks and elephant trophies whereas Botswana placed a suspension on trophy hunting in 2014 and a zero-export quota for exports of hunting trophies has been set since 2015 (CITES, 2019c; Thouless* et al., *2016).*

Under their Appendix II status, Botswana, Namibia and Zimbabwe were permitted two "one-off" sales of registered raw ivory from the government-owned stocks; the first to Japan in 1999 (*ca. 50 t*), and the second to Japan and China in 2008 *(ca. 102 t)* (South Africa was also permitted to sell its ivory in this sale). This resulted in a nine-year moratorium until 2017, in accordance to the agreement reached at CoP14. According to the CITES Trade Database, reported legal trade in *Loxodonta africana* directly from African range States between 2015 and 2016 principally involved wild-sourced hunting trophies including tusks; direct export of 12,543 kg and 133 wild-sourced tusks were reported by African range States. Countries of import recorded the import of 752 tusks and 124 kg of tusks. *All trade in tusks by weight was from Zimbabwe and was primarily reported as hunting trophies. Trade in tusks by weight declined by 36% between 2015 and 2016 and remained lower than the levels reported in 2014 (8,206 kg). Trade in tusks reported by number increased three-fold between 2015 and 2016 according to data reported by range States, while the number of tusks reported by importers decreased by 56%. Direct trade in wild-sourced ivory carvings reported by African range States was notably lower than in 2014 (38 kg and 89 items reported by exporters 2015–2016 compared to 7,889 kg of ivory carvings reported in 2014) (CITES, 2018a).*

Additional Information

Threats

Across the continent, habitat loss and fragmentation through conversion of forests, savanna and corridors to plantations, subsistence agriculture and settlement, is the most significant long-term threat to elephant populations. Continual declines of the species' range have occurred since 1998; for example, between 2007 and 2015, a 6% decrease in approximate total range was reported, from approximately 3.3 million km^2 to 3.1 million km^2, respectively. Other underlying drivers of population declines include human–elephant conflict and the impacts of climate change.

The most immediate threat to *Loxodonta africana* populations is poaching for ivory and illegal trade, and the recent decline in population is attributed to this threat. Data from the CITES programme for Monitoring the Illegal Killing of Elephants (MIKE) indicate that poaching levels peaked in 2011, since the programme began in 2002, with a moderately declining trend thereafter. *According to the latest figures from the MIKE programme, the Proportion of Illegally Killed Elephants (PIKE) level shows a slight decline in 2017 over 2016 but remains a concern with a value above 0.5, suggesting that more of the elephant deaths reported are attributed to illegal killing than to natural causes (CITES, 2018a). The sub-regional PIKE estimates for 2017 compared with 2016 are as follows (CITES, 2018a; CITES 2019a):*

- *Eastern Africa – decline from 0.32 in 2016, to 0.22 in 2017;*
- *Southern Africa – increase from 0.41 in 2016, to 0.48 in 2017;*
- *Central Africa – concerningly high levels, with a value of 0.76 over the last three years; and*
- *West Africa – due to low sample sizes, reliable inferences based on the year-on-year trend for West Africa was challenging, however for sites where reporting has been conducted, PIKE estimates were extremely high.*

While overall, poaching has not had the same impact in Southern Africa as in other regions, it has severely affected populations in Zimbabwe, Angola, Mozambique, and to a lesser extent, Zambia (Thouless et al., *2016)*. Notable increases in PIKE were recorded in Kruger National Park (South Africa), with an increase of 44% between 2016 (0.27) and 2017 (0.39). PIKE also increased in Chobe National Park, Botswana from 0.00 to 0.21 between 2016 and 2017. *Two sites in Zimbabwe (Chewore and Nyami Nyami) reported PIKE levels, with Chewore reporting 0.21 and 0.10 and Nyami Nyami reporting 0.27 and 0.21, in 2016 and 2017 respectively. In Namibia, PIKE levels are reported from Etosha and Zambezi, with Etosha reporting 0.00 and 0.07 and Zambezi reporting 0.33 and 0.28, in 2016 and 2017 respectively. In Zimbabwe, poaching of elephants has increased in the last decade, particularly in the north of the country (Thouless* et al., *2016). The emergence of poisoning as a poaching technique has also been reported, with over 100 elephants reportedly killed in a single incident in Hwange National Park in late 2013 (Muboko* et al., *2014).*

The National Ivory Action Plans (NIAPs) process is a practical tool under the direction of the Standing Committee to address illegal ivory trade by strengthening ivory trade controls, supporting law enforcement and improving

awareness by identifying countries in three categories of concern pursuant with Annex 3 of Resolution Conf. 10.10 (Rev. CoP17). Zimbabwe was identified as a Category C country (parties affected by the illegal trade in ivory) for the first time in the latest ETIS report as a country of origin or export of raw ivory and worked ivory products between 2015 and 2017 (CITES, 2019b). South Africa has also been reported as a country with an involvement in illegal ivory trade for several years (CITES 2013; CITES, 2016; CITES, 2019b), for example, in 2017, Viet Nam seized ca. 2.5 t of ivory from South Africa (CITES, 2019b). After CoP17, however, the Standing Committee agreed that South Africa would no longer be included in the NIAP process (CITES, 2017). Seizure records indicate that small quantities of raw ivory from countries such as Botswana, Malawi, Mozambique and Zimbabwe are sometimes entering South Africa and the country is also used for processing for export, which with increased elephant poaching in the country could make the country a more prominent exit point for illegal ivory to Asia (CITES, 2019b).

ETIS data also suggest increased ivory processing within Africa for the export of products, particularly chopsticks, name seal blocks, bangles, beads and pendants, to Asian markets, with approximately 24 cases from four African countries (Zimbabwe, Angola, Democratic Republic of the Congo and Ethiopia) representing 1.11 t of worked ivory moving from Africa to Asia in 2017 (CITES, 2018a).

Conservation, management and legislation

Conservation management measures vary considerably across the continent. They range from the creation of migration corridors, population monitoring, development of artificial waterholes, regulating legal offtake and enforcement efforts to prevent poaching. *Some community-based conservation programmes in which revenue from the trophy hunting of elephants is fed directly to local communities have proved effective in increasing tolerance to elephants, and thus indirectly in reducing levels of human-elephant conflict (Naidoo* et al., *2006; Blanc, 2008). Transboundary elephant populations are increasingly being co-managed by relevant range States and large-scale conservation and management efforts are in place at the national and regional levels (Blanc, 2008) (Table 2). Although up to 70% of the species' range is believed to lie in unprotected land, most large populations occur within protected areas (Blanc, 2008) (Table 2). For example, Namibia's elephants occur across the northern region of the country, mostly in national parks and community areas, while South Africa's largest population inhabits Kruger National Park, and the majority of the remaining population exists in relatively small fenced areas, many of which are privately owned (Thouless* et al., *2016).*

***Table 2**. Summary of protected range and national management plans for the countries with Appendix-II listed* Loxodonta africana *populations (Thouless* et al., *2016; CITES, 2018c; IAPF, 2018; CITES, 2019a).*

Country	***Range area (% of range protected)***	***National Management Plan***
Botswana	*228,073 km² (16%)*	*The National Policy and Strategy for the Conservation and Management of* Loxodonta africana *was published in 2003, however it is now outdated and in need of revision.*
Namibia	*164,069 km² (18%)*	*Namibia's elephant management plan was published in 2007.*
South Africa	*30,651 km² (93%)*	*The South African Government published a set of Norms and Standards for Managing* Loxodonta africana *in 2008, which were under review in 2014.*
Zimbabwe	*81,228 km² (61%)*	*Zimbabwe published a national elephant management plan covering the period 2015 to 2020.*

The *Loxodonta africana* populations of Botswana, Namibia, South Africa, and Zimbabwe are subject to varying levels of national protection:

- In Botswana, hunting is regulated by the Wildlife Conservation and National Parks Regulations. Hunting of all wildlife, including *L. africana*, has been banned since January 2014;
- In Namibia, *L. africana* is classified as a "Specially Protected" species under the Nature Conservation Ordinance. Under this legislation, hunting, capture, transport, possession and international trade in raw ivory, live animals or other derivatives are subject to permits and conditions;
- *In South Africa, trade in specimens of threatened or protected species including* L. africana *is not permitted without registration as a wildlife trader under the Biodiversity Act (10 of 2004) and the Threatened or Protected Species Regulations (CITES, 2018b).*
- In Zimbabwe, *L. africana* is not designated as a Specially Protected Animal and mandatory custodial penalties only apply to illegal trade in ivory, not to illegal killing of elephants.

Other comments

The range States for these Loxodonta africana *populations do not support the proposal.*

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Inclusion of Woolly Mammoth *Mammuthus primigenius* in Appendix II

Proponent: Israel

Summary: The Woolly Mammoth *Mammuthus primigenius* was the final surviving member of the *Mammuthus* genus, with the last known populations surviving on Wrangel Island, East Siberian Sea (around 3,700 years ago) and St Paul Island, Alaska (around 5,600 years ago). During the last glacial period (around 115,000–12,000 years ago), Woolly Mammoths were at their most widespread and were present across North America, northern Asia and Europe. Woolly Mammoth extinction is thought to have been caused by a reduction in suitable habitat due to temperature increases, combined with an increase in anthropogenic hunting pressure.

The current primary Woolly Mammoth commodity in trade is ivory, which is largely recovered from the permafrost in Siberia, where ivory has not become fossilised. Little is known about the trade in mammoth ivory, but it is thought that the main trade route is from Russia to Hong Kong SAR and then tusks are mostly exported to mainland China for processing. While information on the global trade in mammoth ivory is not available, import and export data from Hong Kong SAR and USA import data are presented below:

Hong Kong SAR customs data (between 2005-2016) report that:

- Hong Kong SAR imports on average 36,000 kg of mammoth ivory (raw tusks and/or unworked tusk pieces) annually, mostly from Russia.
- The majority of mammoth ivory is re-exported (on average 29,000 kg annually) to mainland China.

USA import data (between 1999–2013) report that:

- Average annual mammoth ivory commodity imports to the USA were 1,600 tusks, 800 kg and 120 pieces of tusk/ivory and 40,000 ivory carvings.
- The majority of these imports were from Hong Kong SAR.

Data on the origin of mammoth ivory traded by both Hong Kong SAR and the USA showed that although the vast majority of mammoth ivory traded was listed as originating in Russia, smaller volumes of trade were reported with origins where mammoth ivory is likely to be fossilised: mainly European countries, but small amounts reportedly originated from African Elephant *Loxodonta africana* range States (e.g. Chad, Gabon, Kenya, Mozambique and South Africa) and Asian Elephant *Elephas maximus* range States (e.g. China, Indonesia and Thailand).

The Supporting Statement makes it clear that this proposal is aimed to help regulation of trade in ivory from living elephants by preventing the laundering/mislabelling of ivory from extant elephant species as Woolly Mammoth ivory. Evidence from mainland China, Hong Kong SAR, Myanmar and the USA suggests that some vendors are mislabelling elephant ivory as mammoth ivory, but there is no comprehensive assessment to suggest how widespread this practice is.

The proposal of an extinct species for inclusion in the Appendices is unusual and CITES provisions for this are fairly limited. The Convention text does not preclude the listing of extinct species although *Res. Conf. 9.24 (Rev. CoP17)* states that “extinct species should not normally be proposed for inclusion in the Appendices”. When higher listings are considered, Annex 3 of *Res. Conf. 9.24 (Rev. CoP17)* states that “Parties are encouraged to note any extinct species in the higher taxon and to clarify whether these are included or excluded from the proposed listing”. The proponent goes on to argue that there are instances where the deletion of extinct species from the Appendices is discouraged, such as in Annex 4 Paragraph D of *Res. Conf. 9.24 (Rev. CoP17)*, which gives four situations where extinct species should not be deleted, including if “they resemble extant species included in the Appendices”.

When whole mammoth tusks are traded it is relatively straightforward to tell them apart from elephant tusks, as mammoth tusks display a twist whereas elephant tusks are generally straight. Cross

sections which display Schreger lines can also be used to distinguish mammoth ivory (average Schreger line angle <90°) from elephant ivory (average Schreger line angle >115°). Identification becomes more of an issue for worked mammoth ivory, especially small pieces (carvings, pendants etc.) which may not display Schreger lines and can often be very difficult to tell apart from elephant ivory. Instances of elephant ivory being painted or intentionally discoloured to appear as mammoth ivory have been observed. Fossilised mammoth ivory cannot be carved and therefore is not a substitute for elephant ivory for carvings or other processed items.

There are few legal provisions for regulation of trade in mammoth ivory. Although many countries have laws banning trade in ivory, this is mostly directed at elephant ivory.

Analysis: The Supporting Statement makes it clear that the purpose of the listing is to prevent illegal trade in living elephants by preventing the mislabelling of elephant ivory as mammoth ivory. Anecdotal evidence of elephant ivory being traded as mammoth ivory is found within the literature and surveys, but the scale of these substitutions is unclear and thought to be quite limited.

Some believe that mammoth ivory should be promoted as an alternative to elephant ivory as mammoths are already extinct, whereas others feel there should be a complete trade ban on all ivory including mammoth in order to close the potential for laundering of elephant ivory. The proponent does not take a position on this, clarifying that its intention is simply to improve documentation and regulation of mammoth ivory trade in support of the conservation of extant elephant species.

Res. Conf. 9.24 (Rev. CoP17) states in Annex 3 that “extinct species should not normally be proposed for inclusion in the Appendices”, but this does not definitively preclude their inclusion.

When traded as tusks or large pieces of tusk with a visible cross section, it is fairly straightforward to distinguish between elephant and mammoth ivory. Difficulties in identification occur with worked pieces of ivory, especially when they are small and the Schreger lines are not apparent. Given that USA customs data show high levels of international trade in mammoth ivory carvings, it would appear that the look-alike criteria in Annex 2b of *Res. Conf. 9.24 (Rev. CoP17)* would be met when non-fossilised mammoth ivory is traded in processed form.

Overall, the regulation of international trade in mammoth ivory through an Appendix II listing may help reduce opportunities for misdeclaration and/or laundering of elephant ivory. However, the extent to which this would contribute to a reduction of global illegal elephant ivory trade flows is unknown and likely to be limited. The Parties will need to weigh these potential benefits against the costs of regulation of significant legal mammoth ivory movements.

Other Considerations: *Res. Conf. 11.21 (Rev. CoP17) Use of annotations in Appendices I and II* indicates that only animal species listed in Appendix III can be annotated to specify the parts and derivatives covered by the listing. However, given the proposal to list an extinct species is somewhat unusual, if the Parties decided to list the species in Appendix II, it may be useful to consider restricting the proposal to whole tusks and the specimens of the species in the form in which they are traded that resemble elephant ivory and are hard to distinguish, namely worked ivory, which would help ensure that effective control of trade in elephants is achieved. Fossils and other artefacts including non-commercial scientific exchanges of mammoth parts (such as bones, skin, hair, and DNA) for research and education by museums and universities could be excluded.

Summary of Available Information

Text in non-italics is based on information in the Proposal and /Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Range

The genus *Mammuthus* includes five extinct species in the family Elephantidae. This family also includes all the living relatives of the mammoths: the elephants. The family Elephantidae first emerged in Africa as tropical animals about 55 million years ago (mya); about 10 million years after the last dinosaurs. The earliest members of the genus *Mammuthus* emerged about 5 mya in Africa and mammoths spread to Europe as forest-living

species about 3-4 mya, apparently via the Levant. From there, mammoths spread to northern and eastern Asia and to North America around 1.5 mya. They spread throughout North America into Mexico, and in Asia throughout Siberia, into Mongolia, China, Japan and India. Mammoth did not spread to South America.

The cold-adapted species, the Woolly Mammoth *M. primigenius*, is the source of almost all mammoth ivory in trade today. The species emerged around 0.5 mya in Europe. By the start of the last ice age around 100,000 years ago *M. primigenius* occurred throughout Europe, northern Asia, and most of North America, and eventually having a large distribution covering almost all of Europe from Portugal and Spain in the southwest, all across Central and Eastern Europe, to Mongolia, northern China, South Korea and Japan up to north-eastern Siberia, and including the American mid-west, and eastern Canada. Remains have also been found from the shelf regions of the Arctic Ocean and north-western Europe to the bottom of the Adriatic Sea and to the mountains of Crimea, Ukraine.

Most populations of Woolly Mammoths went extinct after the last ice age ended, around 10 to 40 thousand years ago, yet remnant populations were still living until about 5,600 years ago on St. Paul Island in Alaska and even more recently, until about 3,700 years ago (around the year 1,650 BC), on Wrangel Island in the East Siberian Sea.

Woolly Mammoths were shown to have been at their most widespread during the most recent glacial period (Pelistocene) with a range of around 33,000,000 km ^{2} including in Europe, North America and Asia (Kahlke, 2015). Woolly Mammoth extinction is thought to have occurred due to the combination of climatic changes reducing suitable habitats and an increase in anthropogenic hunting pressure (Nogues-Bravo et al.*, 2008).*

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

The species from which most mammoth ivory is currently in trade, the Woolly Mammoth *M. primigenius*, was about the same size as living African Elephants *Loxodonta africana*. Studies of bone development showed that Woolly Mammoths continued to grow throughout their life. Females were slightly smaller than males. Both male and female Woolly Mammoths grew tusks. Woolly Mammoths were born without tusks. Small milk tusks, only a few centimetres long, erupted at about six months age. After about one year they were replaced by the permanent tusks which grew continuously throughout life, at a rate of 5-15 cm per year. Woolly Mammoths apparently had a lifespan of about 60 years.

Only the tusks of the cold-adapted Woolly Mammoths have been used in recent decades for carving and for decoration. Recovered tusks from other species of *Mammuthus* are apparently too brittle to be used for carving. Unlike those of elephants, mammoth tusks have a twist, twisting in opposite directions with the tips eventually crossing in the centre. The largest known Woolly Mammoth tusk is 4.2 m long and weighs 84 kg. Typically, males' tusks reached a length of 2.4-2.7 m weighing less than 50 kg. Females' tusks are smaller, thinner, and less tapered, with a length of 1.5–1.8 m, weighing 9-11 kg.

Large, whole Woolly Mammoth tusks are distinguishable from elephant ivory by their shape (twisted and not straight), but worked mammoth ivory, especially small pieces, are difficult to differentiate from elephant ivory. Like living elephants, mammoths do not have enamel on their tusks.

One study has mentioned that mammoth ivory is graded from grades A to D and tusks are graded on their colour. The whiter (more similar to elephant ivory) a mammoth tusk, the higher the grading. Prices in 2011 were: USD 400/kg for grade A, USD 300/kg for B, USD 260/kg for C and USD 120/kg for D in Hong Kong SAR (Martin & Vigne, 2011).

Grade A mammoth ivory, nick-named "ice" by ivory carvers, can easily be passed off as elephant ivory as it looks so similar, especially when carved into small items. Unpainted cross sections can reveal cross-hatchings known as Schreger lines, which in mammoth ivory run through at a 90-degree angle rather than at a 115-degree angle as with elephant ivory. But this method does not work for carvings where the Schreger lines are not evident (such as small pieces that are not cross-sectioned).

It should be noted that an average of several of both the convex and concave Schreger lines should be used to identify if the ivory is from a mammoth or an elephant. It is also suggested that only the outer Schreger lines are used to identify the ivory (Espinoza & Mann, 2010). This suggests that it may be difficult correctly to identify mammoth ivory unless there is a clear cross section visible in the tusk. Smaller pieces or processed/carved ivory may be very difficult to tell apart from elephant ivory.

Mammoth ivory will occasionally display intrusive brownish or blue-green coloured blemishes caused by an iron phosphate called vivianite, whereas elephant ivory will not; however, this discoloration is often imperceptible to the naked eye.

Although these physical differences can in some cases provide a means of identifying mammoth ivory by expert enforcement officers, they are not always visible or obvious even to well-trained enforcement agencies tasked with determining the legality of items in international trade. In addition, elephant ivory in trade is sometimes painted or intentionally discoloured to make it appear older or more like mammoth ivory when in trade.

National utilisation
Historically mammoth ivory has been unearthed and sold domestically throughout its range. Domestic use is for decoration and jewellery only. The demand and use of mammoth ivory has been increasing over the past few decades, as it has become more available, especially since the beginning of the global moratorium on elephant ivory trade in 1989.

China is the main ivory (all types including elephant, mammoth, hippopotamus and walrus) manufacturing centre in the world and has also witnessed increased domestic use. A review of the market for elephant and mammoth ivory in Beijing and Shanghai (China) found that 90% (both elephant and mammoth) of purchases were for domestic customers, as opposed to the situation in 2002 when foreigners were the major consumers.

In another example, a recent review of the domestic mammoth ivory market in Macau SAR compared mammoth ivory sales from 2004 to sales in 2015, and found a fourfold increase in mammoth ivory sales over this period of time. China has announced a ban on domestic sales of elephant ivory, but this does not extend to mammoth ivory. *China's ivory ban came into force on the 31st December 2017* (Meijer et al., 2018).

Legal trade
International trade in mammoth ivory is not illegal in most countries and is poorly documented. Some studies have been carried out to attempt to estimate the quantities of mammoth ivory in international trade.

The major legal exporter of mammoth tusks is Russia. Mammoth tusk imports via Hong Kong SAR, one of the main trade routes into mainland China, have greatly expanded from an average of less than 9 tonnes per year from 2000 to 2003 to an average of 31 tonnes per year from 2007 to 2013. According to prices paid by some factories in Beijing, wholesale prices of mammoth ivory tusks have increased greatly recently due to the rise in demand in China.

The mammoth ivory trade route from Russia to Hong Kong SAR is thought to be the main trade route for mammoth ivory globally as there is no import tax to pay when the tusks arrive in Hong Kong SAR. Tusks are then exported to Guangzhou (mainland China), which has the two largest ivory manufacturing centres in southern China, where the ivory is carved (Esmond Martin & Vigne, 2011).

Hong Kong SAR customs data between 2005-2016 show:

- *total imports of mammoth ivory (raw tusks and/or unworked tusk pieces) of 430,000 kg, which averages 36,000 kg annually.*
- *Re-export data from Hong Kong SAR over the same time period shows total re-exports of mammoth ivory (raw tusks and/or unworked tusk pieces) of 350,000 kg, averaging 29,000 kg annually.*
- *Between 2005 – 2016 data shows that most of Hong Kong SAR's mammoth ivory is imported from Russia (395,000 kg) and most if its re-exports are destined for mainland China (330,000 kg), as per the traditional trade route mentioned above.*
- *Customs data from Hong Kong SAR also lists the origin of the specimen and imports predominantly originate from Russia (410,000 kg).*
- *10,000 kg of imports have origins that are not places where preserved mammoth ivory is found (e.g. the Netherlands and Germany were listed as the origin for a total of 8,250 kg of mammoth ivory imports)*
- *More than 1,000 kg of mammoth ivory imports were reported with the origin being an African Elephant range State (Mozambique, South Africa, Kenya or Chad).*
- *The destinations of 340,000 kg of mammoth ivory re-exports were China and Macao SAR.*

USA customs data between 1999-2013 from the U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS) shows total imports into the USA of 22,480 whole mammoth tusks, 1,625 mammoth ivory pieces, 11,608 kg of mammoth ivory pieces and 546,743 mammoth ivory carvings (full breakdown given in Table 1).

- *The origin of the mammoth ivory was also reported in the US trade data and small amounts of mammoth ivory were reported as originating in African Elephant range States (Nigeria, South Africa and Tanzania) and Asian Elephant range States (China, Indonesia and Thailand).*
- *Mammoth commodities (including non-ivory related commodities such as bone carvings, teeth and bones) imported into the USA from South Africa (including being reported as South Africa being the origin of the commodity), totalled 1,270 items have been exported from South Africa to the USA between 1999 and 2013.*

***Table 1**. USA trade data from the U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS) for import of mammoth ivory commodities between 1999 and 2013.*

Year	Number of Whole Tusks	Total Ivory Pieces		Number of Ivory Carvings
		Number of	Weight (kg)	
1999	79	-	-	456
2000	1	-	-	515
2001	139	-	-	169
2002	172	-	-	19,187
2003	17,894	-	1,620	27,209
2004	94	4	5,152	4,182
2005	967	258	1,889	78,499
2006	1,103	44	1,874	113,212
2007	183	295	856	159,100
2008	63	387	217	15,549
2009	49	120	-	15,816
2010	153	463	-	98,972
2011	649	15	-	4,028
2012	161	16	-	4,835
2013	773	23	-	5,014
TOTAL	**22,480**	**1,625**	**11,608**	**546,743**
Top Three Exporters				
1	Hong Kong SAR (21,746)	Hong Kong SAR (551)	Russia (11,303)	Hong Kong SAR (486,804)
2	Indonesia (333)	Indonesia (418)	Germany (300)	Indonesia (46,426)
3	South Africa (126)	Germany (321)	Indonesia (5)	Taiwan POC (9,990)
Top Three Origins				
1	Russia (20,696)	Russia (890)	Russia (11,303)	Russia (435,481)
2	Hong Kong SAR (1,048)	USA (666)	Germany (300)	Hong Kong SAR (65,066)
3	USA (320)	South Africa (30)	USA (5)	USA (33,090)

Reports have found evidence of elephant ivory being sold as mammoth ivory and although most Chinese buyers claim to prefer elephant ivory, many are not able to distinguish between carved pieces of mammoth and elephant ivory. A recent study published on the US ivory market provides examples of actual cases in the USA where elephant ivory was sold under the claim that it was mammoth ivory, such as a felony conviction in New York of a Manhattan-based antiques merchant for intentionally mislabelling illegal elephant ivory as “carved mammoth tusks”.

Evidence has also been presented from the USA that reports 55 imported carvings (from Hong Kong SAR), declared as mammoth ivory were in fact a mixture of both mammoth and elephant ivory (HSUS, 2002).

A report in 2011 showed that some vendors in southern China were selling elephant ivory as mammoth ivory, but the proportion of this mislabelling is unknown (Vigne & Martin, 2011).

Recent reports have also suggested that Chinese tourists in Myanmar are purchasing elephant ivory products which are mislabelled as mammoth ivory (Lucy Vigne & Martin, 2018).

There is no empirical evidence showing the amount/proportion of elephant ivory that is traded under the guise of mammoth ivory, many of the reports referenced are based on single events of elephant ivory being sold as mammoth ivory, although it is clear these substitutions are occurring, the level to which they are happening is unknown.

Parts and derivatives in trade
There is apparently only a relatively small demand for commercial international trade in parts and derivatives of Woolly Mammoths other than carved ("worked") mammoth ivory. There are collectors and traders who specialise in trade of fossils and other artefacts and a review of some of their web sites shows that their trade in mammoths is mostly in whole tusks, with some availability of Woolly Mammoth hair, bones and molar teeth, too. These parts and derivatives are deemed not to have an impact on elephant ivory trafficking.

There are non-commercial scientific exchanges of mammoth parts (such as bones, skin, hair, and DNA) for research and education by museums and universities.

Actual or potential trade impacts
The rationale for listing Woolly Mammoth in Appendix II is the potential trade impact of mammoth ivory on living elephants. Listing the species in Appendix II will put the onus upon exporting countries to make proper identification and determination of legal acquisition before issuing export permits (according to Article IV of the Convention). Exporting countries will need to make sure that specimens marked as mammoth ivory are indeed mammoth ivory and not elephant ivory.

The legal mammoth ivory trade has other impacts: permanent ecological damage is caused during the work to unearth them in the tundra regions of Siberia. The digging is done using very high-pressure water pumps (like those used on fire trucks) run by large petrol engines, to pound away at the permafrost and gouge out whole hillsides and deep pits in the ground. This work causes irreversible ecological damage to the permafrost with run-off polluting the streams and rivers.

In addition, some scientific knowledge that could be reaped from mammoth remains and from other artefacts of paleontological interest that are unearthed (including remnants of other pre-historic animals like sabre-toothed cats, Woolly Rhinoceros and others), is lost to science in this process.

Additional Information

Conservation, management and legislation
National
Many countries have laws regulating or banning ivory trade, but in all the ones we are aware of this is directed at elephant ivory. India is the only country known to ban import and export of mammoth ivory. We were not able to obtain information on domestic laws concerning mammoth ivory from all countries, but we found out information about a few. For example, some countries are currently working to amend their regulations to include mammoth ivory within their definitions of ivory. Trade in mammoth ivory is not illegal in the European Union. Canada regulates trade in mammoth ivory as a "fossil" or as ancient relics as part of their antiquities or cultural property laws. Federal law in the USA does not regulate mammoth ivory trade. However, many States in the USA have State laws banning or regulating trade in ivory, and in some of these States the definition of ivory also includes mammoth ivory.

China's elephant ivory ban came into force on the 31st December 2017. Prior to the ban, a market survey of 50 accredited ivory outlets in 22 cities in China showed that 12 stores had changed their business to trade in mammoth ivory and 38 had closed (Zhao et al., *2017). Research looking into ivory (all types) consumption and consumer perception towards the ivory ban in China showed that across 15 cities, incidences of ivory purchases had declined significantly since the implementation of the ivory ban (Meijer* et al., *2018).*

Hong Kong SAR will also adopt an ivory ban which will come into force in 2021, but concerns have been raised that as mainland China accounts for 90% of Hong Kong SAR's ivory (all types) purchases that this will enable a four-year overlap where large amounts of laundering may occur with elephant ivory being smuggled into mainland China under the guise of being mammoth ivory (Cheung et al.*, 2017; Martin & Vigne, 2015).*

It has been reported that mammoth tusk hunters require Russian government permits to sell mammoth ivory, but many hunters circumvent this legislation and continue hunting without permits (Farah & Boyce, 2019).

International: It may be argued that the UNIDROIT Convention on Stolen or Illegally Exported Cultural Objects (Rome, 1995) could possibly relate also to mammoth specimens that have been "stolen" (i.e., unlawfully excavated and/or exported). The UNIDROIT Convention enables Parties to demand restitution from other Parties regarding stolen objects which include also "specimens of fauna, flora, minerals and anatomy, and objects of paleontological interest".

Artificial propagation/captive breeding
Media reports claim that there are projects to try to clone or "revive" Woolly Mammoths using ancient DNA, such as the project by Prof. George Church at Harvard University in the USA. If successful, Woolly Mammoths (or a mammoth-elephant hybrid) could possibly be reintroduced into the wild in the future. A site in northern Siberia has been proposed for them.

Implementation challenges (including similar species)
It would not be possible to perform a non-detriment finding (NDF) on mammoth ivory as an extinct species. CITES permits would need to be issues on the basis of a Legal Acquisition Finding only.

Potential risk(s) of a listing
The inclusion of Woolly Mammoth in Appendix II could set a precedent that could lead to future proposals of other extinct prehistoric species such as the Woolly Rhinoceros Coelodonta antiquitatis*. Proposals such as these could detract from CITES' aim of ensuring that international trade in specimens of wild animals and plants does not threaten their survival.*

Potential benefit(s) of listing for trade regulation
Given the issues with some of the origins of mammoth ivory explained above (in both US and Hong Kong SAR customs data) a CITES listing means Parties would be required to report imports and exports of Woolly Mammoth ivory which would enable an understanding of the scale and nature of the mammoth ivory trade.

Other comments
The SS states that listing the Woolly Mammoth in Appendix II is not intended to stop trade in mammoth ivory but rather to facilitate documentation of the international trade in mammoth ivory in order to better understand it and its implications for living elephant populations.

The "look-alike provision"
Paragraph 2 of Article II of the Convention on "Fundamental Principles" explains the reasons why a species should be listed in Appendix II, as follows:
2. Appendix II shall include:
(a) all species which although not necessarily now threatened with extinction may become so unless trade in specimens of such species is subject to strict regulation in order to avoid utilisation incompatible with their survival; and
(b) other species which must be subject to regulation in order that trade in specimens of certain species referred to in sub-paragraph (a) of this paragraph may be brought under effective control.

Sub-paragraph (a) clearly explains that the goal of Appendix II is to prevent extinction of species that are or may be threatened due to trade, while sub-paragraph (b) provides for the listing of "look-alike" species in Appendix II even if not they are not threatened. Sub-paragraph (b) adds the notion that "other species" shall be listed in Appendix II when it will assist with the "effective control" of trade in those species threatened with extinction. It is important to note that no biological criteria are attached to sub-paragraph (b) and as such, the requirement that a species be "threatened with extinction" does not apply here. The criteria for inclusion of species under sub-paragraph (b), are listed in Annex 2b to *Resolution Conf. 9.24 (Rev. CoP17)*, as follows:

"Species may be included in Appendix II in accordance with Article II, paragraph 2 (b), if either one of the following criteria is met:
A. The specimens of the species in the form in which they are traded resemble specimens of a species included in Appendix II under the provisions of Article II, paragraph 2 (a), or in Appendix I, so that enforcement officers who encounter specimens of CITES-listed species are unlikely to be able to distinguish between them; or
B. There are compelling reasons other than those given in criterion A above to ensure that effective control of trade in currently listed species is achieved."

These criteria have led to the nick-naming of Article II, sub-paragraph (b) of the Convention as "the lookalike provision", and they have been used in the past for listing a number of species.

Listing an extinct species
According to the SS, CITES legal experts have determined that there is nothing in the Convention or Resolutions against listing an extinct species. The SS also notes that at CoP17 (Johannesburg, 2016), Israel submitted a working document on trade in mammoth ivory. In the Secretariat's comments to that document, they wrote that regulating mammoth ivory trade "may appear to fall outside of the legal scope of the Convention". The Secretariat's comment did not consider whether an extinct species could be listed under Article II, paragraph (2)(b) of the Convention and as such does not provide an actual legal analysis. A full review of the Convention and of *Resolution Conf. 9.24 (Rev. CoP17)* on "Criteria for Amendment of Appendices I and II" shows instead that the listing of Woolly mammoth in Appendix II fully conforms to the Convention.

Resolution Conf. 9.24 (Rev. CoP17) on "Criteria for Amendment of Appendices I and II" addresses the inclusion of extinct species in the Appendices in a few places, as follows:

First, Annex 3 of *Resolution Conf. 9.24 (Rev. CoP17)* states that "Extinct species should not normally be proposed for inclusion in the Appendices. Extinct species already included in the Appendices should be retained in the Appendices if they meet one of the precautionary criteria included in Annex 4.D." This suggests that extinct species should not generally be listed in the Appendices; it does not explicitly state that such species should never be listed. The Resolution states that extinct species should be retained in the Appendices if there is a precautionary necessity. Furthermore, Annex 3 to *Resolution Conf. 9.24 (Rev. CoP17)* also recognises that in species listing proposals "Parties are encouraged to note any extinct species in the higher taxon and to clarify whether these are included or excluded from the proposed listing."

In addition, Annex 4 on "Precautionary measures", Paragraph D calls for retention of extinct species in the CITES Appendices in any one of four circumstances:
1. they may be affected by trade in the event of their rediscovery; or
2. they resemble extant species included in the Appendices; or
3. their deletion would cause difficulties implementing the Convention; or
4. their removal would complicate the interpretation of the Appendices.

Wider views on the mammoth ivory trade
There is a basic dichotomy of thought regarding the regulation of mammoth ivory trade in terms of its impact on living elephants. One view holds that mammoth ivory trade should be banned along with the trade in elephant ivory so as to prevent laundering of elephant ivory. Under this view, great emphasis should be put on demand reduction by teaching consumers not to use any ivory. An alternative view holds that mammoth ivory trade should be promoted as an alternative to elephant ivory, since mammoths are extinct anyway.

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Proposals 14-17 and 20 and 21: Australian endemic species proposals resulting from the Periodic Review of the Appendices

Proponent (for all proposals): Australia

Introduction

These six proposals result from the Periodic Review of the Appendices *(Res. Conf. 14.8 (Rev. CoP17*)), undertaken by the CITES Animals Committee. The Periodic Review recognises there is a need to conduct reviews of species listed in Appendices I and II to ensure that species are appropriately listed, based on current biological and trade information, and that this properly reflects its conservation needs. Many of the taxa reviewed through this process are species which were listed very early in the history of the Convention and for which little or no trade has since been recorded. Due to the similarities in the proposals they are discussed together here.

The reviews that resulted in these proposals were all undertaken by Australia and concern two birds and four mammals, all endemic to Australia. These have been listed in Appendix I since the early days of CITES, when a number of Parties, including Australia, included their threatened species in the Appendices regardless of whether trade was an important conservation issue for them or not. All four mammals and one of the birds are still extant and one bird subspecies is extinct. In all cases, Australia has determined that trade is not, and never has been, a concern for the species, with all the extant species fully protected under national legislation. None of these species therefore meets the trade criteria for inclusion in Appendix I, although some may meet the biological criteria. Furthermore, it is unlikely that a transfer of any of the species to Appendix II will stimulate trade in these or any other species included in Appendix I thus meeting the Precautionary Measures of *Res. Conf. 9.24 (Rev. CoP17)* Annex 4 A2. The Supporting Statements provide comprehensive and up-to-date accounts of the status of each of the species and of conservation measures currently in place and these will not be further discussed in detail here.

All these species are proposed for transfer to Appendix II. This is because under *Res. Conf. 9.24 (Rev. CoP17*), extant species in Appendix I that do not meet the criteria for inclusion in the Appendices should first be transferred to Appendix II for a period of two intervals between CoPs before being deleted from the Appendices. In one case (Prop. 15 *Pseudomys fieldi praeconis*) a taxonomic change is also proposed to bring the listing into line with CITES standard nomenclature.

Prop. 14 Transfer of the Greater Stick-nest Rat *Leporillus conditor* from Appendix I to Appendix II

Summary and Analysis: The Greater Stick-nest Rat *Leporillus conditor* is endemic to Australia and is classified as Near Threatened by IUCN (2016). It was listed in Appendix I in 1975. No trade has been recorded in the CITES Trade Database since the species was listed. It does not meet the trade criteria for inclusion in Appendix I and it is unlikely that a transfer of the species to Appendix II will stimulate trade in this or any other species included in Appendix I thus meeting the Precautionary Measures of *Res. Conf. 9.24 (Rev. CoP17*) Annex 4 A2ai).

Prop. 15 Transfer of the Djoongari or Shark Bay Mouse *Pseudomys fieldi praeconis* from Appendix I to Appendix II, with the new listing modified to *Pseudomys fieldi* in accordance with standard CITES nomenclature

Summary and Analysis: The Shark Bay Mouse is currently listed in Appendix I as *Pseudomys fieldi praeconis.* It was first listed as *P. praecornis* in Appendix I in 1975 and a second taxon, "*Pseudomys fieldi*", was also listed at that time. In 1979 "*Pseudomys fieldi*" was removed from the Appendices (known at the time only from one specimen on the mainland, and subsequently declared as extinct there). Fifteen years later *P. praecornis* was synonymised with *fieldi*, with *fieldi* taking

priority over *praeconis*. With no other subspecies extant the taxon should correctly be identified as *Pseudomys fieldi* under standard CITES nomenclature. This species is endemic to Australia and is currently classified as Vulnerable by IUCN (2016). No trade has been recorded in the CITES Trade Database since the taxon was listed. It does not meet the trade criteria for inclusion in Appendix I. No other species of *Pseudomys* are included in the Appendices and therefore transfer of this taxon will not stimulate trade in any other species included in Appendix I thus meeting the Precautionary Measures of *Res. Conf. 9.24 (Rev. CoP17)* Annex 4 A2ai).

Prop. 16 Transfer of the False Swamp Rat *Xeromys myoides* from Appendix I to Appendix II

Summary and Analysis: The False Swamp Rat *Xeromys myoides* is endemic to Australia and is classified as Vulnerable by IUCN (2016). It was listed in Appendix I in 1975. No trade has been recorded in this species. It does not meet the trade criteria for inclusion in Appendix I. No other species of *Xeromys* are included in the Appendices and the transfer of this taxon will therefore not stimulate trade in any other species included in Appendix I thus meeting the Precautionary Measures of *Res. Conf. 9.24 (Rev. CoP17)* Annex 4 A2ai).

Prop. 17 Transfer of the Central Rock-rat *Zyzomys pedunculatus* from Appendix I to Appendix II

Summary and Analysis: The Central Rock-rat *Zyzomys pedunculatus* is endemic to Australia and is classified as Critically Endangered by IUCN (2016). It was listed in Appendix I in 1975. No trade has been recorded in the CITES Trade Database and the species does not meet the trade criteria for inclusion in Appendix I. No other species of *Zyzomys* are included in the Appendices and therefore transfer of this taxon will not stimulate trade in any other species included in Appendix I thus meeting the Precautionary Measures of *Res. Conf. 9.24 (Rev. CoP17*) Annex 4 A2ai).

Prop. 20 Transfer of the Lesser Rufous Bristlebird *Dasyornis broadbenti litoralis* from Appendix I to Appendix II

Summary and Analysis: The Lesser Rufous Bristlebird *Dasyornis broadbenti litoralis* is extinct, having last been reliably recorded in 1906. It was listed in Appendix I in 1975. It is noted in the Appendices that it is “possibly extinct”. The subspecies is listed as extinct under the Environment Protection and Biodiversity Conservation Act 1999 and presumed extinct under the Western Australian Wildlife Conservation Act 1950. It was a subspecies of the Rufous Bristlebird *Dasyornis broadbenti,* an Australian endemic, which is not listed in the CITES Appendices and has been classified as Least Concern by BirdLife International and IUCN since 2004. No trade has been reported in this species since it was listed. This subspecies did somewhat resemble *Dasyornis longirostris,* which is also included in Appendix I and is the subject of CoP18 Prop. 21 to transfer the species to Appendix II. However, it is considered unlikely that a transfer of this extinct subspecies to Appendix II will stimulate trade in any other species included in the Appendix I thus meeting the Precautionary Measures of *Res. Conf. 9.24 (Rev. CoP17)* Annex 4 A2ai).

Prop. 21 Transfer of the Long-billed Bristlebird *Dasyornis longirostris* from Appendix I to Appendix II

Summary and Analysis: The Long-billed Bristlebird *Dasyornis longirostris* is endemic to Australia and is classified as Endangered by BirdLife International and IUCN (2016). It was listed in Appendix I in 1975. No trade has been reported in this species or the extinct *Dasyornis broadbenti litoralis* in the CITES Trade Database. No other species of *Dasyornis* are listed in the Appendices. It does not meet the trade criteria for inclusion in Appendix I. It is unlikely that a transfer of the species to Appendix II will stimulate trade in this or any other species included in Appendix I thus meeting the Precautionary Measures of *Res. Conf. 9.24 (Rev. CoP17)* Annex 4 A2ai).

Inclusion of Reeves's Pheasant *Syrmaticus reevesii* in Appendix II

Proponent: China

Summary: Reeves's Pheasant *Syrmaticus reevesii* is a distinctly plumaged pheasant endemic to central China. Adult males have black and white-banded tail feathers that can measure up to 2.4 m, the longest of any pheasant species. *Syrmaticus reevesii* was widely distributed and relatively common in central China until the mid-20th century, but since then appears to have declined rapidly and is now primarily concentrated in three fragmented subpopulations (the Dabie and Qinling Mountains and the Shennongjia mountainous massif). Surveys in 2011-2012 at 89 sites across the species' known post-1980 range indicated that *S. reevesii* had disappeared from 46% of survey sites and declined in a further 52% of sites. The total effective population size of the species is estimated to have declined by at least 50% in the last ten years, equivalent to two generations. A survey published in 2009 estimated the population size at 23,000 individuals, while the IUCN Red List assessment in 2018 estimated 3,000-5,000 mature individuals and up to 15,000 individuals in total, and categorised the species as Vulnerable with a decreasing population trend. The species has been introduced to Pakistan, the USA and several European countries for sport hunting and for ornamental purposes, and some populations have naturalized.

The main threats to the species are reported to be illegal hunting, habitat loss and fragmentation, and deliberate poisoning in farmland. While the species is protected from hunting under Chinese legislation, surveys in 2011-2012 found evidence of poaching in 83% of surveyed sites where the species was still known to be present. The species is reportedly hunted by local communities for food, while eggs are taken and live individuals are captured to supply zoos and captive breeding centres, although it is not clear whether this is to meet domestic or international demand. There is international demand for feathers that are reportedly used for circus costumes, home decorating and flower arranging, and fly-tying for angling. Although exports of the species for commercial purposes have reportedly been prohibited under Chinese legislation since 1989, the EU and the USA have reported imports of relatively large quantities of feathers originating in China for commercial purposes (approximately 40 kg and 1,500 wild-sourced and 1,800 captive-bred/born sourced feathers imported by the EU between 2007 and 2015; 5 kg and 27,000 wild-sourced/ranched and 127 kg and 90,300 captive-bred/born sourced feathers imported by the US between 2007 and 2013). No imports of the species have been reported by the EU since 2015 (US import data for 2014 onwards were not available for analysis).

Reported trade in feathers is likely to consist of the longer tail feathers (of which each adult male has two) but may comprise other feathers as well. It is anecdotally reported that feather imports to Europe have substantially increased in price and decreased in length, which may indicate that availability has decreased.

Analysis: *Syrmaticus reevesii* has a relatively extensive but fragmented range in central China, with a maximum population size estimated as 15,000 individuals. There is evidence of declines and local extirpations of many previously known populations since the mid-20th century. The total population is estimated to have declined by at least 50% in the last ten years (two generations), and therefore meets the biological criteria for inclusion in Appendix I. While habitat loss is reportedly the principal threat to the species, hunting is also reported to be a threat, despite protection under national legislation. Although commercial trade in the species from China has been prohibited since 1989, commercial trade in wild, ranched and captive specimens from China has been reported. Although the extent to which international trade is driving the observed population declines is uncertain, the species meets the criteria for inclusion in Appendix II in accordance with the precautionary measures in Annex 4 of *Res. Conf. 9.24 (Rev. CoP17)*.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy
Scientific synonyms: *Phasianus reevesii* J. E. Gray, 1829.
Range
China.

IUCN Global Category
Vulnerable A2cd+3cd+4cd; C2a(i) (Assessed 2018, Criteria version 3.1).
Assessed as endangered (A2cd+3cd+4cd) in the China Biodiversity Red List published in 2015.

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I
B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

Endemic to central China. *The extent of occupancy has been estimated at 983,000 km^2 (BirdLife International, 2018).*

Syrmaticus reevesii *was widely distributed and quite common in central China in the relatively recent past (Cheng* et al., *1978).* While the species was widely known from western, central to eastern provinces of China historically, now there remain three main shrinking fragmented subpopulations in the Dabie Mountains, the Qinling Mountains and the Shennongjia mountainous massif, *and the species' range has become divided into eastern and western regions (Zhou* et al.*, 2015).* The population size began to decrease during the mid-20th century due to illegal hunting and habitat destruction. According to the report of the First National Terrestrial Wildlife Resource Survey published in 2009, there were about 23,000 *S. reevesii* in the wild in China. In 2018, the Red List estimated the population size to be 3,500-15,000 (*approximately 3,000 – 5,000 mature individuals (BirdLife International, 2018)*), with a decreasing population trend. The effective population size of this pheasant has decreased by about 99% in the past one thousand years, and the species was found to have experienced a severe genetic bottleneck. Surveys from 2011-2012 in 89 historical distribution sites *covering the species' entire post-1980 known range in central China* found that the pheasant had disappeared from 46% of the surveyed sites and revealed a population decline in a further 46 sites (52%) (*including protected areas, although densities were higher than in non-protected areas), and the authors estimated a population reduction of ≥50% over the last 10 years (Zhou* et al., *2015), equivalent to two generations (the species has a generation length of 5 years according to BirdLife International, 2018).*

From 2007-2015, 41 kg and 3,674 pieces of feathers of *S. reevesii* were commercially imported into the EU reportedly originating from China. However, since *S. reevesii* is a nationally protected animal in China, no specimens have been approved to be used or exported for commercial purposes from China.

Trade reported in the SS is recorded in the CITES Trade Database (CITES Trade Database) because Syrmaticus reevesii *has been listed in EU Annex D since 1997 (applies only to live specimens; whole or substantially whole, dead specimens; and any feathers or any skin or other part with feathers on it;*
Imports of S. reevesii *from China comprised:*

- *41 kg of feathers (all reportedly wild-sourced, for commercial purposes)*
- *3,674 individual feathers (1,500 reportedly wild-sourced, 400 with source unspecified and the remainder sources "C" and "F", all for commercial purposes with the exception of 400 with purpose unspecified).*

According to data included in US Fish and Wildlife Service's LEMIS database for the period 2007-2013, trade in S. reevesii *arriving at USA borders originating from China comprised:*

- *117,218 feathers (1% wild-sourced, 22% ranched and the remainder captive born/bred); of this trade, 600 feathers (all wild-sourced) were refused entry and seized*
- *129 kg of feathers (3% wild-sourced, 1% ranched and the remainder captive-bred)*
- *3,159 items of jewellery and 55 unspecified items (>99% wild-sourced).*

The vast majority (>99%) of trade was reportedly for commercial purposes.

The demand for birds as pets or exhibition specimens or for profitable farming has increased the possibility of poaching of wild pheasants. Eggs are collected, chicks and sometimes even adults are illegally captured to meet the demand for recruits for some zoos or breeding centres (*although it is not clear whether this is domestic or international demand)*, which could directly cause population decline and a decrease in reproductive rate.

Online surveys indicate that feathers are the main parts in trade (the tail feathers of the male of the species can measure up to 2.4m in length, the longest of any pheasant). In the past the tail feathers were used as a decoration in traditional opera costumes in China. Now, operas generally use plastic or dyed stitched feathers as

substitutes. *Demand for feathers in Europe is reportedly for circus costumes, home decorating and flower arranging, and fly tying for the fishing industry. It appears most of the feather markets are rather niche, and there are alternative sources of materials for all of these uses (J. Carroll,* in litt., *2019). It has been reported that feather imports to Europe have dramatically decreased in length (which is age related) and increased in price, suggesting that availability has decreased (J. Carroll,* in litt., *2019).* Removal of tail feathers might affect the success of courtship and mating; one adult male pheasant can only grow two long tail feathers in each year. *It is not clear whether tail feathers are generally removed from live or dead birds.*

The majority of adverts for S. reevesii *specimens found online (whether Chinese or English language) do not specify whether the source is wild or captive; those that do (generally for live birds/eggs rather than feathers) specify that they are sourced from captive birds. Some adverts include a copy of the relevant permit for the breeding facility but most do not. Feathers from captive-bred and wild birds would be indistinguishable (J. Carroll,* in litt., *2019).*

Additional Information

Threats

Illegal hunting, habitat destruction, and poison in farmland are the three main threats to the survival of this species. *Surveys conducted in 2011-2012 in 43 sites with presence of the species found 83%, 26% and 20% of sites had evidence of poaching, habitat loss and use of poison, respectively (Zhou* et al.*, 2015). According to the IUCN Red List assessment, the main threat to the species is the continuing deforestation within its range, which is reducing and fragmenting its habitat (BirdLife International, 2018).* The species is threatened by poaching for food by local communities and their eggs are also illegally collected for food or breeding (see section above).

According to the Database of Legal Instruments of China, 11 *S. reevesii* were known to be poached for food by indigenous farmers and one individual bird was illegally traded in Anhui Province from 2013-2017. *One person who illegally hunted four* S. reevesii *was sentenced to five years in prison and 30,000 RMB fine (http://miliwealth.com/plus/view.php?aid=938).*

Conservation, management and legislation

Syrmaticus reevesii has been listed as a second class species on the list of Wildlife under Special State Protection of China since 1989, which is under the protection of the Law of the People's Republic of China on the Protection of Wildlife, and Regulations on the Nature Reserves of the People's Republic of China. The hunting, killing, selling, buying and utilisation of this pheasant and its products is strictly prohibited. Administrative approvals by provincial conservation competent departments are needed for selling, buying and utilisation of this pheasant and its products for special purposes like scientific research, captive breeding and public exhibitions. Permission from the relevant responsible department of the State Council must be obtained for the export of this pheasant and its products.

The Chinese Government has taken a series of actions to conserve forests in recent decades, including a ban on logging in the middle and upper reaches of large rivers and imposing forests under the National Forest Protection Programme. At present, there are 49 national nature reserves (NNRs) which include *S. reevesii* on the lists of protection objectives. The total area of all relevant NNRs is more than 17,040 km^2. Some of the protected areas focus on the conservation of this pheasant with long-term scientific research, population monitoring and habitat protection. However, the species is evaluated as one of the ten Chinese wildlife species that have the lowest ratio of habitats protected by NNRs to total habitat, and there is still poaching, illegal logging, cropland development and tourism in these nature reserves.

Artificial Propagation/captive breeding

About 32 live *S. reevesii* have been approved by the forestry department in Shandong Province for captive breeding and exhibition in zoos from 2016-2018. The total number of individuals that have been approved for captive breeding and scientific exhibition in China is unknown.

The species has been introduced to many countries outside China since the beginning of the last century for sport hunting and for ornamental purposes. The UK first introduced this pheasant in 1831. After that, many European countries and the USA began to introduce this pheasant directly or indirectly from China. Introduced populations are reported in the Czech Republic, France, Pakistan and the USA. Among them, the populations in Pakistan and the USA may rely on recruits due to the low population size and the effect of inbreeding degradation, while independent populations can be found in the wild in the Czech Republic and France. It is treated as a game or sport hunting species in the Czech Republic and Slovakia. *In the UK on some estates there is a history of mixing a few* S. reevesii *in with other pheasant species for driven shooting. Most of those are reared locally. There should be no market for importing wild* S. reevesii *from China although it might take place in*

the mistaken belief wild birds would naturalise more readily (J. Carroll, in litt., *2019). However, further deliberate releases of this species into the wild are illegal in the UK (S. Dowell,* in litt.*, 2019).*

The Zoological Information Management System managed by Species360 provided holding records of *S. reevesii* indicating that 258 individuals can be found in 60 institutions from Asia, Europe, North America and South America. The average number of individuals in these institutions is fewer than ten, except for some zoos, and 47 chicks were hatched within the last year. *This species is widely held and bred successfully in captivity so the population is likely to be much larger than the known zoo population. At present there is no formal studbook for the captive population of this species (S. Dowell,* in litt., *2019).*

Implementation challenges (including similar species)
Distinctive stripes and colour patterns on the tail feathers of pheasants can be used in distinguishing between different Phasianidae species, including the other species in the genus (three other species in the genus have been included in CITES Appendix I since 1975: *S. humiae* and *S. ellioti* occur in mainland China while *S. mikado* occurs in Taiwan, Province of China).

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Transfer of Black Crowned-crane *Balearica pavonina* from Appendix II to Appendix I

Proponents: Burkina Faso, Côte d'Ivoire and Senegal

Summary: Black Crowned-crane *Balearica pavonina* is a distinctive African waterbird that has a low reproductive capacity, with an average of one juvenile reared by each breeding pair annually. The species occurs from Senegal and the Gambia to central Ethiopia, northern Uganda and northern Kenya; it is native in 13 countries and vagrant in a further 10. Two subspecies are recognised: *B. p. pavonina* occupies the western part of the species' range from Senegal and the Gambia to Chad, while *B. p. ceciliae* occurs from Chad to Sudan, South Sudan, Ethiopia, Eritrea and northern Kenya.

In 2004, the total population of the species was estimated at 43,000–70,000 individuals, or ca. 28,000–47,000 mature individuals. The species has been categorised as Vulnerable on the IUCN Red List since 2010 on the basis of an estimated worst-case decline of 30–49% over the previous three generations (45 years). However, it was noted that the true extent of the decline was uncertain and could be greater since the accuracy of both the more recent (2004) and historic (1985) population estimates available for *B. p. ceciliae* was questionable. No more recent total population estimates for the species or either subspecies are available. Efforts to obtain more accurate estimates are considerably limited by political instability across large parts of the species' range. Declines have been reported in populations in Benin, Burkina Faso, the Gambia, Mali, Nigeria, Sudan, South Sudan and Togo, although the extent of these declines is unclear.

Live trapping for local domestication or international trade is reportedly one of the most significant threats to the species. Hunting of the species for food, use of parts in traditional medicine, and use of feathers in traditional dance have also been reported as a threat in certain areas. Legal international trade was mainly for commercial purposes and zoos. Hunting and trapping are considered to have contributed to the near-extinction of the species in Mali and Nigeria and localised declines in Senegal. Since 2007 exporters have reported a total of 524 live *B. pavonina* in trade, 36% of which were reportedly captive-born or bred, although the species is considered difficult to maintain and breed in captivity. Concerns over the sustainability of the reported trade in wild-sourced birds led to the species being included in the Review of Significant Trade (RST) and resulted in recommendations to suspend trade from Guinea, Sudan, South Sudan and Mali which remain in place. Illegal trade, including cross-border trade, is reported to be a concern in at least seven range States, however the extent of this trade is unclear.

Habitat loss and degradation, human and livestock disturbance, and direct poisoning to reduce crop depredation also reportedly pose a threat to the species. *Balearica pavonina* is legally protected in most range States, but this protection is considered to be largely ineffective due to low public awareness and insufficient resources for enforcement.

Analysis: *Balearica pavonina* has a wide but fragmented distribution and low productivity. It has an estimated population of 43,000–70,000 individuals. In 2010, the population was estimated to have declined by 30–49% over three generations (45 years), but the true decline may be greater depending on the status of one of the two subspecies, *B. p. ceciliae*, for which reliable population estimates are not available due to political instability within its range. Although the species is legally protected in most range States, live trapping for local domestication and international trade has reportedly been the cause of severe declines in certain populations. Concerns regarding implementation of the Appendix II listing have been raised through the RST process, with three range States (and one non-range State) currently subject to recommendations to suspend trade. While current reported levels of trade in wild specimens are low, illegal international trade is reported to be a concern, although the extent of this trade is unclear. Since the species is affected by international trade and the estimated population decline may be close to and could exceed 50% over the last 45 years, *B. pavonina* is likely to meet the criteria for inclusion in Appendix I.

Other Considerations: Trade suspensions are in place for several range States through the RST process, and it appears that much of the international trade that currently takes place in wild specimens is illegal; it is therefore unclear what additional protection inclusion in Appendix I would provide. However, given concerns over illegal trade and reported declines caused by harvest, a suspension of further trade from wild sources may be in the conservation interest of the species.

B. pavonina is considered similar to the Grey Crowned-crane *B. regulorum*, which occurs in Eastern and Southern Africa, and was not previously distinguished as a separate species. *B. regulorum* is currently included in Appendix II and therefore a transfer of *B. pavonina* to Appendix I may present implementation difficulties.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

CITES history

Family Gruidae was listed in CITES Appendix II in 1985. A proposal was submitted to transfer B. pavonina *to Appendix I at CoP9 in 1994, but this was withdrawn.* In April of 2009 at AC24, *B. pavonina* was included in the Review of Significant Trade (RST) as an urgent case. Range States included in the RST were Benin, Burkina Faso, Burundi, Cameroon, Central African Republic, Chad, Côte d'Ivoire, the Democratic Republic of the Congo, Eritrea, Ethiopia, Gabon, Gambia, Ghana, Guinea, Guinea Bissau, Kenya, Mali, Mauritania, Niger, Nigeria, Senegal, Sierra Leone, Sudan, Togo, and Uganda. The Animals Committee (AC) retained all of these range States in the RST in 2011 as no response from the range States had been received. At AC26, in March of 2012, the majority of range States received a provisional category of "least concern" and were removed from the RST. However, the AC found that there was "urgent concern" for Guinea and "possible concern" for Nigeria, Sudan, and South Sudan (as well as non-range State United Republic of Tanzania). In March of 2013 at SC63, it was noted that Nigeria had complied with all recommendations concerning *B. pavonina* and was removed from the RST process. It was also agreed that all Parties should suspend trade in this species from Guinea, Sudan, South Sudan, and Tanzania. These trade suspensions remain in place *(Notification No. 2018/006)*. B. pavonina *from Mali was included in the RST at AC29 (July 2017) and categorised as "action is needed" at AC30 in July 2018 (AC30 Com. 11 (Rev. by Sec.)). The working group noted that the recorded trade levels in wild specimens of this species were very high (90 birds between 2015 and 2016) considering the population in Mali was estimated to be 100 birds in 2004. It was suggested that birds may be coming from neighbouring countries but were not reported as re-exports, noting that Guinea is subject to a trade suspension for this species. The AC recommended a number of actions to be taken by Mali, including publication of an interim zero export quota within 30 days (which does not appear to have been published yet) and studies on the status of the species and harvest levels for use in making NDFs within two years. At SC70 the SC requested the Secretariat follow up with Mali to clarify the origin, provenance, and legal acquisition of specimens that were exported in 2015 and 2016 (SC70 Sum. 8).*

Taxonomy

Two subspecies are recognised: *B. p. pavonina* and *B. p. ceciliae*. *The Grey Crowned-crane* B. regulorum*, which occurs in Eastern and Southern Africa, is considered a separate species by both the current and former CITES Standard References for birds (Sibley & Monroe, 1990; Dickinson, 2003), although historically was not considered a separate species by some.*

Range

Native to Cameroon, Chad, Ethiopia, Gambia, Guinea, Guinea-Bissau, Kenya, Mali, Mauritania, Niger, Senegal, South Sudan, and Sudan. Vagrant in: Benin; Burkina Faso; Central African Republic; Congo, The Democratic Republic of the; Côte d'Ivoire; Egypt; Eritrea; Sierra Leone; Togo; Uganda. Possibly Extinct in Nigeria (BirdLife International, 2016).

IUCN Global Category

Vulnerable A4bcd (2016) ver. 3.1.

Biological criteria for inclusion in Appendix I

A) Small wild population

In 2004, it was estimated that the number of *B. p. pavonina* was approximately 15,000 individuals and *B. p. ceciliae* was approximately 28,000–55,000 individuals in the wild; this would bring the total population estimate to 43,000–70,000 individuals or roughly 28,000–47,000 mature individuals.

B) Restricted area of distribution

Balearica pavonina does not have a restricted area of distribution. The species occurs from Senegal and the Gambia to central Ethiopia, northern Uganda, and northwestern Kenya, and is known to occur as far south as Difule on the Uganda-Sudan border, the northwest corner of Murchison Falls National Park and the northern portion of Lake Turkana. *Balearica pavonina pavonina* occupies the western part of this range and scattered populations occur throughout sub-Saharan West Africa from Senegal and the Gambia to Chad, while *B. p. ceciliae* occurs in eastern sub-Saharan Africa from Chad to Sudan, South Sudan, Ethiopia, Eritrea, and northern Kenya, especially in the upper Nile River basin. Within this range, populations are limited to wetland habitats and have become severely fragmented.

C) Decline in number of wild individuals

This species has shown rapid population and range declines in recent surveys which are predicted to continue, primarily due to habitat loss and trapping for domestication or illegal international trade.

B. p. pavonina: Survey results indicate that the range of *B. p. pavonina*, which was historically nearly contiguous across West Africa, has been severely fragmented with large gaps between many of the subpopulations. It is estimated that *B. p. pavonina* declined from 15,000–20,000 individuals in 1985 to 15,000 individuals in 2004.

B. p. ceciliae: Trend data for *B. p. ceciliae* is poorly known but may have undergone a more substantial decline from 50,000–70,000 individuals in 1985 to 28,000–55,000 individuals in 2004. However, the accuracy of both the 1985 and 2004 counts for *B. p. ceciliae* is questionable and basing trends on this data is not advisable.

Based on the 1985 and 2004 counts for *B. p. pavonina* alone, IUCN Red List assessment estimates a decline of between 0–25% from 1985–2004, with a provisional estimate of a worse-case decline of 30–49% over 3 generations (45 years), though the true figure may be higher depending on the status of *B. p. ceciliae (the same estimates are included in the current (2016) and previous (2012 and 2010) Red List assessments for the species, all of which categorised the species as Vulnerable). No more recent total population estimates for the species are available, although AC26 Doc. 12.2 (Annex) cites more recent national-level population data for:*

- *Chad (at least 5,000 individuals (Tursha & Boyi, 2011), compared to 5,500 in 2004)*
- *Ethiopia (461 individuals recorded during censuses conducted in 2011 in all major wetlands (F. Debushe (CITES MA of Ethiopia)* in litt.*, to UNEP-WCMC, 2011) compared to 2,500 in 2004)*
- *Sudan (26,000 individuals (A. Al-Makki (CITES MA of Sudan)* in litt.*, to UNEP-WCMC, 2011), compared to 25,000-52,000 in 2004 for both Sudan and South Sudan combined.*

Population declines are reported to have taken place in Benin, Burkina Faso, the Gambia, Nigeria, Mali, South Sudan, and Togo, but the extent of these declines is unclear (AC26 Doc. 12.2 (Annex)). The Mali population may also be extinct (no cranes were reported during a count in 2014 in the country (AC30 Doc. 12.2 (Rev. 1)).

Efforts to undertake monitoring and surveys in Sudan have been limited due to political instability and insufficient funding. *Since accessing large sections of the range is difficult to impossible due to unrest and insurgent control, we do not have the data and trend statistics to fully understand the extent of the decline in the species (K. Morrison,* in litt.*, 2019). Should the worst-case scenario for* B. p. ceciliae *prove to be accurate this species may warrant uplisting (i.e. to Endangered) in the future (BirdLife International, 2016).*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

International trade over the period between 1986 and 2017 consisted principally of live birds, with small quantities of bodies, skins, feathers, trophies, and scientific specimens also reported in trade. The majority of trade involved wild-sourced birds, with trade in captive-bred specimens also reported. Trade was primarily for commercial purposes, with live birds also traded for breeding in captivity, as personal possessions and, to a lesser extent, for zoos, circuses or travelling exhibitions, education, and scientific purposes.

Tanzania was the biggest exporter of *B. pavonina* between 1986 and 1995 even though the species is not found in this country; it is unclear whether the trade was misidentified, and the birds exported were actually Grey Crowned-crane *B. regulorum* for which Tanzania is a range country, or the birds were imported from other countries and sold in Tanzania. Guinea also exported a large number of cranes, having its highest exports between 1991 and 1995, but this number decreased in subsequent years. Between 1996 and 2000 Mali was a significant exporter with a decrease in exports between 2001 and 2010 but rising again between 2011 and 2015. Sudan was the biggest exporter between 2001 and 2005.

According to the CITES Trade Database, from 2007 onwards exporters reported direct exports of 524 live birds (304 or 58% wild-sourced) while importers reported imports of 357 live birds (226 or 63% wild-sourced). Mali and Sudan were the top exporters of wild-sourced birds in this period (177 and 48 birds reported by Mali and Sudan

respectively, and 120 and 50 birds reported by importers from these two countries); Sudan was also the second biggest exporter of live birds reported as source "C" (74 birds reported by Sudan and 30 reported by importers). Non-range States reported exports of 169 live birds in this period; the main exporter was the Netherlands (exporting 77 birds). The main importer was China (41% of exporter-reported live birds), followed by Qatar (8%), Oman (5%), Kuwait (5%), and Russia (5%). According to exporter-reported data, direct trade in wild-sourced live birds from range States from 2007 onwards peaked in 2011 (80 birds) and subsequently declined (10 birds in 2015 and none reported in 2016 onwards). However, importers reported imports of 90 wild-sourced live birds direct from Mali between 2015 and 2016 (and not reported by Mali as either direct exports or re-exports, although annual reports have been submitted by the country for both years).

Nigeria reported exports of eight live, wild-sourced *B. pavonina* in 2014 which is of concern since recent surveys indicate this species is extinct in the wild in Nigeria (*although Edet* et al*. (2018) report that there is a "small population"*). *There were also a number of exports of wild-sourced B. pavonina reported by non-range States from 2007 onwards: 13 birds from Togo, 20 birds from Ghana, and 20 birds from Democratic Republic of the Congo (CITES Trade Database)*.

A suspension has been in place for trade in this species from Guinea, Sudan, and South Sudan since 2013; no wild-sourced exports from these countries are recorded in the CITES Trade Database from 2014 onwards (noting that South Sudan is not a Party to CITES and therefore does not report trade). *An interim zero-export quota was recommended to be published by Mali in 2018 (AC30 Com. 11 (Rev. by Sec.)) but does not appear to have been published.*

Both the legal and illegal trade in *B. pavonina* is reported to have depleted the species in the wild. In 2011 there was reported to be evidence of cross-border illegal trade between Chad, Cameroon, and Nigeria which is considered a major threat to the species; *cranes were reportedly sold at high prices in Nigeria for export to the Middle East (Tursha & Boyi, 2011). The capture of live individuals for export to international private markets was reported to be a particularly significant problem in Guinea; around 20 individuals were imported illegally into South Africa in December 2011, with unconfirmed reports that they originated in Guinea (AC26 Doc. 12.2 Annex). Kone* et al.*, (2007) reported that trade in cranes was extremely common in Mali; between 1998 and 2000, 524 individuals were captured in a region in central Mali with a wild population of only about 1,500 individuals, and exports of the species continued despite being made illegal in 1998. The capture of individuals for international trade was reported to be a particularly significant threat to the species in Sudan/South Sudan (AC26 Doc. 12.2 Annex). The International Crane Foundation/Endangered Wildlife Trust Partnership commissioned a rapid assessment of* B. pavonina *trade from Sudan in 2009/2010, conducted by the Sudanese Wildlife Society, which concluded that only 12% of all* B. pavonina *exports were given CITES export permits and appeared in the CITES Trade Database (Hashim, 2010). Sudan reported the export of 30* B. pavonina *in 2009-2010, therefore an additional ca. 220 cranes were apparently exported without permits (it is not known how many of either the reported or unreported cranes were wild-caught as opposed to captive-bred; according to K. Morrison (*in litt.*, 2019) it is likely that a large proportion, if not all, were wild).*

Additional Information

Threats

Estimates of change in the habitat available for *B. pavonina* have not been made. However, habitat loss poses a significant threat and has had an impact on population numbers. Habitat loss and degradation occur through conversion of wetlands to agriculture, over-exploitation of wetlands, overgrazing, wetland drainage, dam construction, cutting of roost trees, agricultural and industrial pollution, and industrial construction. *Human and livestock disturbance also play a role in reducing breeding productivity (K. Morrison,* in litt.*, 2019). In addition, indiscriminate pesticide application may be leading to harmful bio-accumulation of toxins, and direct poisoning to reduce crop depredation has been reported in East Africa (Williams* et al.*, 2003). A study from 2003 noted that warfare and political instability affects nations across the range of the species, particularly in South Sudan where the implementation of crane conservation measures has not been able to proceed (Williams* et al.*, 2003), it is likely that this is still the case.*

In addition to habitat loss, live trapping may be the most significant threat to *B. pavonina*. It has been reported that this species is highly prized in private collections. Birds are either trapped or eggs and chicks are removed from the nests and the individuals are raised in captivity and sold on the local, regional, or international market for considerable profit. It was reported in 2001 that a profit of ca. *USD 213* could be made in the trade of one bird in the Kano market in Nigeria. Poaching for food and the use of heads and wings in traditional medicine has also been reported, although in the Casamance region of Senegal and in parts of Burkina Faso the eating of cranes is taboo. Around the area of Niokolo-koba National Park in southeastern Senegal, near the Guinea-Bissau border, *B. pavonina* feathers are used in ritualised traditional dance and this has seriously affected the species. The hunting and trapping of *B. pavonina* have resulted in the virtual elimination of the species in Nigeria and could

lead to its extinction in Mali*; 86% of crane captors interviewed in Mali in 2001 had noticed a decline in the number of cranes in the Inner Niger Delta, and directly attributed the decline to crane capture (Kone* et al., *2007).*

Cranes have a low reproductive capacity. Most often, initial attempts at breeding fail and individuals usually do not successfully reproduce until they are four to eight years old. There is usually one dominant chick and if food is scarce the subordinate chick will die. Balearica pavonina *normally nest only once each year, and the average clutch size is about 2.5 eggs/nest, with an average of one juvenile reared by each breeding pair annually (Williams* et al., *2003).*

Conservation, management and legislation
Balearica pavonina is legally protected in most range States but this protection is thought to be inadequate; many countries do not have the financial resources to control illegal hunters. For example, in Mali, *B. pavonina* was listed as a fully protected species in Law No. 95-031 on the management of wildlife and habitats, and exports of the species from Mali were made illegal in 1998. However, interviews revealed that few crane owners were aware of the legislation, and exports continued, albeit limited by the high costs of transportation and taxes. In Niger, protection in most wetlands was reported in 1996 to be insufficient and hunting and capture, although illegal, was still taking place on a small scale.

In 1999, the Black Crowned Crane Programme was launched by the International Crane Foundation and Wetlands International to identify key areas where effective projects could be conducted to help in the conservation of the cranes and their habitat. As part of this effort, a network was established across 20 nations in West, Central, and East Africa to identify key areas where effective projects could be established for conservation of the species and their habitat.

A Conservation Management Plan was published in 2003 which identified 226 sites that supported *B. pavonina*. Approximately 21% or 48 of these sites have some degree of official habitat protection that includes National Parks (12%), Ramsar sites (4%), reserves (4%), and locally protected sites (1%).

Artificial Propagation/captive breeding
Breeding success of cranes in captivity is considered to be very low and birds are known generally to be short-lived and prone to diseases and injury. In Mali, breeding attempts have been unsuccessful. Nigeria conducted an experimental release in 1992 in association with the West African Crane Conference. There have been discussions of developing a captive breeding program in Borno State, Nigeria and a release program in the Chingurme-Duguma section of the Chad Basin National Park but only if habitat conditions are assessed and a habitat management plan is implemented.

The Chinese Association of Zoological Gardens (CAZG) and the Association of Zoos and Aquaria (AZA in the USA) both have B. pavonina *studbooks in place that are actively managed. This will improve the captive situation for the species within these two formal zoo association structures. The European Association of Zoos and Aquaria (EAZA) too has plans for a studbook to be developed to improve the captive management of this species. Note that only captive facilities that are members of these associations can be included in the studbook (at least to a large degree), and that very few private captive facilities are included in these studbooks (K. Morrison,* in litt.*, 2019).*

Implementation challenges (including similar species)
The Grey Crowned-crane *B. regulorum*, which is also currently included in Appendix II, looks very similar to *B. pavonina*. The Black Crowned-crane is distinguished by the red found in the lower part of the cheek patch, a darker neck and smaller wattle. *The Grey Crowned-crane looks rather similar and is classified at a higher level of threat on the IUCN Red List (Endangered; assessed in 2016) and the literature suggests it may be even more impacted by (international) trade (R. Thomas,* in litt., *2019).*

Other comments
A large proportion of the population may be in Sudan and South Sudan; South Sudan is not currently a Party to CITES, and a trade suspension is already in place for both Sudan and South Sudan.

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Transfer of the Mexican population of American Crocodile *Crocodylus acutus* from Appendix I to Appendix II

Proponent: Mexico

Summary: The proponents seek to transfer the Mexican population of American Crocodile *Crocodylus acutus* from Appendix I to Appendix II. Since submission, the proponent has indicated to the CITES Secretariat its intention to amend the proposal to include a zero export quota for wild specimens for consideration at CoP18. The species was included in Appendix II in 1975 and transferred to Appendix I in 1981; the population of Cuba and several Colombian populations were transferred to Appendix II in 2005 and 2017, respectively.

Crocodylus acutus is a widely distributed species, occurring in 17 range States from the USA and Mexico through to Central America, the Caribbean and northern South America. In Mexico, the species is found in both fresh and saltwater habitats in coastal and inland areas, with a distribution area estimated at just under 200,000 km^2 from Sinaloa to Chiapas states on the Pacific coast to the eastern coast of the Yucatán peninsula.

The species was categorised as globally Vulnerable on the IUCN Red List in 2009. While severely depleted historically due to overexploitation for skins, substantial recovery is reported to have taken place in several countries including Mexico, and the global population is determined to be increasing. Surveys have indicated continued increases in certain localities in Mexico, and increases in reported incidences of human-crocodile interactions in the country may be indicative of an increasing population. Although there is no reliable estimate of the current population size in Mexico, available survey data are not consistent with the wild population being small.

Reported threats in Mexico include illegal hunting for skins and meat, and habitat loss and degradation particularly as a result of tourism developments in coastal areas. There is evidence of inbreeding in certain populations that have been fragmented by tourism developments in the Yucatán peninsula. Genetic introgression with Morelet's Crocodile *Crocodylus moreletii* is also reported as a natural occurrence in this area and may pose an additional threat.

The species is in demand for international trade in skins and there are plans to develop and implement a management scheme that will aim to replicate that already in place for *C. moreletii* in the country, in consultation with the national CITES Authorities and experts in the species. This scheme will involve a combination of ranching and captive breeding, with egg collection limited to localities where monitoring indicates that populations are healthy and stable. The intended prohibition of trade in wild specimens is expected to mitigate potential negative impacts on wild populations while the proposed management scheme is refined.

Analysis: The available information indicates that the Mexican population of *Crocodylus acutus* does not meet the biological criteria for inclusion in Appendix I: it has a wide distribution within the country, and the population appears to have recovered substantially since the Appendix I listing, with continued increases in certain areas. Regarding the Precautionary Measures outlined in Annex 4 of *Res. Conf. 9.24 (Rev. CoP17)*, the species is in demand for international trade and a managed ranching/captive-breeding programme will be developed in co-ordination with the national CITES Authorities and other experts. The proponents have indicated their intention to amend the proposal to include a zero export quota for wild specimens, although it is not clear whether the quota would also apply to ranched specimens. If confirmed, the zero export quota for wild specimens would appear to be an adequate precautionary measure to allow the establishment of appropriate management systems. According to paragraphs 1b) ii) and d) of *Res. Conf. 11.21 (Rev. CoP17),* removal or amendment of a quota that is an integral part of the listing would need to be the subject of an amendment proposal, which would normally be considered at a future meeting of the Conference of the Parties.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

CITES background

The species was listed in Appendix II in 1975 and transferred to Appendix I in 1981 (the USA population was transferred to Appendix I in 1979). The Cuban population and certain Colombian populations were transferred back to Appendix II in 2005 and 2017, respectively.

Taxonomy

Although there are studies evaluating the possibility that a subpopulation of *C. acutus* on Mexico's Caribbean coast is a distinct species, CITES currently only recognises the existence of a single species.

Range

Crocodylus acutus occurs in the coastal zones of *17 countries* of the American continent: Belize, Colombia, Costa Rica, Cuba, Ecuador, El Salvador, Guatemala, Haiti, Honduras, Jamaica, Nicaragua, Mexico, Panama, Peru, Dominican Republic, the USA and Venezuela (*the SS lists 18 countries including Cayman Islands*). The Cayman Islands population may have been extirpated, with only occasional sightings of individuals that have dispersed from Jamaica, Cuba and Central America. *Cayman Islands are not mentioned as being part of the species' range in the Red List assessment for the species (Ponce-Campos* et al., *2012).*

IUCN Global Category

Vulnerable A2cd (2009) ver. 3.1.

Biological criteria for inclusion in Appendix I

A) Small wild population

Although there is no current estimate of the size of the national population of the species, a review of 50 monitoring studies conducted over the last 30 years in 86 water bodies across the species' distribution range in Mexico (all states with the exception of Yucatán) found that around 50% of studies reported encounter rates greater than or equal to 5 individuals/km, a value higher than that of *Crocodylus moreletii* in Mexico, for which the encounter rate is 3.23 individuals/km and has healthy populations (currently listed in Appendix II). The percentage of studies with encounter rates greater than or equal to 5 individuals/km was 56% for studies in the period 1988–1997 (range 0.4–73.4 individuals/km), 47% for 1998–2007 (0.1–51 individuals/km) and 47% for 2008–2018 (0.3–163 individuals/km). *Although no "guestimates" of the absolute number of individuals in the population can be made with any confidence from the available survey data, the survey data are consistent with the wild population not being small. The population unequivocally meets the criteria for Appendix II (IUCN SSC Crocodile Specialist Group,* in litt., *2019).*

Crocodylus acutus was previously listed in the Official Mexican Standard NOM-059-ECOL-1994 as Rare. In 2001 it was re-categorised as Subject to Special Protection (the lowest risk within the Standard) and has been maintained in this category up to the most recent version of the Standard (NOM-059-SEMARNAT-2010).

B) Restricted area of distribution

The global distribution area estimated by IUCN is ca. 2,533,582.33 km^2.

In Mexico, the species occurs in the states of Sinaloa, Nayarit, Jalisco, Colima, Michoacán, Guerrero, Oaxaca, Chiapas, Quintana Roo and Yucatán (with isolated records in Sonora and Campeche). For the present proposal, a re-estimation of the national distribution area was made based on bibliographic information and the experience of experts; the result was an estimated area of ca. 199,765 km^2, equivalent to 10.2% of the national territory.

C) Decline in number of wild individuals

The global population trend was determined to be increasing in the 2009 IUCN Red List assessment of the species: *overall numbers are still depleted in some countries such as Colombia and Ecuador, but substantial recovery has taken place in other areas including Cuba, Costa Rica, Mexico and Venezuela (Thorbjarnarson* et al.*, 2006).*

In the review of 50 monitoring studies described above, repeated monitoring over a period of five or more years was conducted in five localities, and in all cases the populations were observed to be stable or increasing.

The population low point regionally and in Mexico was in the 1960s–70s, prior to hunting bans implemented in the 1970s. Populations today are probably all greatly increased relative to that point, regardless of whether they are now stabilised or are still increasing (IUCN SSC Crocodile Specialist Group, in litt., *2019).*

Reports of the population structure of *C. acutus* in Mexico frequently show a greater abundance of offspring and juveniles than sub-adult and adult individuals, which is considered to be indicative of a population in equilibrium or in recovery.

Additionally, there has been a 65% increase in the number of human–crocodile interactions reported in CrocBITE, from 34 incidents in the period 2009–2013 to 54 incidents from 2014–2018.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

There is an international market for *C. acutus* skins. The legal exports of *C. acutus* from Mexico recorded between 2000 and 2017 were mostly (93%) specimens for scientific purposes (blood samples, tissue, etc.) and in a much smaller proportion pieces of bone, bodies, skulls, and skins for personal purposes. See criterion B below.

Precautionary measures

Species not in demand for trade; transfer to Appendix II unlikely to stimulate trade in, or cause enforcement problems for, any other species included in Appendix I

The species will be in demand for trade, but its transfer to Appendix II with a zero-export quota (see below) is unlikely to stimulate trade in, or create any additional enforcement issues for, any other species included in Appendix I (IUCN SSC Crocodile Specialist Group, in litt.*, 2019).*

Species likely to be in demand for trade, but its management is such that the CoP will be satisfied with:

A) Implementation by the range States of the requirements of the Convention, in particular Article IV; and

B) appropriate enforcement controls and compliance with the requirements of the Convention

Mexico has announced its intention to amend the present proposal at CoP18 to include a zero-export quota for wild-sourced specimens and has submitted documents to this effect to the CITES Secretariat (H. Benítez, in litt.*, 2019). This precautionary measure is to allow time to develop management systems in order to ensure compliance with Article IV, although it is not clear whether the quota would also apply to ranched specimens. Any removal of this quota would need to be the subject of a new amendment proposal thus allowing the Parties to assess the progress made by Mexico.*

The objective of transferring the species to Appendix II is to lay the foundations to replicate the successful management scheme implemented for *C. moreletii* in Mexico (monitoring and habitat conservation programme in combination with ranching of eggs from the wild and captive breeding) and sustain a trade that benefits local communities, the species and its habitat. Ranching will be limited to localities where monitoring indicates that populations are healthy and stable, in local communities committed to the conservation and sustainable use of the species and its habitat, with the endorsement of the Group of Specialists in Crocodilians in Mexico (GEC-Mexico) and the national CITES Authorities. Therefore, no hunting or trade of specimens other than skins produced on farms will be authorised. It is expected that the ranching activities to be carried out under the Wildlife Conservation Management Units (UMA) scheme will promote the conservation of *C. acutus* habitat as a source of revenue for the communities and serve to mitigate land use change (see “Threats” below).

The GEC-Mexico is preparing a *C. acutus* monitoring programme which aims to monitor the status and trends of the main wild populations of *C. acutus* in the entire area of its distribution in Mexico. In order to facilitate the implementation of this programme, the operating model has been taken from the *C. moreletii* monitoring programme for Mexico, Guatemala and Belize. This programme is intended to be implemented from 2019 at the national level, and will be promoted by state authorities, academia, private initiatives and communities. *The IUCN SSC Crocodile Specialist Group (*in litt., *2019) considers that the management programme developed by Mexico for* C. moreletii *is exemplary and demonstrates that Mexico clearly has the capacity to manage wild populations of crocodilians such as* C. acutus.

Export quota or other special measure

Mexico has confirmed its intention to amend the present proposal at CoP18 to include a zero-export quota for wild-sourced specimens (H. Benítez, in litt.*, 2019).*

Additional Information

Threats

In Mexico, habitat modification resulting from tourism developments in coastal areas has been one of the main threats to the species, since it has replaced or modified the sandy beaches where *C. acutus* normally nests. *Habitat loss resulting from tourism developments in the Yucatán peninsula has resulted in population fragmentation with evidence of inbreeding (Machkour-M'Rabet* et al.*, 2009).* In the Pacific, breeding of cattle near crocodile nesting areas has a negative impact by reducing the hatching rate through trampling. A poorly evaluated threat is the use of agrochemicals and spills of industrial and domestic sewage into rivers and coastal areas. In the Yucatán peninsula, hurricanes also pose a threat, as well as genetic introgression with the sympatric species *C. moreletii*.

In some coastal areas of Chiapas, crocodile hunting is practiced for meat and to sell skin illegally, and in some communities they use them as pets. Crocodiles sized 170 cm have also been caught accidentally in gillnets. According to data provided by the Federal Attorney for Environmental Protection (PROFEPA), 49 confiscation events involving a total of 151 items were recorded between 2012 and 2018. *There are records of seizures of live* C. acutus *in Mexico (in one case, in 2018, 225 animals—mostly juveniles—were seized from a wildlife centre that did not have the required legal documentation (PROFEPA, 2018)).*

By 2002 there was estimated to have been a 2.5% decrease in the species' historical distribution area in Mexico, in the Northwest and Central Pacific bio-region. This decline was attributed to the fact that the populations of the states of Sonora and Sinaloa are located in the geographic limits of the species, where the climatic conditions may be unfavourable for populations to be viable.

Conservation, management and legislation

Crocodylus acutus has been classified as Subject to Special Protection since 2001 in the Official Mexican Standard NOM-059-SEMARNAT-2010, which allows its use under certain conditions stipulated in the General Law of Wildlife and its Regulations.

In Mexico there are three legal schemes through which crocodiles are managed: Wildlife Conservation Management Units (UMA), which may be free-ranging or intensive, and Premises and Facilities that Manage Wildlife in Confined Form (PIMVS). The UMA have among their main objectives the conservation of populations and habitat, while the PIMVS undertake controlled reproduction for exclusively commercial purposes. The Secretary of Environment and Natural Resources (SEMARNAT) is in charge of evaluating the management plans and registering the UMA that have free-ranging populations of *C. acutus*. Likewise, it grants licences to the properties that are used for captive management (intensive UMA and PIMVS). It is also responsible for evaluating population studies or inventories to respond to requests for use of the species. The UMA and PIMVS must submit an annual activity report summarising the use that has taken place. The owners of UMA with free-ranging populations of *C. acutus* commit to conserving their habitat and their populations, in exchange for sustainable use that until now has been mostly for ecotourism and scientific purposes. Currently, Mexico has 58 UMAs (15 free-ranging and 43 intensive) and 24 PIMVS for the breeding of *C. acutus*. In the period 2000–2018, eight animals from free-ranging UMA and 39 animals from intensive UMA and PIMVS were authorised for national utilisation of various products including skins, meat and oil.

There are 47 protected natural areas within the distribution of *C. acutus*. In 2018, the National Commission of Natural Protected Areas (CONANP) and the General Directorate of Wildlife of the Secretariat of Environment and Natural Resources (DGVS-SEMARNAT) published the "Programme of Action for the Conservation of Species (PACE): Crocodylia (*Crocodylus acutus*, *Crocodylus moreletii* and *Caiman Crocodilus chiapasius*)" which establishes guidelines for the management and conservation of the three Mexican species.

A Group of Specialists in Crocodilians in Mexico (GEC-Mexico) was created in 2010, made up of experts from different sectors (government, academia, private enterprise, civil society, producers and communities), which is consulted periodically to support decision-making regarding the conservation and sustainable management of Mexican crocodilian species, and will be consulted in making decisions related to *C. acutus* to improve regulation and sustainable management of populations. The present proposal was developed during the GEC meeting held in 2018 with the support of the CITES Authorities of Mexico.

Artificial propagation/captive breeding

See above.

Implementation challenges (including similar species)
An identification guide for *C. acutus* and *C. moreletii* has been developed; features are highlighted to distinguish *C.acutus* from *C. moreletii* and *C. crocodilus*. However, it is known that there is sympatry and historical natural genetic exchange between *C. moreletii* and *C. acutus* in the populations of the Yucatan peninsula. This exchange dates from between 2.4 million years to 230,000 years ago. These sympatric populations do not meet the definition of hybrid specimens according to *Res. Conf. 10.17 (Rev. CoP14)*, since this only applies to hybrids in a recent lineage (last three generations i.e. 120 years for *C. acutus*). *It would be beneficial to understand better the extent of natural hybridisation in the wild population, and how this is going to be handled in trade (IUCN SSC Crocodile Specialist Group,* in litt.*, 2019).*

Although there are studies that are evaluating the possibility that a subpopulation of *C. acutus* on Mexico's Caribbean coast is a distinct species, CITES currently only recognises the existence of a single species.

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Inclusion of Garden Lizards *Calotes nigrilabris* and *Calotes pethiyagodai* in Appendix I

Proponent: Sri Lanka

Summary: The Garden Lizards *Calotes nigrilabris* and *C. pethiyagodai* are members of the agamid lizard genus *Calotes* which comprises some 25 species in total occurring in South and South-east Asia. Both species are endemic to Sri Lanka, where they inhabit the high elevation areas of the Central Highlands and Knuckles Massif.

Calotes nigrilabris is found within the Central Highlands region, primarily within montane and submontane cloud forests mainly at altitudes above 1,400 m, and has an estimated area of occupancy of around 300 km^2, divided between five known sites. *Calotes pethiyagodai,* described in 2014*,* has only been recorded within the Knuckles Massif, at elevations of between 900 m and 1500 m above sea level, with an estimated range of less than 25 km^2.

No estimates of total population sizes are available for either species. In 1988, a density estimate for *C. nigrilabris* of 220 individuals per hectare was obtained and recent observations have suggested the population may be declining.

Both species are reported to be affected by deforestation, the removal of forest understorey to grow cardamom, pesticides, road mortality and the spread of opportunistic predators, although there is little information on the direct impact of these threats.

Both species have been offered for sale at relatively high prices both online and in physical markets in the USA and Europe, but trade instances appear to be low. Both species are protected under Sri Lankan law and harvest and export have been prohibited since 1993. It seems unlikely that all individuals observed for sale are the offspring of animals exported pre-1993 (particularly *Calotes pethiyagodai* which was only described in 2014), therefore it seems probable that wild animals are illegally entering the trade.

Analysis: On the basis of a restricted area of distribution (<25 km^2), with the extent and quality of this habitat in decline, *Calotes pethiyagodai* meets the biological criteria for inclusion in Appendix I. For *C. nigrilabris*, the area of distribution is larger (estimates range from 300 to 500 km^2), but this habitat is also fragmented and likely to be declining. It is possible that it meets the Appendix I biological criteria. In recent years, both species have been offered for sale within the hobbyist trade (although numbers appear low) and illegal collection from the wild is suspected.

It is thought that adults of both species can be differentiated from each other, although it is more difficult with juveniles. As it seems that it is mainly adults in trade, if Parties decide that only one species meets the Appendix I criteria, the other should not necessarily be listed in Appendix II as a look-alike (for which there is no provision in Appendix I).

Other Considerations: An additional six species of *Calotes* occur in Sri Lanka, of which four are also endemic. While identification guides exist, other *Calotes* species could be affected by a shift in harvest pressure were either species proposed here to be included in the Appendices, even though trade in all lizards is already prohibited under national legislation in Sri Lanka. Listing of other species in the genus native to Sri Lanka in Appendix III could also be considered; stipulating a zero export quota with the listing would reflect that export from Sri Lanka is illegal.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS), text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy
The Black-cheek Lizard Calotes nigrilabris *was first described in 1860. Pethiyagoda's Crestless Lizard* Calotes pethiyagoda*, was first described in 2014 (Amarasinghe* et al., *2014). The genus* Calotes*, consists of at least 25 species (Hartmann* et al., *2013; Amarasinghe* et al., *2014).*

Range
Sri Lanka.

IUCN Global Category
Neither of these species has been assessed on the IUCN Red List.

Biological criteria for inclusion in Appendix I

A) Small wild population
Calotes nigrilabris: Since 2012, *C. nigrilabris* has been classified as endangered in the National Red List of Sri Lanka, previously (2007) it was listed as vulnerable. The population is highly fragmented. *Over 20 years ago Erdelen (1988) estimated the density of* C. nigrilabris *to be 220 individuals per hectare. More recent field observations by Karunarathna* et al., *(2011) suggest that the population is in decline. Given the emergence of new threats to this species during the last decades, and the uncertainty surrounding its range and population density, it is not possible accurately to estimate a population size for this species.* It is also recommended that the taxonomy of the isolated population of C. nigrilabris occurring in the Knuckles Massif be verified in the future. If this population represents a new species, the status of *C. nigrilabris* would need to be reassessed.

Calotes pethiyagoda: *Due to its recent description, no estimates of population size or density for this species have been made.* Calotes pethiyagodai *is not listed in either the IUCN Red List or the National Red List of Sri Lanka.* The authors who described the species noted it would meet criterion B2b (iii) *(area of occupancy <10 km^2, and continuing decline in area, extent and/or quality of habitat)* of the IUCN Red List criteria and should be considered endangered.

B) Restricted area of distribution
Calotes nigrilabris: Endemic to Sri Lanka. *This species is found within the Central Highlands region, primarily within montane and submontane cloud forests >1,400 m above sea level (Amarasinghe* et al., *2011; Somaweera & Somaweera, 2009) in Sri Lanka's wet zone.* Calotes nigrilabris *is classified as endangered within the National Red List of Sri Lanka, based on an area of occupancy <500 km^2, in which its habitat is fragmented (MOE, 2012). Bahir and Surasinghe (2005) estimated this species' area of occupancy to be around 300 km^2, divided between five known sites. The largest of these is the Horton Plains, ~2,200 m above sea level.*

Calotes pethiyagodai: Endemic to Sri Lanka, *and has only been recorded within the Knuckles Massif, at elevations of >900 m (Amarasinghe* et al., *2014). Across this mountain range, only 35 peaks exceed 915 m elevation (IUCN, 2010) and, based on observations at six spotting sites, this species is thought to be restricted to an area of occupancy of <25 km^2 and an extent of occurrence of <180 km^2, with the extent and quality of this habitat in decline (Amarasinghe* et al., *2014).*

C) Decline in number of wild individuals
According to personal observations by Karunarathna et al., *(2011) the population size of* Calotes nigrilabris *may be declining.* Calotes pethiyagodai *is known to occur in only a small area, where its habitat is also declining in extent and quality (Amarasinghe* et al., *2014). Without baseline population figures and detailed population studies, the extent of any declines for either species cannot be quantified.*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade:
Both of these species are known to be in the pet trade. The proposal states that some legal exports for commercial purposes took place from Sri Lanka until the mid-1980s *although as* Calotes pethiyagodai *was not identified until 2014 it seems unknown if this species was among those exported*. Since 1993, however, national legislation strictly prohibits any capture, trade and export of agamids from Sri Lanka. *It is therefore conceivable though unlikely that those offered for sale outside Sri Lanka are descendants from this stock, and in recent years, the appearance of adult specimens of* C. nigrilabis *and* C. pethiyagodai *in online advertisements for the international pet trade has raised concerns of wild-caught individuals entering the market (Altherr, 2014; Auliya* et al., *2016; Janssen and de Silva,* in prep.*).*

In Europe, the first documented online advertisement was made in 2011 by a Russian trader, who offered for sale several Sri Lankan agamids, including *Calotes nigrilabris.* Three years later, *C. nigrilabris* was offered by another Russian trader, in connection with "Hamm" Germany, one of the world's largest reptile trade shows. Since then,

C. nigrilabris has also been recorded on European online platforms and in different Facebook groups. The published prices in the adverts for the European pet trade market vary between EUR 100 (*ca. USD 113*) and EUR 250 (*ca. USD 280*) per animal. The majority of the offers are advertised by German traders, but traders from the United Kingdom, Russia, Spain and Italy have also been documented. Although some traders declared their animals as "captive-bred", the high proportion of adult animals involved may indicate illegal capture from the wild. The Supporting Statement claims that *Calotes pethiyagodai* was offered for sale for the first time in November 2016 by a UK trader. While only discovered in 2014, some specimens are already sold as "captive-bred", which may suggest the illegal collection of gravid females from the wild.

In the USA, the U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS) records trade in several Calotes *species since 1999. These records are dominated by imports of large numbers of* C. versicolour*, although in many instances, specimens are only recorded to genus level.* Calotes emma *has also appeared in trade since 2007 and* C. mystaceus *since 2009. Of these species however, only* C. versicolour *occurs in Sri Lanka, with none of these imports originating from Sri Lanka (Viet Nam being a principal exporter of this species). The only* Calotes *specimen originating from Sri Lanka between 1999 and 2013, was a skull of* Calotes calotes*, re-exported from Australia in 2011 for scientific purposes.*

The SS notes that the LEMIS database does not document any trade in either C*alotes nigrilabris* or *C. pethiyagodai*, to 2017. Evidence of online trade of *Calotes nigrilabris* or *C. pethiyagodai* within the USA has, however, been observed. In 2015, a trader from the USA offered two breeding pairs of *C. nigrilabris* for USD 1,000 per pair, highlighting that these were the only specimens in North America. The first offer for *C. pethiyagodai* took place in 2017. The trader stated that he had imported the animals as "captive-bred" from Europe and was now selling one pair from his small group.

*Between September 2016 and October 2018, Janssen and de Silva (*in prep.*), found further evidence of online trading, in the form of 20 online advertisements for* Calotes nigrilabis*, and four for* C. pethiyagodai*, which originated from traders in Germany, Canada and the USA. These adverts all occurred in 2017 and were among nine species observed for sale in this year that were not observed for sale in 2016 or 2018, suggesting that these species are not yet established in trade, possibly indicative of a smuggling event (Janssen & de Silva,* in prep.*).*

The SS also notes that the illegal collection of reptiles often targets gravid females, so that offspring can later be presented as "captive-bred", and that any genuinely captive-bred specimens of recently smuggled adult specimens would be a result of illegally acquired breeding stock.

Additional Information

Threats

Calotes nigrilabris and *C. pethiyagodai* face a range of threats including habitat destruction and fragmentation. Deforestation is the main threat faced by the reptile fauna of Sri Lanka, where natural forest cover has been reduced from 80% to less than 16% over the last 130 years. Other causes of mortality include the use of pesticides, road kills, and exposure to predatory species that opportunistically move in to human habitations, including the Jungle Crow *Corvus macrorhynchos*, Sri Lanka Whistling Thrush *Myophonus blighi*, and feral cats.

Wild harvest and the targeting of gravid females would serve to compound the other threats that are known to affect them. The SS suggests that due to these species' small areas of distribution, habitat specialisation and low reproductive output, offtake of even small numbers of *Calotes pethiyagodai*, especially of mature or gravid females, may severely damage remaining populations.

Conservation, management and legislation

Since 1993, all reptile species of Sri Lanka—except five highly venomous snakes—are protected by law, in accordance with the Section 30 of the 'Seventh amendment to the Fauna and Flora Protection Ordinance (FFPO) of Sri Lanka. This means that collection of these species anywhere in the country, is illegal and export is strictly forbidden.

In 2010, the Central Highlands of Sri Lanka, including the Peak Wilderness Protected Area, the Horton Plains National Park and the Knuckles Conservation Forest, *where these species are known to occur,* were recognised as a World Heritage Site.

Artificial propagation/captive breeding

Little is known about the captive breeding potential of either Calotes nigrilabris *or* C. pethiyagodai*. Observations in the wild by Karunarathna* et al*. (2011) suggest that the breeding season for* C. nigrilabris *occurs in the February–March period of the year, during which time the female lays 3–4 eggs in open areas lacking canopy.* The SS cites a 2017 study in which, based on the maturity age structure, it was assumed that *C. nigrilabris* has

two breeding seasons, although this is not yet confirmed. *Amarasinghe* et al. *(2011) reported successful hatching of* C. nigrilabris *eggs in a carefully controlled captive setting. Captive breeding of* Calotes calotes *is reportedly possible Zoffer (1996), although it is noted that males need particularly large enclosures due to their territoriality, and it is unclear to what extent these techniques could be applied to other members of the genus. Some experts have expressed doubts over the potential for these species to breed well in captivity (J. Wu,* in litt.*, 2019),* while in September 2015, an online advert for *C. nigrilabris* from the USA claimed that there were "very few in the U.S." *and that they had* "never been bred here before".

Implementation challenges (including similar species)
The genus Calotes *is currently represented by some 25 species, eight of which are native to Sri Lanka, and six of which are endemic to Sri Lanka (Somaweera & Somaweera, 2009; Hartmann* et al., *2013; Amarasinghe* et al., *2014). Comprehensive identification guides for each of the Sri Lankan species exist (e.g. Somaweera and Somaweera, 2009; Amarasinghe* et al., *2014).*

Calotes nigrilabris *and* C. pethiyagodai *are said to be distinguishable from one another by a number of morophological differences, including the absence of supratympanic spines in* C. pethiyagodai *(present as a continuous row in* C. nigrilabris*), and the lack of a gular pouch in* C. pethiyagodai *(present in* C. nigrilabris*) (Amarasinghe* et al., *2014).*

As these diagnosable features appear at the subadult stage (K. Ukuwela in litt., *2019), the identification of adults in trade should be possible. While the juveniles of most species of* Calotes *spp. are hard to distinguish (K. Ukuwela* in litt., *2019), they equally are not regularly advertised in trade.*

Potential risk(s) of a listing
As there are three other endemic species of Calotes *within Sri Lanka, there may be potential for any illegal collection to switch to other non-CITES listed species, including* C. liocephalus *(which* C. pethiyagodai *looks most similar to), which also inhabits the Central Highlands and is only known from a small number of high elevation sites (Amarasinghe* et al., *2014).*

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Inclusion of Horned Lizards *Ceratophora* spp. in Appendix I

Proponent: Sri Lanka

Summary: Horned Lizards *Ceratophora* spp. are a genus of small lizards known for their spectacular colouration and distinctive horn-like appendages. The genus is represented by five species, all of which are endemic to Sri Lanka. Limited recent information is available on population size and distribution, but from what is known it appears that the species are generally range restricted and considered threatened:

- *Ceratophora erdeleni* and *C. karu* were both described in 1998 and categorised nationally as critically endangered in 2012, and are restricted to one forest reserve, each with a range estimated at between 10 and 100 km^2. Both have been described as "rare" and the population "small". In 2017, 12 and 10 online adverts respectively were observed.

- *Ceratophora aspera* was assessed on the IUCN Red List as Vulnerable in 2009 due to a continuing decline in quality and extent of its habitat. The extent of occupancy was estimated at 700 km^2 in 2005 and less than 500 km^2 in 2012. Twelve online adverts were observed in 2017.

- *Ceratophora tennentii* was assessed on the IUCN Red List as Endangered in 1998, and nationally as critically endangered in 2012. In 2005 the area of occupancy was estimated to be around 130 km^2 (divided between three known sites) and in 2012 it was suggested it could be as low as 10 km^2. This species is considered to be one of the most common in trade. Forty online advertisements were recorded for this species during 2017–2018, while trade data record the import of 10 specimens into the USA between 2016–2017.

- *Ceratophora stoddartii* is considered nationally to be endangered, with an estimated area of occupancy of 200 km^2. In 2005 it was described as one of the more abundant species in the genus and is also considered to be one of the most common in trade. Fifty-seven online advertisements were recorded during 2017–2018, and 25 specimens imported into the USA between 2013 and 2017.

All species are threatened by continued habitat loss, fragmentation and degradation, with tolerance to habitat disturbance varying between species.

The distinctive appearance of these species makes them sought after by reptile collectors. Exports from Sri Lanka were banned in 1993. In recent years all species have been offered for sale, often at high prices outside of Sri Lanka, and are sometimes reported as being wild-caught.

Due to differences in colouration and morphology, it is said to be possible to distinguish all five species in their adult form, but not as juveniles, and most observed trade appeared to involve adults.

Analysis: All species have been globally or nationally assessed as endangered or critically endangered, except *Ceratophora aspera,* which was assessed as Vulnerable in 2009. Several of the species have a restricted distribution, which is likely to be fragmented, and are declining due to deforestation, and therefore appear to meet the biological criteria for inclusion in Appendix I: *Ceratophora erdeleni, C. karu*, *C. tennentii* and *C. stoddartii.*

In recent years, all species within the genus have been offered for sale within the hobbyist trade, some of which was reported as, or suspected to be, wild-sourced, and therefore illegal.

The area of occupancy for *Ceratophora aspera* has most recently been estimated at less than 500 km^2. In 2009 the habitat was declining in extent and quality and the species was categorised as

Vulnerable. While the decline has likely continued, *C. aspera* is unlikely to meet the biological criteria for inclusion in Appendix I.

It is thought that adults of all species can be differentiated from each other, and although it is more difficult with juveniles it seems that currently it is mainly adults in trade. Therefore, if Parties decide that not all species meet the Appendix I criteria, the other(s) should not necessarily be listed as look-alikes in Appendix II (for which there is no provision in Appendix I).

Other Considerations: Sri Lanka could consider an Appendix-III listing for *Ceratophora aspera*; stipulating a zero export quota with the listing would reflect that export from Sri Lanka is illegal.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS), text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy and IUCN Global Category

There are five species within the genus (Table 1).

Table 1: Species within the genus Ceratophora.

Species name and year of description	Global Red List assessment	National Red List assessment
Ceratophora aspera (1864)	*Vulnerable B1ab(iii) (2009) ver. 3.1*	*Not assessed*
Ceratophora erdeleni (1998)	*Not assessed*	*critically endangered (MOE, 2012).*
Ceratophora karu (1998)	*Not assessed*	*critically endangered (MOE, 2012).*
Ceratophora stoddartii (1834)	*Not assessed*	*endangered (MOE, 2012).*
Ceratophora tennentii (1861)	*Endangered B1+2bc (1996) ver. 2.3.*	*critically endangered (MOE, 2012).*

Range

Sri Lanka.

Biological criteria for inclusion in Appendix I

A) Small wild population

No quantitative data regarding the population size and trends of these species were readily available. Available qualitative information is provided below, although much of it is over a decade old.

Ceratophora aspera *and* C. stoddartii *were the most abundant species and were described as having "appreciable" (large) populations (de Silva* et al., *2005).* The species *C. aspera* was described in 2005 as "uncommon", while *Somaweera and de Silva (2010)* reported it as "the most common horned lizard in the lowlands", *and common in lowland rainforests.*

Ceratophora tennentii has also been described as "uncommon". *In 2015, a field survey recorded an average of 10 individuals per 100 m transect within mixed cardamom forests, eight individuals per 100 m transect in natural forests, four individuals per 100 m transect in cardamom plantations, and no individuals within pine plantations (Somaweera* et al., *2015).*

Ceratophora erdeleni and *C. karu* were both described in 2005 as "rare" and populations of both species as small (although no size was given).

B) Restricted area of distribution
The five species are distributed within a number of specific habitat types and locations within Sri Lanka:

Ceratophora aspera: *Ceratophora aspera* occurs in southwest Sri Lanka, in the lowland wet zone between 60 and 990 m above sea level. This species' extent of occurrence is approximately 10,300 km². Bahir and Surasinghe (2005), estimated its area of occupancy to be 700 km^2*, while the National Red List assessment estimated it to be <500 km^2 (MOE, 2012).* It is thought to be restricted to undisturbed (but fragmented) forests, although it has also been observed in home gardens adjacent to forested areas.

Ceratophora stoddartii: *Ceratophora stoddartii* is found in the cloud forests of Sri Lanka's Central Massif (1,200–2,200 m elevation). The species only inhabits a small amount of its former range, with most of the lower elevations (under 1,800 m) of the Central Massif having been cleared during the past century for the cultivation of tea. *The area of occupancy is thought to be <500 km^2 (Somaweera & de Silva, 2010), and may be as low as 200 km² divided between >10 known sites (Bahir & Surasinghe 2005).* At one of these locations, (Namunukula), this species is restricted to a site of around 200 ha, *with individuals within this isolated subpopulation showing morphometric differences to other subpopulations within the Central Massif (Pethiyagoda & Manamendra-Arachchi, 1998).*

Ceratophora tennentii: *Ceratophora tennentii* is restricted to the Knuckles forest region*, where it is found in natural forests, abandoned cardamom plantations and home gardens, between elevations of 700 to 1200 m (Somaweera & Somaweera, 2009).* Bahir and Surasinghe (2005), estimated its area of occupancy to be around 130 km^2, divided *between three known sites although Somaweera* et al., *(2015), suggested that this may be an overestimate, given the patchy distribution of the species within its range, while the 2012 National Red List assessment of critically endangered suggests its distribution may be as low as <10 km^2.*

Ceratophora erdeleni: *Ceratophora erdeleni* is limited to the Morningside Forest Reserve at elevations of between 1,000–1,300 m. *The National Red List of Sri Lanka categorises this species as critically endangered, based on an extent of occurrence of <100 km^2 (MOE, 2012).* Bahir and Surasinghe (2005), estimated its area of occupancy to be <10 km^2.

Ceratophora karu: *Ceratophora karu* is limited to the Morningside Forest Reserve at elevations of between 1,000–1,300 m. *The National Red List of Sri Lanka categorises this species as critically endangered, based on an extent of occurrence of <100 km^2 (MOE, 2012).* Bahir and Surasinghe (2005), estimated its area of occupancy to be <10 km^2.

C) Decline in number of wild individuals
The population trends for all species are unknown. Long term population declines might be inferred based on the loss and degradation of their habitat (Bahir & Surasinghe, 2005).

In 2005, the National Red List of Sri Lanka classified *Ceratophora aspera* as vulnerable, and *C. tennentii* as endangered. In 2012 they were re-assessed as endangered and critically endangered respectively*. Ceratophora aspera is reported as being "extremely intolerant of habitat disruption", and it therefore seems likely that this species had been heavily impacted by the expansion of agriculture within Sri Lanka (Senanayske (1980), in Somaweera & de Silva, 2010).*

Field observations noted a possible decline in some populations. For example, in a 2011 field survey of the Morningside area, neither *Ceratophora karu* nor *C. erdeleni* were observed. In 2017, a survey of habitat favoured by C*. aspera* in Kithulgala also failed to reveal any individuals, perhaps due to the presence of a nearby development.

Trade criteria for inclusion in Appendix I

International trade in all five species has been documented, with live specimens offered for sale as pets in *Europe, the USA and Asia.*

As regulation of agamid exports from Sri Lanka was less restrictive in the past, *Ceratophora* individuals may have been legally exported. However, the SS suggests that, based on the numbers of adult Sri Lankan agamids that have been observed in the international pet market during the past 15 years, there is a strong indication of smuggling activity. The SS notes that illegal collection of reptiles often targets gravid females, so that offspring can later be presented as "captive-bred", and that any genuinely captive-bred specimens of recently smuggled adult specimens would be a result of illegally acquired breeding stock. It is unknown if this has occurred with any species of *Ceratophora.*

The availability of each species in trade appears to vary:

Ceratophora stoddartii: Of the five species, *Ceratophora stoddartii* is most commonly seen in trade. A survey of European reptile fairs in 1998 found specimens of *C. stoddartii* among the 15 most expensive non-CITES listed species on sale, priced at the equivalent of EUR 176 (*ca. USD 200*). By 2014 prices had increased to EUR 2,000–2,500/pair (USD 2,250-2,800/pair). Since then observed prices appear to have decreased to around EUR 750–1,200/pair (USD 850–1,360/pair). *In a study of online trade platforms, between 2016 and 2018, Janssen and de Silva (*in prep.*) reported average prices of around EUR 312/specimen (ca. USD 350) for this species.*

The SS states that the first documented online offer for *Ceratophora stoddartii*, was on a European online platform in January 2011. Since mid-2013 regular advertisements have been observed on European online pet trade websites and in Facebook groups. For example, in 2013 at least three Russian and one German trader advertised several species of Sri Lankan agamids, including *C. stoddartii.* Since then, similar offers have been made by Swiss, French, Russian, Italian, British, Czech and Spanish nationals. *In a survey of online advertisements conducted between 2016 and 2018, Janssen and de Silva (*in prep.*) documented 57 adverts for* C. stoddartii *in Denmark, Slovakia, the USA and Canada. The majority of these specimens were advertised as captive-bred, although in some cases no specific origin was mentioned, and in three cases, wild origin was reported. Forty-four of these advertisements were documented in 2017, and 13 in 2018, with this spike in adverts potentially indicative of a smuggling event (Janssen & de Silva,* in prep.*).*

Ceratophora stoddartii was also found in trade within Asia and was recorded in a 2004–2005 survey of pet shops in Taiwan, Province of China. In 2013, adult specimens were offered for sale by a Japanese national in a Facebook group.

The U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS) data show that between 2013 and 2017, the USA imported 25 live specimens of *C. stoddartii*. Of these, three were declared as wild and 22 as captive-bred. In 2014 an Italian trader offered adult *C. stoddartii* for delivery to the USA for USD 1,250 a pair and, since then, several US nationals have offered this species for sale.

Ceratophora tennentii: Ceratophora tennentii *appears to be one of the most commonly traded* Ceratophora *species.* In 2014, several offers for *C. tennentii* were observed, at prices of USD 850/pair. In 2014, a trader from Malaysia offered specimens of *C. tennentii* (along with specimens of *C. stoddartii*) on a European online platform. In 2016, eight captive-bred specimens of *C. tennentii* were imported into the USA, followed by two individuals in 2017, which were declared as wild-caught. All specimens imported into the USA were for commercial purposes and were exported from Germany and Poland. *In 2017, Janssen and de Silva (*in prep.*), documented 28 online adverts for* C. tennentii *in Germany, the Czech Republic, Spain, Slovakia and the US, which was followed by 12 adverts in 2018. The authors of this study suggest that the changes in the volume of these adverts in different years, points to a smuggling event.*

Ceratophora aspera: Since December 2014, several online offers for *Ceratophora aspera* have been documented, at prices of up to USD 2,850/pair. *In 2017, Janssen and de Silva (*in prep.*) documented 12 online adverts for* C. aspera *in Germany, with no adverts recorded the following year.*

***Ceratophora erdeleni*:** In August 2017, a trader *described as being* from Malaysia (*although the SS does not indicate which country this trader was based in),* advertised two pairs of *Ceratophora erdeleni*, *which the SS claims is the first time this species was offered for sale online. In 2017, Janssen and de Silva (*in prep.*) documented 12 online adverts for* C. erdeleni *in Germany and Malaysia, with no adverts recorded the following year.*

Ceratophora karu: In August 2017, *the same trader from Malaysia noted above* offered a pair of *Ceratophora karu* for sale, *which the SS claims is the first time this species was offered for sale online. In 2017, Janssen and de Silva (*in prep.*) documented 10 online adverts for* C. karu *in Germany and Malaysia, with no adverts recorded the following year.*

In addition to the trade data noted above, the U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS) data also show the import of two wild-caught specimens of *Ceratophora* spp. in 2009.

Additional Information

Threats

Deforestation is the main threat faced by the reptile fauna of Sri Lanka, where natural forest cover has been reduced from 80% to less than 16% in the past 130 years. Habitat loss, fragmentation and degradation are

considered the most serious threats to this genus. Severe deforestation has occurred in Sri Lanka due to the clearing of land for agricultural purposes, conversion to plantations for crops such as tea, mining, logging and pressures associated with expanding human settlements. In 2005, it was estimated that only 5% of the island's original wet zone forest remained intact. *Bahir and Surasinghe (2005), also cite rainwater acidification, pesticides and the effects of climate change as further principal threats.*

Somaweera et al., (2015*) found* Ceratophora tennentii *to be adaptable to disturbed habitats (particularly mixed cardamom forests), provided these retain forest canopy cover and vegetation of sufficient structural complexity. Other members of the genus, however, may be less tolerant to habitat degradation.* For example, C. aspera is *thought to be "extremely intolerant of habitat disruption" and "disappears when primary rain forest cover is lost" (Senanayske (1980), in Somaweera & de Silva, 2010).*

It has been reported that some farmers may kill C. tennentii *under the misapprehension that they damage cardamom crops (Somaweera,* et al., *2015).*

The impact of any illegal collection of Ceratophora *spp. for the international pet trade is unknown. However, any illegal offtake may compound the threats identified above, particularly for* C. edelini *and* C. karu*, which are considered rare and occupy a very small range.* The SS describes the reproductive rate of *Ceratophora spp.* to be low. Clutch size is between one and ten eggs and varies between species. Females of *C. aspera* have been observed as depositing only one to two eggs per clutch, a clutch size of two eggs is reported for *C. karu*, while *C. stoddartii* produces up to eight eggs per clutch. Sexual maturity is reached at an age of six months. Ceratophora stoddartii, C. tennentii *and* C. aspera *are also noted as being slow moving species (Somaweera & Somaweera, 2009), which may make them more susceptible to illegal collection if this occurs on an opportunistic basis (P. Bowles,* in litt., *2019).*

Conservation, management and legislation

The range of some species partially falls within a number of protected areas. For example, *C. aspera* is known to occur in the Sinharaja Natural Heritage Wilderness Area. However, even here, illegal logging, mining and human encroachment remain a threat. *Bahir and Surasinghe (2005) note that much of the range of* C. tennentii *does not have protected status and is not subject to conservation management.*

Since 1993, all trade in *Ceratophora*, (including wild collection, ranching and breeding), has been strictly prohibited*. There are concerns that adult specimens recently seen in trade may be wild-caught (and therefore illegal), rather than originating from long-held captive stock (Auliya* et al., *2016; Altherr, 2014; Janssen & de Silva, 2019; J. Wu,* in litt., *2019).*

Artificial propagation/captive breeding

The majority of specimens in trade are described as captive-bred (Janssen and de Silva, in prep.). Limited anecdotal reports of captive breeding exist (for example within German magazine articles dating from 2007 featuring captive breeding of C. stoddartii*, (K. Ukuwela,* in litt., *2019)), but* overall, accounts of successful captive breeding are scarce. *In a scientific study, Kravac* et al., *(2015) attempted to breed* C. aspera *in carefully controlled conditions, with some success, noting that captive breeding techniques still require development as a "tool of last resort" for the conservation of the species. Many experts doubt that individuals observed for sale online were legally acquired prior to 1993 (Altherr, 2014; Auliya* et al., *2016; P. Bowles,* in litt., *2019; J. Wu,* in litt., *2019; Janssen & de Silva, (*in prep*)).*

Implementation challenges (including similar species)

All five species within the genus can be identified in adult form (P. Bowles, in litt., *2019; K. Ukuwela,* in litt., *2019). Among other distinguishing features, adults display distinctive rostral appendages ("horns"), which are larger in males than females and differ in appearance between species (Somaweera & Somaweera, 2009). Horns are absent in juveniles which are not easily distinguished, (K. Ukuwela,* in litt., *2019), but equally are not regularly advertised in trade.*

Only one other genus within Agamidae, Harpesaurus*, contains a prominent horn-like appendage, although their structure is fundamentally different to the "horns" seen in* Ceratophora *(Somaweera & Somaweera, 2009).*

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Inclusion of Pygmy Lizards *Cophotis ceylanica* and *Cophotis dumbara* in Appendix I

Proponent: Sri Lanka

Summary: The Pygmy Lizards *Cophotis ceylanica* and *C. dumbara* are small lizards endemic to the high altitude regions of central Sri Lanka and the only members of their genus. Both species are threatened by a number of factors including habitat loss and fragmentation: timber extraction and clearing of forest for tea plantations have already destroyed large areas of habitat.

Cophotis ceylanica was classified in 1998 as endangered nationally. This species has a restricted range with an area of occupancy estimated at less than 500 km^2 and perhaps as low as 60 km^2. The species was considered "rare" in 2005. *Cophotis ceylanica* was estimated to have decreased by more than 50% in the decade prior to 1998, and was predicted to decline further, although the threats identified at the time did not include over-collection. Die-offs of hundreds of individuals occurred due to drought in the 1990s. In 2017 and 2018, 69 online adverts were observed in Europe and the USA; most individuals were described as captive-bred.

Cophotis dumbara was described in 2006 and classified as Critically Endangered on the IUCN Red List in 2008. The population size is not known, nor is the current trend, but its habitat is severely fragmented and its area of occupancy is thought to be less than 10 km^2. Eight online adverts were found in Germany and the USA for *C. dumbara* in 2017 to 2018.

There is evidence of trade. Both species have been offered for sale at high prices, particularly within the USA and Europe, but trade instances appear to be relatively low. Both species are protected under Sri Lankan law, and harvest and export have been prohibited since 1993. It seems unlikely that all individuals observed for sale are the offspring of animals exported pre-1993 (particularly *C. dumbara*, which was only described in 2006), therefore it seems probable that wild animals are illegally entering trade.

It is considered that adults of the two species can be distinguished by finer taxonomic details, such as the number of spines and the appearance of scales in certain regions of the body, and although juveniles are very difficult to tell apart it appears that most current trade is in adults.

Analysis: On the basis of a restricted area of distribution (10 km^2) that is declining and fragmented, *C. dumbara* meets the biological criteria for inclusion in Appendix I. For *C. ceylanica*, the area of distribution is larger (<500 km^2 but perhaps as low as 60 km^2), which is likely to be declining and fragmented. The species was said to be rare in 2005, apparently having undergone a marked decline in the 1990s, and is highly vulnerable to extrinsic factors such as drought. Therefore, it is possible that *C. ceylanica* also meets the biological criteria for inclusion in Appendix I. In recent years, both species have been offered for sale (although numbers appear relatively low) within the hobbyist trade and illegal collection from the wild is suspected.

Other Considerations: *Res. Conf. 12.10* (*Rev. CoP15*) outlines that inclusion in Appendix I would mean commercial captive breeding operations would need to meet the provisions of *Res. Conf. 10.16 (Rev.)* to be registered with the CITES Secretariat, and that registered operations should ensure an appropriate and secure marking system to identify all breeding stock and specimens in trade. This enhanced oversight could help allay any concerns over fraudulent claims of captive breeding and wild offtake for breeding stock.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS), text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy
Until recently, the genus was thought to comprise a single species, Cophotis ceylanica*. In 2006 a second species, which is geographically isolated from* C. ceylanica*, was described. This was named* C. dumbara*, after the Dumbara Range (known as the Knuckles Hills), which is the only region within Sri Lanka where it is known to occur (Manamendra-Arachchi* et al., *2006; Samarawickrama* et al., *2006).*

Range
Sri Lanka.

IUCN Global Category
Cophotis dumbara: Critically Endangered B1ab(iii) (2008) ver 3.1.

Biological criteria for inclusion in Appendix I

A) Small wild population
Cophotis ceylanica *has not been assessed by IUCN.* Nationally assessed as endangered since 1998. There do not appear to be any population estimates for *C. ceylanica*, although Bahir and Surasinghe (2005) described this species as "rare".

Cophotis dumbara *has rarely been observed, and there is little information available on the abundance of this species (Samarawickrama* et al., *2009).*

B) Restricted area of distribution
Cophotis ceylanica *is restricted to tropical montane cloud forests at elevations in the Central Highlands of Sri Lanka, >1,700 m above sea level (Manamendra-Arachchi* et al., *2006). It was nationally assessed as endangered, based on an area of occupancy of less than 500 km², with this habitat declining in extent and/or quality (MOE, 2012). Bahir and Surasinghe (2005), estimated the area of occupancy to be even smaller at 60 km^2, and considered the species to be restricted to four known sites. Somaweera and Somaweera (2009), list seven known locations for this species.*

Cophotis dumbara *is only known from the Knuckles Mountains (Dumbara Hills) of central Sri Lanka, where it is restricted to cardamom plantations (Somaweera and Somaweera, 2009), at elevations of between 1,000–1,550 m (Samarawickrama* et al., *2009). The species' extent of occurrence is thought to be <100 km^2, and its distribution within this severely fragmented (Samarawickrama* et al., *2009). The National Red List of Sri Lanka considers its area of occupancy to be <10 km^2 (MOE, 2012).*

C) Decline in number of wild individuals
Recent accurate data on the current population trends of *Cophotis ceylanica* and *C. dumbara* are not available. The severe decline in certain types of habitat favoured by the two species is assumed to mean that their populations are decreasing.

Bahir and Surasinghe (2005) describe Cophotis ceylanica as "rare", and the population is said to have undergone declines in the past. *In 1992, and 1994–1995, hundreds of dead specimens were found within a few days at Hakgala and Nuwara Eliya, with this mortality caused by an extended drought (Somaweera & Somaweera, 2009).* By 1998 the population was thought to have declined by more than 50% within the previous decade, and future declines were predicted.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade:
Both of these species are known to be in the pet trade. Since 1993 national legislation has prohibited any capture, trade and export of agamids from Sri Lanka. The proposal states that some legal exports for commercial purposes took place from Sri Lanka until the mid-1980s. *It is conceivable though unlikely that recent trade outside of Sri Lanka is of descendants from this pre-1993 stock, raising concerns about the illegal capture of* Cophotis *spp. from the wild to supply these markets (see Auliya* et al., *2016).*

The SS also notes that the illegal collection of reptiles often targets gravid females, so that offspring can later be presented as "captive-bred", and that any genuinely captive-bred specimens of recently smuggled adult specimens would be a result of illegally acquired breeding stock.

Cophotis spp. are slow-moving animals, which makes them particularly easy to capture, and *C. dumbara* can be easily collected during seasonal clearing of cardamom plantations. *Of the two species,* C. ceylanica *appears to be most frequently offered for sale,* but specimens of *C. dumbara* have also been advertised in trade since at least 2015. *The trade appears to be focused on markets in Europe and the USA:*

***Europe*:** The first documented online advertisement for *Cophotis ceylanica* was in 2013, for sale in "Hamm", Germany, one of the largest reptile trade shows worldwide. Since that time, regular advertisements have been found on European online platforms: the prices vary between EUR 280 to 600 *(USD 317 to 680)* per male, while adult breeding pairs were sold for up to EUR 1,500 (USD 1,701). The majority of these offers are advertised within Europe. In 2015, *C. dumbara* was also offered for sale in Europe.

USA: *LEMIS data show that in 2015, five live specimens of* Cophotis ceylanica *were imported into the USA from Poland, reportedly captive-bred (U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS), 2015).* Both species have also been seen in recent advertisements by traders in the USA. *Cophotis ceylanica* (first seen in online adverts in 2015) was offered for USD 1,350/pair, while males were offered for USD350–500. In 2018, two adult pairs of *C. dumbara* were advertised in the USA for USD 2,250, with the trader highlighting that "this species was discovered in 2006 and is very rare and critically endangered", claiming these animals would be "the first ones ever imported into the USA".

Asia: In 2014, a trader from Malaysia advertised specimens of a variety of Sri Lankan endemic agamids, including *Cophotis ceylanica,* on a European online platform.

Further evidence of international trade is also provided by a recent study of online advertisements. In this study, Janssen and de Silva (unpubl.) found Cophotis ceylanica *offered for sale in 69 online adverts. Of these, 52 were posted in 2017, followed by a further 17 in 2018, originating from Germany, the USA, France, Austria, Poland and Slovakia. The same study also found eight online advertisements for* C. dumbara*, two of which occurred in 2017, and six in 2018, with the adverts originating from Germany and the USA. The majority of specimens were described as captive-bred, although two pygmy lizards (*Cophotis *spp.) were described as F1, suggesting at least one of the parents was taken from the wild. Of 17 Sri Lankan species offered for sale, the authors found* C. dumbara *to be the most expensive, offered at prices of EUR 1,443 (USD 1,667). Auliya* et al., *(2016), has also documented* C. ceylanica *for sale at high prices of EUR 2,200/pair (USD 2,495). Bowles (in litt., 2019) notes that as these prices, within Western markets, are lower for* Cophotis *spp. than for other Sri Lankan agamids such as* Lyriocephalus scutatus*, this may increase their appeal to a broader section of the hobbyist community.*

Additional Information

Threats

Deforestation, leading to loss of habitat and habitat fragmentation, are the main threats faced by the reptile fauna of Sri Lanka. Timber extraction and clearing of forest for tea plantations has already destroyed large areas, and Sri Lanka's natural forest cover has been reduced from 80% to less than 16% over the last 130 years. Given the likelihood of ongoing deforestation, it is feared that by the end of 2030, less than 10% of forest cover will remain in Sri Lanka.

Areas of forest habitat suitable for C. dumbara *are also currently being lost through die-back, and from fires set to clear land for agricultural use (Samarawickrama* et al., *2009). Additionally, when the understorey of the cardamom plantations they inhabit is seasonally cleared, this increases the threat of predation (Samarawickrama* et al., *2009).*

While the extent of any illegal collection of Cophotis *spp. from the wild is currently unknown,* the SS raises concerns that any offtake, even of small numbers, may have a significant impact on their remaining populations, particularly for *C. dumbara* given its restricted area of distribution.

Conservation, management and legislation

Prior to 1993, some legal exports of endemic reptiles from Sri Lanka for commercial purposes were permitted. Since 1993, however, all reptile species of Sri Lanka—except five highly venomous snakes—are protected by law, in accordance with the Section 30 of the "Seventh amendment to the Fauna and Flora Protection Ordinance (FFPO) of Sri Lanka". This means that collection of these species anywhere in the country is illegal. A permit issued by the Department of Wildlife Conservation is mandatory to perform any ex-situ or in-situ activity that involves a protected reptile species. Ranching and breeding of reptile species is also forbidden.
In 2010, the Central Highlands of Sri Lanka, including the Peak Wilderness Protected Area, the Horton Plains National Park and the Knuckles Conservation Forest, were recognised as a World Heritage Site. Populations of both species of *Cophotis* lie partially within this conservation area.

Artificial propagation/captive breeding

The extent to which Cophotis *spp. can be successfully bred in captivity is unclear. Some online adverts offer photographic evidence of juvenile specimens,* but reptile smugglers are believed to collect gravid females, *and therefore this does not provide conclusive proof that all captive specimens originate from animals collected prior*

to 1993. The smuggling of endemic reptiles from Sri Lanka is suspected to occur (see SS and Auliya et al., *2016), and some experts raise doubts that* Cophotis *spp. could be successfully bred in captivity (J. Wu,* in litt., *2019). It is therefore considered to be unlikely that all specimens seen in trade have been produced from legally acquired captive breeding stock.*

Implementation challenges (including similar species)

The genus *Cophotis* can be distinguished from other agamid genera by the presence of a fleshy bulb on its snout, along with a number of other physical features. *To non-experts,* Cophitis ceylanica *and* C. dumbara *are very similar in appearance but can be differentiated on the basis of finer taxonomic details including: the number of spines in the nuchal crest (two in* C. dumbara *and three in* C. ceylanica *(K. Ukuwela,* in litt., *2019); the scales on the gular sac (smooth in* C. dumbara *and rough in* C. ceylanica*), and size of the gular sac (less well developed in* C. dumbara*) (Manamendra-Arachachi* et al., *2006; T. Amarasinghe,* in litt., *2019). A high-quality photograph of this region would seem likely sufficient for a specialist to tell the species apart if these were supplied by monitoring officials (P. Bowles,* in litt., *2019). The juveniles of both species are not easily distinguished, (K. Ukuwela* in litt., *2019), but equally are not regularly advertised in trade.*

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Wu, J. (2019). *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

Inclusion of the Hump-nosed Lizard *Lyriocephalus scutatus* in Appendix I

Proponent: Sri Lanka

Summary: The Hump-nosed Lizard *Lyriocephalus scutatus* is a medium-sized lizard with a striking appearance. It is the largest of Sri Lanka's endemic agamid lizards and is the only member of the genus. It occurs in the south-west of the country in an area of somewhat less than 17,000 km^2. It is found in a variety of habitats, including forests, plantations and gardens from 25 m to 1600 m above sea level. Individuals reach sexual maturity within a year and a female may produce up to 30 eggs a year.

Population size and trends are unknown. Loss and fragmentation of natural habitat has been severe within its range, one population was nearly extirpated due to intensive logging, although there is evidence that the species can adapt to modified habitats. The species was assessed on the IUCN Red List in 2009 as Near Threatened and considered "not rare" within its range. Collection for the pet trade was said to be reducing population numbers.

The distinctive appearance of the species makes it sought after by reptile collectors. According to one estimate, around 500 specimens were collected from the wild for export in the past 30 years. Export from Sri Lanka has been prohibited since 1993, although the species has been offered for sale at high prices outside of the range State. Given reported challenges in captive breeding it is thought unlikely these animals are the progeny of individuals imported pre-1993.

Analysis: The species is in demand for the hobbyist trade within the EU, USA and Asia, with illegal collection from the wild suspected, although the overall volume of trade is not known. The population size of *Lyriocephalus scutatus* is unknown but unlikely to be small. It was not considered to be rare within its range ten years ago, and despite severe and ongoing deforestation within its range, it does not have a restricted distribution. While at least one population has reportedly been almost extirpated due to logging, and collection for the pet trade is said to be causing a decline, it is unclear if the species overall has undergone a marked decline. Therefore, there is insufficient information to determine if the species meets the criteria for inclusion in Appendix I.

Other Considerations: Sri Lanka could consider an Appendix III listing. In this case stipulating a zero-export quota with the listing would reflect that export from Sri Lanka is illegal.

Res. Conf. 12.10 (*Rev. CoP15*) outlines that inclusion in Appendix I would mean commercial captive breeding operations would need to meet the provisions of *Res. Conf. 10.16 (Rev.)* to be registered with the CITES Secretariat, and that registered operations shall ensure an appropriate and secure marking system to identify all breeding stock and specimens in trade. This enhanced oversight could help allay any concerns over fraudulent claims of captive breeding and wild offtake for breeding stock.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS), text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy
Described in 1758, the genus is monotypic.

Range
Sri Lanka.

IUCN Global Category
Near Threatened (2009) ver. 3.1.

Biological criteria for inclusion in Appendix

A) Small wild population

While population studies are highly localised and no long-term monitoring has been carried out to date, anecdotal and opportunistic observations suggest that *Lyriocephalus scutatus* is not uncommon within its range. *According to the IUCN Red List assessment undertaken in 2009, the species was considered "uncommon", and in Kosgama lowland forest the population is almost zero due to intensive logging activities (Somaweera & de Silva, 2010).*

In 2009, Lyriocephalus scutatus *was assessed as Near Threatened on the IUCN Red List. The Red List assessment states that its distribution is not severely fragmented, it occurs in more than 10 locations, and is not considered to be rare within its range, and therefore does not qualify for a higher threatened category (Somaweera & de Silva, 2010).*

B) Restricted area of distribution

Lyriocephalus scutatus is endemic to Sri Lanka. *Widespread in the wet lowlands and mid-hill areas to the south-west of the country, it is found* in cool and shady forested areas, plantations and gardens, *at elevations from ~25 m to 1,600 m above sea level (Somaweera & Somaweera, 2009).*

The IUCN Red List estimates its extent of occurrence as <17,400 km^2. This agrees with a classification of vulnerable in the National Red List of Sri Lanka, based on an extent of occurrence of <20,000 km^2 (MOE, 2012). While it is an island endemic, it occurs in more than 10 locations, its distribution is not severely fragmented (Somaweera, & de Silva, 2010), and it inhabits a number of habitat types at both high and low elevations (Somaweera & Somaweera, 2009).

C) Decline in number of wild individuals

No population monitoring has been conducted for this species, although collection for the pet trade is reducing population numbers and it has experienced severe and ongoing deforestation and subsequent habitat loss due to human activities (Somaweera & de Silva, 2010).

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade:

Some legal exports for commercial purposes took place from Sri Lanka until the mid-1980s, *at a time when the export laws were apparently more relaxed (Auliya* et al., *2016).* Since 1993, however, national legislation prohibits any capture, trade and export of agamids from Sri Lanka*. Despite this protection, it has been estimated that during the previous three decades, around 500 specimens have been collected from the wild (Somaweera & de Silva, 2010). Karunarathna and Amarasinghe (2013), also suggest that illegal smuggling has occurred over the past few decades.*

Lyriocephalus scutatus possesses features which are known to be highly desirable in the pet industry. This demand in the international pet trade is reflected in prices ranging from USD 1,600 to USD 5,500 a pair.

Since at least 2011, a growing number of *Lyriocephalus scutatus* specimens have been documented within the European, USA and Asian pet markets. In Europe, in 2013 and 2014, *L. scutatus* were being offered for up to USD 3,400/pair. *In 2017, Janssen and de Silva (*in prep.*), recorded six online advertisements for this species in Germany and the Czech Republic, and a further 19 the following year. The 2017 adverts coincided with advertisements for a number of other endemic Sri Lankan species that were not observed for sale in either 2016 or 2018, possibly indicating a smuggling event.*

Lyriocephalus scutatus specimens have been on sale in the USA pet market since at least 2015. Adult breeding pairs are sold for between USD 2,200 and USD 5,500, and the species praised as “a truly rare reptile in today’s hobby” and “very rare, almost never offered for sale”. In 2016, a USA trader advertised a number of “captive-bred” Sri Lankan endemic agamid species, including *L. scutatus*. According to U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS) data for 2000–2017, no *L. scutatus* specimens were traded until 2016–2017, when ten live individuals were imported for commercial purposes. Of these, two were declared as wild and eight as captive-bred, and were imported from the Netherlands, Germany and Canada. Nine live individuals were exported during 2016–2017; four to the Netherlands and five to South Korea. All nine were declared as captive-bred and traded for commercial purposes.

In Asia, *the species is reportedly a kept as a pet species in China, although the origin of these specimens is not known (L. Chou,* in litt., *2019).* There are also anecdotal reports of this species being traded in Taiwan, Province of China, Japan, and Malaysia. In 2004 and 2005, *Lyriocephalus scutatus* was recorded on sale in pet shops in southern Taiwan, Province of China. In May 2011, a Japanese trader offered five specimens for export,

underlining their rarity ("very few been in trade before"), and other examples of the online sale of adult specimens has also been noted in Japan.

Although it is impossible to rule out that these animals in trade are offspring from specimens exported before national legislation became more restrictive, *it is thought unlikely that breeding stock originating from this time is being exclusively used to supply the pet market*—wild sourcing is strongly indicated for some animals in trade, based on *customs data that declares some specimens as wild (as noted above),* and descriptions in adverts, such as one from a trader in Poland who noted that specimens for sale had been "8 months in captivity". Many advertisements also offer "adult specimens", *rather than juveniles, which may also indicate illegal capture from the wild (Wakao* et al., *2018). The sporadic appearance of online advertisements for Lyriocephalus scutatus and other Sri Lankan species in some years but not others may also be indicative of smuggling events (Janssen & de Silva,* in prep.*).* The SS notes that the illegal collection of reptiles often targets gravid females, so that offspring can later be presented as "captive-bred", and that any genuinely captive-bred specimens of recently smuggled adult specimens would be a result of illegally acquired breeding stock.

Additional Information

Threats

Deforestation is the main threat faced by the reptile fauna of Sri Lanka, in 2007 only 17% forest coverage was recorded. The forests of the wet zone and the central hill range, key habitat for *Lyriocephalus scutatus,* are highly fragmented. In addition, infrastructure developments are likely to cause further fragmentation of *L. scutatus* habitat in the future. *Male L. scutatus show a high degree of site fidelity (Bandara (2012), which may make them more vulnerable to habitat loss and more vulnerable to over-collection.*

Other threats include: mortality related to man-made forest fires; application of agrochemicals; road kills; and predation by farm and domestic animals such as cats and poultry.

Conservation, management and legislation

Since 1993, all reptile species of Sri Lanka—except five highly venomous snakes—are protected by law, in accordance with Section 30 of the "Seventh amendment to the Fauna and Flora Protection Ordinance" (FFPO) of Sri Lanka. This means that collection of these species anywhere in the country is illegal.

This species occurs in protected areas, including the Sinharaja Natural Heritage Wilderness Area, although illegal logging, mining and human encroachment remain a threat even within this protected area. In 2010, the Central Highlands of Sri Lanka, including the Peak Wilderness Protected Area, the Horton Plains National Park and the Knuckles Conservation Forest, were recognised as a World Heritage Site.

Artificial Propagation/captive breeding

In the wild, females mature at 10–12 months and lay up to 10 eggs, up to three times a year. While there have been some legal exports of agamids from Sri Lanka until the mid-1980s, there is no recent reliable information on the success of past or current captive breeding. *Anecdotal reports of captive breeding exist (for example within magazine articles dating from the 1970s and 1980s), but the extent of any successful breeding is unclear (K. Ukuwela,* in litt., *2019).* A pet keeper's resource notes: issues with lighting and feeding in captivity; that the species "was thought to be keepable but not breedable"; and that "captive-bred animals are available from time to time". *Zoffer (1996), states that this species has a "cult following in the herpetocultural hobby and would probably be more popular if it was available more often" and adds that this species' specialised feeding habits present a difficulty in keeping them.* O*ther experts have also expressed doubts over the likely ability of this species to breed well in captivity (J. Wu,* in litt., *2019).*

Implementation challenges (including similar species)

Lyriocephalus scutatus is comparatively easy to identify *in adult form, which is the form in which it appears to be predominantly traded.* The species has a distinctive rostral knob, which is more developed in males than in females, and which makes the species easy to discriminate from other agamids such as the *Ceratophora* genus.

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Inclusion of Leopard Geckos *Goniurosaurus* spp. (populations of China and Viet Nam) in Appendix II

Proponent: China, European Union and Viet Nam

Summary: *Goniurosaurus* is a genus of lizards comprising 19 species; 13 of which are native to China and/or Viet Nam (and are subject to this proposal) and six of which are endemic to Japan (not subject to this proposal). Eleven of the 13 species occurring in China and Viet Nam were described from 1999 onwards.

Very little is known about the ecology of most of the species, although they appear to show a high degree of adaptation to specific microhabitats and most are considered to have limited ranges, with many only known from a single mountain range or island. In general, the species are nocturnal, associated with rock/karst-type topography, and are found near to streams and within primary rainforest. Reproductive capacity is likely to be low, males are thought to reach maturity at around one year of age and females lay clutches of two to three eggs annually.

Population estimates are lacking for almost all species. Published IUCN Red List assessments are available for three of the species: one Critically Endangered (*G. huuliensis*), one Endangered (*G. catbaensis*) and one Vulnerable (*G. lichtenfelderi*). A further five species have assessments accepted for publication in March 2019: one Critically Endangered (*G. yingdeensis*), three Endangered (*G. bawanglingensis, G. liboensis* and *G. zhelongi*), and one Data Deficient (*G. zhoui*).

Due to their attractive appearance and colour patterns, many species (ten out of 13) are known to be traded internationally as pets, with Europe, Japan and the USA identified as key markets. Comprehensive trade data are lacking, but imports reported by the USA from 1999-2018 totalled nearly 17,000 individuals with 70% of imports reported as wild sourced. Only three species were not reported in global trade and they are three of the most recently described species *G. kwangsiensis*, *G. liboensis* and *G. zhoui*. However, it is believed that recently discovered species may be particularly vulnerable to exploitation. Local extirpations have been recorded for three species to feed demand from the pet trade, and newly described species are often advertised at higher prices. The most highly traded species based on available data are *G. lichtenfelderi, G. hainanensis and G. luii;* trade in *G. catbaensis* may also be significant (see below). There also appears to be harvest for the domestic pet trade and medicinal purposes.

Goniurosaurus catbaensis is endemic to Cat Ba Island, Viet Nam. Described in 2008, the species was assessed as globally Endangered in 2016 and has an estimated area of occupancy of 120 km^2 (which is declining) and a severely fragmented population which may be less than 250 mature individuals. Habitat destruction is a major threat, and a flood in 2015 appears to have caused local extirpations. *G. catbaensis* was the species most commonly advertised of seven Vietnamese endemic reptiles observed for sale online in Europe and Japan, and is frequently observed in pet shops in southern Viet Nam where they are reported to have been collected from the wild.

Goniurosaurus luii is native to northern Viet Nam and western Guangxi, China. No estimates of the total population size are available. The species was subject to over-exploitation for commercial sale before it was described in 1999, which has led to extirpation from its type locality. Some trade has been reported to the USA (from 1999-2018 ca. 600 were imported) and it has been observed in pet shops in southern Viet Nam (believed to be collected from the wild) and in Japan. Occasionally *G. luii* is harvested from the wild in China for use in traditional medicine.

Goniurosaurus lichtenfelderi is known from three provinces and one archipelago in Viet Nam and was assessed as globally Vulnerable in 2017. Over 7,000 individuals were reportedly imported into the USA from 1999-2018, accounting for 44% of the USA's reported imports of *Goniurosaurus*. Trade in *G. lichtenfelderi* has also been reported in Europe, Japan and locally in Viet Nam.

In addition to over-harvest, major threats to *Goniurosaurus* include habitat loss, tourism and exceptional weather events. In China, collection of certain species in the genus is prohibited under national legislation, while collection of the remaining species is subject to quotas and the issuance of a permit. *Goniurosaurus* species are not currently protected in Viet Nam, but a proposal to include these species in national legislation to regulate international trade will be considered in 2019 (and will be automatic if the genus is included in Appendix II). Some of the species have part of their range within protected areas.

In terms of species identification, there are features that distinguish the six *Goniurosaurus* species from Japan from the Chinese and Vietnamese species, but this is reportedly difficult for non-experts. Similarly, the Vietnamese and Chinese species can be difficult to distinguish from each other without genetic analyses, particularly if the geographic origin is not accurately known.

Analysis: Thirteen species of *Goniurosaurus* lizards are native to China and/or Viet Nam. Most species in the genus show a high degree of habitat specificity and have a very limited range, and the majority have been observed in international trade. *Goniurosaurus catbaensis* meets the criteria for inclusion in Appendix II in Annex 2aA of *Res. Conf. 9.24 (Rev. CoP17)*, and may already meet the criteria for inclusion in Appendix I, given its small (or very small) populations, restricted range with fragmented and decreasing habitat. *Goniurosaurus lichtenfelderi* and *G. luii* also meet the criteria on the basis of large trade volumes, local extirpations and/or small populations.

Species of *Goniurosaurus* can be difficult to distinguish from each other without genetic analyses and therefore enforcement officers who encounter specimens of CITES-listed species are unlikely to be able to distinguish between them, therefore the populations other *Goniurosaurus* spp. of Viet Nam and China meet the criteria in Annex 2b on the basis of reported identification difficulties.

Other Considerations*:* There may be implementation issues with the six Japanese *Goniurosaurus* species that are not included in the proposal as although there are features that distinguish them from the Chinese and Vietnamese species (the precloacal pores are absent, and claws are not sheathed by scales for the Japanese species) it is reportedly difficult for non-experts, particularly if the geographic origin is not accurately known.

Where species are subject to national protection in China, China could consider publishing a zero-export quota for wild specimens on the CITES website to reflect national legislation.

If the proposal is adopted, Japan may wish to list its six native *Goniurosaurus* species in Appendix III to monitor any potentially increased trade in this species as a result of the other species of the genus being listed.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy, Range and IUCN Global Category

Species and year of description	*IUCN Global Category*	*Range*
G. huuliensis (2008)	*Critically Endangered (2017) B1ab(iii) ver 3.1*	*Viet Nam*
G. yingdeensis (2010)	*Critically Endangered (2018) B1ab(iii) ver 3.1**	*China*
G. bawanglingensis (2002)	*Endangered (2018) B1ab(iii,v) ver 3.1**	*China*
G. catbaensis (2008)	*Endangered (2016) B1ab(ii,iii)+2ab(ii,iii) ver 3.1*	*Viet Nam*
G. liboensis (2013)	*Endangered (2018) B1ab(iii,v) ver 3.1**	*China*
G. zhelongi (2014)	*Endangered (2018) B1ab(v) ver 3.1**	*China*
G. lichtenfelderi (1897)	*Vulnerable (2017) B1ab(iii,v) ver 3.1*	*Viet Nam*
G. luii (1999)	*Vulnerable (2017) B1ab(iii,v) ver 3.1**	*China, Viet Nam*
G. zhoui (2018)	*Data Deficient (2018) ver 3.1**	*China*
G. araneus (1999)	*Not assessed*	*China, Viet Nam*
G. hainanensis (1908)	*Not assessed*	*China*

G. kadoorieorum (2015)	*Not assessed*	*China*
G. kwangsiensis (2015)	*Not assessed*	*China*

** = accepted for publication in the March 2019 Red List update.*

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

The most recent investigations on Goniurosaurus species confirmed both high degrees of adaptation to specific microhabitats and high levels of local endemism: many species are only known from a single mountain range or island. Current evidence suggests most species in the Goniurosaurus genus to have very limited ranges (Yang & Chan, 2015) and within those ranges to show a high degree of habitat specificity associated with rock/karst topography within primary rainforest (Blair et al.*, 2009). For this reason, most species probably have rather small population sizes and are considered to be vulnerable to various impacts. Species of Goniurosaurus have low densities in the wild. Data on the reproduction of Goniurosaurus species has been mainly obtained from observations in captivity. A gravid female carries only two to three eggs once a year and sexual maturity of* G. lichtenfelderi *males is reached at approximately one year of age (Golinski* et al.*, 2015). These factors all reduce the ability of Goniurosaurus species to recover from natural catastrophes and overharvesting.*

Population studies have only been attempted for G. bawanglingensis*,* G. catbaensis *and* G. luii*, none of which have been able to provide a comprehensive population size or density estimate for the species in question (Ngo* et al.*, 2016; Qi* et al.*, 2011).*

Not a single specimen of *G. catbaensis* could be observed at a formerly populated site after an extreme flood in August 2015 in Viet Hai Village on Cat Ba Island. Since northern Viet Nam has experienced an increase of heavy storms and floods in the last years, it can be assumed that natural catastrophes will lead to a decline or even extirpation of local populations in the future.

Eleven of the thirteen species in the Goniurosaurus genus subject to this proposal have been described in the last twenty years. For G. huuliensis*,* G. kadoorieorum*,* G. kwangsiensis*,* G. liboensis*,* G. zhoui*,* G. yingdeensis *and* G. zhelongi*, the papers describing the species are the only known literature available (Orlov* et al.*, 2008; Wang* et al.*, 2014; Wang* et al.*, 2010; Wang* et al.*, 2013; Yang & Chan, 2015; Zhou* et al.*, 2018).*

There is evidence that wild sourced *Goniurosaurus* specimens are subject to domestic and international trade. *Goniurosaurus* have been popular in the pet market since the 1990s, due to their attractive appearance and colour patterns. Rare species fetch high prices, giving traders incentive for poaching and excessive collection. Most, if not all *Goniurosaurus* species are said to be being sold in the international pet market, mainly in Europe and the USA. It is believed that recently discovered species are particularly vulnerable to exploitation. The authors that described *G. kadoorieorum* and *G. kwangsiensis* in 2015 and *G. zhoui* in 2018 did not make the type localities public for fear of them being exploited. *Evidence suggests that the description of a new species within the Goniurosaurus genus prompts an increase in demand from hobbyists. In the case of* G. luii*, specimens were traded for USD500 prior to the species being described (under an older name), but the price increased to USD1,500 after the species was described (Grismer* et al.*, 1999; Stuart* et al.*, 2006). This suggests that some of the more recently described Goniurosaurus species (e.g. G. kadoorieorum,* G. kwangsiensis*,* G. zhoui *and* G. zhelongi*) could also face commercial exploitation to supply the pet hobbyist market.*

According to the LEMIS database of the U.S. Fish & Wildlife Service, a total of 16,714 specimens of *Goniurosaurus* spp. were imported into the USA between 1999 and 2018. *Goniurosaurus* species were imported into the USA from 15 countries, and the majority of specimens imported were live (90%), 11,515 specimens (69%) were wild-caught and 5,086 specimens (30%) were allegedly bred in captivity, for mainly commercial purposes (97%). *It should be noted that the overall trade figures reported here also include Japanese species of the genus Goniurosaurus; more than 7,000 specimens imported were reported as* Goniurosaurus spp*. These data were not available to the IUCN/TRAFFIC Analyses Team and therefore it is not possible to determine what proportion of specimens were exported from Japan.*

A European market survey in March 2018 confirmed that the trade in *Goniurosaurus* specimens mainly takes place online as well as at international reptile fairs in Europe. Specimens fetch prices between USD35–200 in the international internet markets. A total of 535 specimens of six *Goniurosaurus* species were observed for sale in 120 different online adverts with prices from USD40–137 between September 2017 and March 2018.
The main species in international trade appear to be G. lichtenfelderi*,* G. hainanensis *and* G. luii*; only three of the thirteen species subject to this proposal (*G. kwangsiensis, G, liboensis *and* G. zhoui*) were not observed in*

international trade (Table 1). The figures presented in Table 1 should be treated with some caution as the online and physical surveys in Japan, Europe, China and Viet Nam only give a snapshot of the trade during a limited time period. LEMIS data shows all imports into the USA from 1999-2018.

***Table 1**. Summary of trade in Goniurosaurus spp. (the 13 species subject to this proposal only) in China (data provided in the SS), Japan (unpublished data in Wakao* et al.*, (2018) from surveys between February and May 2017), Viet Nam (data provided in the SS), Europe (data provided in the SS) and USA (LEMIS data from 1999-2018 provided in the SS). Data presented in the format: source of data, range of prices, number of specimens reported. Range States for each species are shaded and therefore trade is/may be domestic.*

Species	China	Viet Nam	Japan	Europe	USA
G. araneus	Yes (traditional medicine, no price or quantity data)	Yes (physical survey, USD 150 no quantity) (some specimens apparently originated in China)	Yes (physical survey, no price, one specimen)	Yes (online, USD 142-175, 56 specimens)	Yes (LEMIS data, no price, 138 specimens)
G. bawanglingensis	Yes (online, USD 130, unknown number of specimens)		Yes (online and physical survey, USD 430, four specimens)	Yes (physical survey, USD 200 per pair, no quantity)	
G. catbaensis		Yes (physical survey, USD 7-150, no quantity)		Yes (online and physical survey, USD 170-225, 29 specimens)	
G. hainanensis	Yes (online, USD 15, unknown number of specimens)		Yes (physical survey, USD 85, eight specimens)	Yes (online and physical survey, USD 40-160, 162 specimens)	Yes (LEMIS data, no price, 999 specimens)
G. huuliensis	Yes (traditional medicine, no price or quantity data)	Yes (online and physical survey, USD 20-100, no quantity)	Yes (online and physical survey, USD 340-380, five specimens)	Yes (online, USD 150, 41 specimens)	
G. kadoorieorum	Yes (traditional medicine, no price or quantity data)		Yes (physical survey, USD 160, one specimen)		
G. kwangsiensis	Yes (traditional medicine, no price or quantity data)				
G. liboensis	Yes (online, USD 130–240, unknown number of specimens)				
G. lichtenfelderi		Yes (physical survey, USD 100, no quantity)	Yes (online and physical survey, USD 90-220, three specimens)	Yes (online and physical, USD 70-100, 97 specimens)	Yes (LEMIS data, no price, 7,281 specimens)
G. luii	Yes (traditional medicine and online, USD 57, unknown number of specimens)	Yes (online and physical survey, USD 20-130, no quantity)	Yes (online and physical survey, USD 130-400, nine specimens)	Yes (online and physical survey, USD 40-160, 150 specimens)	Yes (LEMIS data, no price, 608 specimens)
G. yingdeensis	Yes (online USD 130–240, unknown number of specimens)		Yes (online and physical survey, USD 800-1,700, four specimens)		
G. zhelongi			Yes (physical survey, USD 2,350, two specimens)		
G. zhoui			*(No data available)*		

Available data on population status and trade are summarised below.

Goniurosaurus araneus
This species was described in 1999 and is known from rural areas surrounding Cao Bang Town in Cao Bang Province, Viet Nam (Grismer et al.*, 1999), and in south-eastern China in Guangxi Province (Chen* et al.*, 2014). Field studies suggest that G. araneus has already been extirpated at its type locality due to over-harvesting for the pet trade* (Ngo et al., 2016). Several extensive field surveys in Cao Bang failed to record any specimens of *G. araneus* in northern Viet Nam. However, *G. araneus* (of Chinese origin) have been observed for sale in Viet Nam in 2018. According to one dealer in Dong Nai Province, southern Viet Nam, 50 animals were illegally imported from China in 2016 and then smuggled to Thailand with a higher price of USD 150 per individual. *There was a reported case of one dealer exporting over 10,000 individuals combined of* G. luii *and* G. araneus *to the USA for the pet trade (breakdown of species unknown)* (Grismer et al., 1999). According to the LEMIS database of the U.S. Fish & Wildlife Service, 138 *G. araneus* specimens were imported into the USA between 1999 and 2018. An online European market survey between September 2017 and March 2018 observed a total of 56 *G. araneus* individuals advertised. *TRAFFIC observed one* G. araneus *in a physical market survey in Japan (from Wakao* et al.*, 2018).* Occasionally *G. araneus* is harvested from the wild in China for use in traditional medicine.

Goniurosaurus bawanglingensis
This species was described in 2002 and is only known from Hainan Island, which has a forested area of only 10% (Chen et al.*, 2016) suggesting a maximum range of 3,290 km^2 (this figure is likely to be an overestimation).* During field surveys in 2017 and 2018, local rangers reported that habitat degradation and illegal harvest were frequently observed and the populations were considered to be in decline. *A population study of* G. bawanglingensis *on Hainan Island produced a population estimate for the species of 380 individuals using both transect and mark-release-recapture methods (Qi* et al.*, 2011). This study concentrated on the three regions of the Bangwangling National Nature Reserve where* G. bawanglingensis *is known to occur. The species is also known to occur in the Yinggeling Nature Reserve, but the population is unknown.*

TRAFFIC observed one G. bawanglingensis *individual for sale online and three individuals in a physical market survey between February and May 2017 in Japan (from Wakao* et al.*, 2018).* One advert was observed during a survey of the online market Tabao.com in November 2018, where *G. bawanglingensis* was offered by a dealer for USD 130 per individual. *In Europe, online adverts on Facebook and on German websites advertised* G. bawanglingensis *for sale for USD 200/pair.*

Goniurosaurus catbaensis
Goniurosaurus catbaensis *is an island endemic, only known from Cat Ba Island, Viet Nam (surface area of 200 km^2). The population was reported to be severely fragmented and the habitat declining (Nguyen & Schingen, 2016). The total wild population has not been estimated. However, preliminary estimations of population size along seven transects (each of 1.1-4.2 km in length) at two sites on Cat Ba Island were repeatedly undertaken in July and August 2014, and again in May 2015. Small numbers (12-17 individuals) were encountered each month.* Goniurosaurus catbaensis *was found in relatively low densities of 1.2 individuals per transect km at sites containing suitable habitat such as limestone cliffs and caves; the species is limited to these formations and is not found throughout the whole of Cat Ba Island (Ngo* et al.*, 2016). Thus, the whole population size is assumed to be relatively small and may be below 250 mature individuals. The population is said to almost certainly be too small to be viable (Nguyen & Schingen, 2016).* Habitat destruction for touristic purposes threatens wild populations. A strong flood event in 2015 appeared to have caused local extirpations of *G. catbaensis* at some sites.

Even though only recently described (2008), *G. catbaensis* has already been regularly observed on offer in European pet markets and in local pet shops in southern Viet Nam as well as in Hang Market, Hai Phong City, northern Viet Nam during recent years. Specimens of *G. catbaensis* were collected at the type locality and subsequently offered for sale for USD 7–25 within Viet Nam. At least 20 specimens of *G. catbaensis* were reportedly smuggled to Thailand with a higher price up to USD 150 per individual. *Goniurosaurus catbaensis* was recorded for sale in Europe for USD 340 per pair. *On the largest European reptile classifieds website,* G. catbaensis *was the species most commonly advertised of seven Vietnamese endemic reptile observed over a one month period in 2017 (Janssen & Indenbaum, 2018).*

Goniurosaurus hainanensis
This species was described in 1908 and is only known from Hainan Island, which has a forested area of only 10% (Chen et al.*, 2016) suggesting a maximum range of 3,290 km^2 (this figure is likely to be an overestimation).* Local rangers reported during surveys in 2017/2018 that habitat degradation and illegal harvest were frequently observed and the populations could be considered to be in decline. *No further population or density data could be found in the literature for G. hainanensis.*

One online dealer in China offered *G. hainanensis* in a large quantity with a price of USD 15 per individual (*no further information on the number of individuals was provided)*. A market survey in March 2018 surveying online

adverts and international reptile fairs observed *G. hainanensis* for sale for USD 40–160 (162 individuals) in Europe. According to the LEMIS database of the U.S. Fish & Wildlife Service, 999 *G. hainanensis* specimens were imported into the USA between 1999 and 2018. *TRAFFIC observed eight* G. hainanensis *individuals in a physical survey in Japan (from Wakao* et al.*, 2018).*

Goniurosaurus huuliensis
This species was described in 2008 and is only known from a single nature reserve in Lang Song Province in Viet Nam (Orlov et al.*, 2008). No population or density estimate could be found for* G. huuliensis *in the literature.*

G. huuliensis specimens have been used locally in traditional medicine and have been advertised online and recorded for sale in shops in Viet Nam. According to an interview with a pet shop owner in Dong Nai Province, southern Viet Nam in April 2018, *G. huuliensis* was not available in stock, as they were very difficult to be collected in the wild at that time. At least 20 specimens of *G. lichtenfelderi* or *G. huuliensis* were recorded to have been allegedly smuggled without any permits from Viet Nam to Thailand with prices about USD 100 per individual in 2016. *TRAFFIC observed five* G. huuliensis *for sale online and at a reptile fair between February and May 2017 in Japan (from Wakao* et al.*, 2018).* An online European market survey between September 2017 and March 2018 observed a total of 41 *G. huuliensis* individuals advertised.

Goniurosaurus kadoorieorum
Described in 2015 and only known from the Guangxi Zhuang Autonomous Region in Southern China, where it was discovered at specific sites, but no further detail was given on the location due to fear of over-harvesting of the species for the international pet trade (Yang & Chan, 2015). No population or density estimates for G. kadoorieorum *could be found in the literature.*

Occasionally *G. kadoorieorum* is harvested from the wild in China for use in the traditional medicine. *TRAFFIC observed one* G. kadoorieorum *in a physical survey in Japan (from Wakao* et al.*, 2018). This was the only evidence of this species in international trade.*

Goniurosaurus kwangsiensis
Described in 2015 and only known from the Guangxi Zhuang Autonomous Region in Southern China, where it was discovered at specific sites, but no further detail was given on the location due to fear of overharvesting of the species for the international pet trade (Yang & Chan, 2015). No population or density estimates for G. kwangsiensis *could be found in the literature.*

Occasionally *G. kwangsiensis* is harvested from the wild in China for the use in the traditional medicine, *but no evidence could be found for* G. kwangsiensis *in international trade.*

Goniurosaurus liboensis
This species was described in 2013 and is only known from two nature reserves in Guangxi Zhuang Autonomous Region in Southern China (Wang et al.*, 2013). Shortly after the description of the species, it was reported that collectors used the species description paper to locate and collect specimens for the international pet trade (Geggel, 2016). No population or density estimate could be found for* G. liboensis *in the literature.*

One advert was observed during a survey of the online market Tabao.com in November 2018, where *G. liboensis* was offered by a dealer in unknown quantities.

Goniurosaurus lichtenfelderi
This species was described in 1897 and is known from three provinces and one archipelago in Viet Nam (Nguyen, 2018). No population or density estimate could be found for G. lichtenfelderi in the literature.

In Viet Nam a survey in 2018 (physical as well as online) found that *G. lichtenfelderi* is being offered locally for USD 100 per animal. According to an interview with a pet shop owner in Dong Nai Province, southern Viet Nam in April 2018, *G. lichtenfelderi* was not available in stock, as they were very difficult to be collected in the wild at that time. At least 20 specimens of *G. lichtenfelderi* or *G. huuliensis* were recorded to have been allegedly smuggled without any permits from Viet Nam to Thailand with prices about USD 100 per individual in 2016. *TRAFFIC observed one* G. lichtenfelderi *for sale online between February and May 2017 in Japan (from Wakao* et al.*, 2018).* According to the LEMIS database of the U.S. Fish & Wildlife Service, 7,281 *G. lichtenfelderi* specimens were imported into the USA between 1999 and 2018. An online European market survey between September 2017 and March 2018 observed a total of 97 *G. lichtenfelderi* individuals advertised.

Goniurosaurus luii
This species was described in 1999 *and is known from northern Viet Nam and the border area of western Guangxi, China (Grismer* et al.*, 1999; Yang & Chan, 2015). Information on the species' natural history is still poorly known and data on its abundance in Viet Nam is lacking to date (Ngo* et al.*, 2016). A study in northern Viet Nam of one population in 2014 observed 15 animals along seven transects and estimated the total population size to be 21; for fear of illegal collection, the exact localities and area of the site were not reported (Ngo* et al.*, 2016).* Evidence from field work suggests that *G. luii* is already extirpated at its type locality due to over-harvesting for the pet trade; It was claimed that *G. luii* reached very high prices per individual in importing countries after it was described in 1999.

Since 2014, wild-caught *G. luii* individuals have been frequently recorded in local pet shops from Dong Nai Province and Ho Chi Minh City, southern Viet Nam (as well as online) - a survey in 2018 found that *G. luii* is being offered locally for USD 20 - 25 per animal. Interviews with two local dealers and two private keepers in southern Viet Nam revealed that they paid villagers living within the species' range in northern Viet Nam to collect specimens of this species. According to local traders, the species is frequently being exported illegally to Thailand and Indonesia (at least 50 specimens per transaction, USD 100-150 per individual) for re-export to Europe and USA. Between 1999 and 2018 608 *G. luii* were reported as imported into the USA. *In 1996, one dealer in China reportedly exported a combination of over 10,000* G. luii *and* G. araneus *to the USA for the pet trade (breakdown of species unknown) (Grismer* et al.*, 1999). TRAFFIC observed one* G. luii *for sale online between February and May 2017 in Japan (from Wakao* et al.*, 2018).* Occasionally *G. luii* is harvested from the wild in China for use in the traditional medicine.

Goniurosaurus yingdeensis
This species was described in 2010 and is only known from the surrounding areas of Guoshanyao village in China (Wang et al.*, 2010). Shortly after the description of the species, it was reported that collectors used the species description paper to locate and collect specimens for the international pet trade (Geggel, 2016).* Local rangers reported during surveys in 2017/2018 that habitat degradation and illegal harvest had frequently been observed and the populations could be considered to be in decline. *No population or density estimate could be found for* G. yingdeensis *in the literature.*

TRAFFIC observed one G. yingdeensis *for sale online between February and May 2017 in Japan (from Wakao* et al.*, 2018).* One online advert was observed in China during a survey of the online market Tabao.com in November 2018, where *G. yingdeensis* was offered by a dealer for between USD 130 – 240 per individual.

Goniurosaurus zhelongi
This species was described in 2014 and is only known from the Shimentai Nature Reserve in China (Wang et al.*, 2014).* Local rangers reported during surveys in 2017/2018 that habitat degradation and illegal harvest had frequently been observed and the populations could be considered to be in decline. *No population or density estimate could be found for* G. zhelongi *in the literature.*

TRAFFIC observed two G. zhelongi *individuals for sale at a reptile fair in Japan (from Wakao* et al.*, 2018).*

Goniurosaurus zhoui
This species was described in 2018 and is only known from Hainan Island, which has a forested area of only 10% (Chen et al.*, 2016) suggesting a maximum range of 3,290 km^2 (this figure is likely to be an overestimation). There are no population or density data for* G. zhoui *and there is no evidence of this species in international trade.*

Additional Information

Threats
Pressure from harvesting is considered a serious threat, since *Goniurosaurus* species are habitat specialists occurring in low densities in the wild, most species have highly limited distribution ranges and at least three species are thought to have been locally extirpated.

Other threats include habitat loss (e.g. due to quarrying, forest clearance for agriculture, illegal timber logging, infrastructure), impacts from tourism activities, climate change and associated exceptional weather events, such as increasingly extreme floods and storms in north-eastern Viet Nam which have resulted in mass die-offs. Some species in the genus are believed to be especially vulnerable to extinction by climate change due to an assumed narrow temperature optimum and limited options for behavioural and physiological adaptation.

Conservation, management and legislation
According to *the Law of the People's Republic of China on the Protection of Wildlife*, anyone who intends to hunt or catch wildlife that is not under special state protection must obtain a license and observe the quota assigned. From 2000 onwards, the former recognised *G. lichtenfelderi* and *G. hainanensis* have been listed as species of terrestrial wildlife that are beneficial or of important economic or scientific value so logically the newly described species of *Goniurosaurus* in China considered as cryptic species of *G. lichtenfelderi* and *G. hainanensis* are also under protection. *Goniurosaurus hainanensis* and *G. bawanglingensis* have been further listed as wildlife under special local protection in Hainan province. In nature reserves, the hunting and catching of wildlife and other activities which are harmful to the survival of wildlife are prohibited.

No member of the genus *Goniurosaurus* is as yet included in any wildlife protection laws in Viet Nam. However, collecting wild animals for the pet trade without permits is illegal in the country. *Permits are required for collection for all forms of trade in Goniurosaurus spp. not specifically for the pet trade (CITES Management Authority of Viet Nam, pers. comm.).* Collecting wild animals including *Goniurosaurus* species has been strictly restricted within protected areas as national parks and nature reserves. *Goniurosaurus* species have been proposed to be listed in the Governmental Decree as Group II B, which will be confirmed in early 2019.*The proposed listing of* Goniurosaurus spp*. in the Governmental Decree will be automatic if the genus is listed in Appendix II. If the genus is not listed in Appendix II, then there would be debate surrounding its inclusion. Inclusion in Group II B would more strictly limit international exports rather than regulate domestic trade (CITES Management Authority of Viet Nam, pers. comm.).*

There are no existing specific measures in either China or Viet Nam to protect the habitat of *Goniurosaurus* species or specifically protect the species. However, some species occur within protected areas.

Artificial Propagation/captive breeding
Goniurosaurus species are also being kept by zoos and other wildlife organisations for scientific purposes and captive breeding. A total of 34 specimens are being kept in seven institutions in USA/Europe:

- *G. hainanensis* n=7 (3 x wild-caught, and 4 x captive-bred)
- *G. lichtenfelderi* n=11 (all captive bred)
- *G. luii* n=12 (all captive-bred)
- *G. spp* n=4 (1 x captive-bred and 3 x undetermined)

G. araneus, G. catbaensis and *G. huuliensis* are also known to be kept in zoos *(numbers unknown)*.

In Viet Nam, three *Goniurosaurus* species (*G. catbaensis, G. lichtenfelderi, G. luii*) are currently kept at the Me Linh Station for Biodiversity in Vinh Phuc Province, to establish an *ex-situ* back-up population. The first successful reproduction of *G. catbaensis* at the Me Linh Station occurred in July 2018.

The SS notes that currently, hundreds of captive-born juveniles of *Goniurosaurus* species are available on the global pet market every year for about USD 40 each. According to the US LEMIS database, a total of 5,086 captive-bred *Goniurosaurus* spp. specimens were imported into the USA between 1999 and 2018 for commercial purposes and scientific uses.

Implementation challenges (including similar species)
The remaining six species within this genus are endemic to Japan: *G. kuroiwae*, *G. orientalis*, *G. sengokuii, G. splendens*, *G. toyamai* and *G. yamashinae*. The Japanese species can be distinguished from other species in the genus since the precloacal pores are absent, and claws are not sheathed by scales (in the other 13 species, precloacal pores are present, and the claws are sheathed by scales).

Studies revealed that morphological characters amongst species within each group of the genus (the Chinese and Vietnamese species versus the Japanese species) are significantly similar. Species identification by non-specialists is therefore difficult, especially if location data are not (accurately) provided. As a result, molecular analyses are necessary to distinguish species.

One study mentions that there are no significant differences in diagnostic characters between G. luii *and* G. kadoorieorum *(Ngo* et al*., 2016). Blair* et al *(2009) indicate that molecular investigation may be necessary to determine species designations in the* G. lichtenfelderi *group (which includes :* G. hainanensis*,* G. lichtenfelderi *and* G. zhoui*).*

Potential risk(s) of a listing

Given that the six Japanese Goniurosaurus species are not included in the proposal, this could lead to species in China and Viet Nam being mis-labelled as Japanese species in order to circumvent CITES controls (although it is reportedly possible to distinguish the two groups of species based on morphological differences). Listing of the Chinese and Vietnamese species could cause an increase in demand for the non-listed Japanese species.

Potential benefit(s) of listing for trade regulation

Given that the Goniurosaurus is a cryptic genus, there is a high likelihood that further species will be discovered in the near future, and it appears that newly described species are quickly over-exploited. If the proposed genus-level listing is adopted, any new species would automatically also be listed in Appendix II and therefore protected from potential overexploitation.

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Inclusion of Tokay Gecko *Gekko gecko* in Appendix II

Proponents: European Union, India, Philippines and United States of America

Summary: The Tokay Gecko *Gekko gecko* is the second largest extant gecko species, with a striking bluish-grey body with red or orange spots. It has a widespread distribution throughout South-east Asia, China, Bangladesh and India. It has also been introduced to several countries including Brazil, Madagascar, the USA and areas of the West Indies. This species inhabits a broad range of habitats from forest to human-modified environments. Over a six-month breeding period Tokay Geckos produce clutches of one to two eggs laid at 30-day intervals, which are deposited in tree holes and guarded by both parents. The species has been used in Traditional Chinese Medicine for hundreds of years and is sold throughout South-east Asia in dried form or preserved in alcohol; to a lesser extent, it is also traded live for pets.

Consumption of *G. gecko* for medicinal purposes does occur domestically in South-east Asian countries, but international trade is assumed to be on a much larger scale and consumption is centred around China and Viet Nam. Thailand and Indonesia (particularly Java) are the main exporters; *G. gecko* is not protected in these countries but subject to regulation by quota/permit:

- Thailand – Exports of *G. gecko* from Thailand alone have been reported to be between two to five million geckos per year going to China, Taiwan Province of China (which reported imports of 11 million from Thailand between 2004 and 2013), Malaysia and the USA. The species is not nationally protected in Thailand, but exports and imports of the species require a permit.
- Indonesia – Three traders were estimated to be exporting 1.2 million dried *G. gecko* annually in 2006, although Indonesia has no quota for exports of dried *G. gecko* for medicinal purposes. In 2006 the quota for live animals was 50,000 individuals, of which 5,000 were intended for domestic consumption and 45,000 were intended for export.

Exports were also reported from Cambodia, Lao People's Democratic Republic (PDR), Malaysia, Myanmar and the Philippines.

A novel trend in *G. gecko* demand emerged in 2009 after consumption of its parts was promoted as a cure for HIV/AIDS. Trade reportedly increased throughout South-east Asia, but it is thought that trade peaked in 2010/2011 and has since declined after improved enforcement and a realisation that the claims were unfounded. The international pet trade in live, wild-caught *G. gecko* is thought to be diminishing (from available data) with imports of wild-caught live *G. gecko* into the USA decreasing by more than 50% from 2007–2016.

There are no empirical population estimates for *G. gecko*, and while the species is thought to be common in most of its range there is contradictory information on national declines. Populations in the main consuming countries have declined although it is not clear on what scale: in China the species was listed nationally as critically endangered (2016), although the Chinese Management Authority (MA) considered the national population to be "large and stable". In Viet Nam the MA reported localised declines due to small-scale harvesting, and the national Red Data Book (2015) estimated it was declining (by no more than 30%), but this was not based on empirical evidence.

A recent IUCN Red List assessment classifies the species as Least Concern (accepted for publication in the March 2019 Red List update). There are anecdotal reports of national population declines in Bangladesh and Thailand, but the more recent global assessment suggests that the population trend overall is unknown. In Bangladesh, a regularly cited, recent, 50% decline in *Gekko gecko* populations has been contradicted by the Red List of Bangladesh, which states that although there is tremendous poaching pressure, the species is common, and the population trend is presumed to be stable. Thailand stated that *G. gecko* was considered abundant countrywide (least concern in 2005), although declines were noted in the north-east, and poaching was causing the population to "dwindle". The Philippines reported declines. Population trends are not clear from other range States.

Gekko gecko has some form of legal protection in Bangladesh, Cambodia, China, India, Lao PDR, Peninsular Malaysia, the Philippines and Viet Nam; but it is not protected (outside of protected areas) in Indonesia, Myanmar or Thailand.

Analysis: *Gekko gecko* has an extensive range across a large part of Asia and is known to be present in a broad range of habitats including human-modified environments. Population information is scarce and although some range States have anecdotal reports of population declines, others have reported stable populations and it is considered to be common in much of its range. The latest IUCN Red List assessment to be published in 2019 categorises the species as Least Concern. While population information is contradictory, there are concerns regarding the population in the main consuming countries of China (critically endangered, 2016) and Viet Nam (near threatened, 2015, localised declines noted). There are also concerns over certain populations in the main importing areas: Thailand (least concern, 2005, with declines in the north-east which borders Viet Nam) and Java, Indonesia (anecdotally reported to be extremely difficult to find, whereas in parts of Bali and Sulawesi it is still common).

The species has been harvested for medicinal purposes for hundreds of years and there is a large body of evidence to show that the species is currently traded in the millions or tens of millions annually (most are presumed to be wild-caught), for use in traditional medicines. While *G. gecko* is still considered to be common throughout most of its range, in the main consuming countries the populations appear to be declining, as do populations in other range States now exporting to these countries. Although there is a large degree of uncertainty regarding the impact of international trade, it may be precautionary to list the species in Appendix II in order to ensure that trade of specimens from the wild does not threaten the species.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Gekko gecko (Linnaeus, 1758).

This proposal does not consider Chinese specimens recently described as Black Gecko *G. reevesii* as a separate species. The CITES Nomenclature Specialist of the Animals Committee considered that "the recognition of *reevesii* as a separate species is too recent to be generally known in the biological and trade management communities, and its distinction from the widespread and widely traded *Gekko gecko* appears insufficient for consistent, reliable identification of specimens in trade. Thus, for practical purposes it appears advisable for CITES purposes to retain the traditional concept of *Gekko gecko*, and to include mildly-different peripheral and island populations (including Chinese specimens attributed to *reevesii*) as part of the species until stronger evidence of taxonomic distinctness and practical ability to differentially identify specimens in trade becomes available." The Chinese Red List recognises *G. reevesii as a species (critically endangered, 2016) (Jiang* et al.*,2016).*

Range: Bangladesh, Cambodia, China, India, Indonesia, Lao PDR, Malaysia, Myanmar, Nepal, Philippines, Singapore, Thailand and Viet Nam

Introduced populations: USA (Florida, Hawaii and Texas), Lesser Antilles (including Martinique), Madagascar, Belize and *Brazil* (Carlos, Junior, Piva, Batista, & Machado, 2015).The species' status in Taiwan Province of China is disputed; there are very few occurrence records, and it is unclear whether they represent introduced individuals.

IUCN Global Category

Least Concern (2017) ver. 3.1 (accepted for publication in the March 2019 Red List update).

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

Gekko gecko is nocturnal, solitary and is principally a sit-and-wait forager that feeds on moths, grasshoppers, beetles, other geckos, small mice and snakes. Males and females come together during the breeding season,

which lasts for approximately six months. Clutches comprise 1–2 eggs, laid at intervals of 30 days, which are deposited in tree holes and guarded by both parents. Tokay geckos give a territorial call, syllabised as "tok-ay", which is uttered 4–9 times in slow succession and can be heard day and night from several metres away.

In captivity, the species has been known to live for 23 years, but longevity in the wild is considered likely to be shorter.

The population status of *Gekko gecko* is largely unknown, the perception of the species as common is considered to be a result of its wide distribution, relatively high fecundity, and ability to thrive in human modified landscapes, but several declines had been recorded.

The IUCN Red List assessment (accepted for publication in the March 2019 Red List update) describes the species as common in its range, but there is no global population estimate and population trends are reported to be unknown. There are no national or local population estimates for Gekko gecko *in any of its range and only a few studies that consider their population density within a given area (Huang* et al., *1995; Xu* et al., *2009). Many reports of declines are anecdotal and based on reports in the media (Thongsa-Ard & Thongsa-Ard, 2003) rather than in the form of empirical evidence from studies over a given time period.*

The availability of information on the status of *Gekko gecko* is geographically patchy.

Gekko gecko was described as Rare in a field guide to the reptiles of **Nepal.**

In **Lao PDR**, it is listed as a category III species in the 2008 Wildlife and Aquatic Law, which is reserved for "common wildlife" that is not "classified in the rare and near extinct categories".

In the **Malaysian** States of Sabah and Sarawak, the species was considered "quite rare".

In **Bangladesh**, populations of *Gekko gecko* were estimated to have "recently" declined by 50% due to collection as a result of novel medicinal claims. Although, the CITES (MA) of Bangladesh reported the species to be common in 2018 and noted that it presumably had a "large population" in the country. *In 2015, the Red List of Bangladesh (Ahsan & Khan, 2015) has assessed G. gecko as least concern, stating that although there is poaching pressure, the species is widely distributed, with a presumed large population and a presumed stable population trend.*

Declines have been reported in **Indonesia**. *Anecdotal accounts from Indonesia suggest that trapping pressure continues to be high, and that* Gekko gecko *is considered to be valuable in trade and therefore trappers will seek them out if they hear them calling (C. Wilcox* pers. comm.*). In Java specifically, it has been reported that the species has been extremely difficult to find in the past few years (A. Miller* pers. comm.*). In other locations such as Bali,* G. gecko *was the most commonly encountered reptile in a study of human modified landscapes (Janiawati* et al., *2016) and in Sulawesi,* G. gecko *was considered common in Lore Lindu National Park (Wanger, 2011).*

Whilst the species was considered abundant in most countrywide areas in **Thailand**, the status of the wild population was unknown according to the CITES MA of Thailand. Declines have been reported in the media in north-eastern Thailand. *Gekko gecko* was classified as a species of least concern in Thailand's 2005 Red Data Book. However, it was reported in the media that "the gecko population has dwindled in **Myanmar** and **Thailand**" after years of poaching. It has been noted, however, that the species is commonly associated with human-modified environments and can be common in houses in suburbia but is at lower densities in urban and agricultural landscapes according to a study conducted in northern Thailand. Declines have been reported in north-eastern Thailand.

In **China**, the population of *Gekko gecko* was reported to have been "drastically reduced" in recent years as a result of habitat destruction and hunting, and the species was categorised as an endangered species in China's 1998 Red Data Book. *In the 2015 version of China's Red Data Book,* G. gecko *is categorised as critically endangered, but it should be noted that* G. reevesii *was considered as a separate species (Jiang* et al., *2016).* However, the CITES MA of China reported that the wild population "in recent years" was stable and large in the country. Habitat destruction was considered to have been a contributing factor to the *G. gecko*'s dramatic decline in China. *Xu* et al., *(2009) reported that* G. gecko *populations were recovering as more calls were heard during a survey in the Longgang Nature Reserve than in the early 2000s. These results should be treated with caution as the survey only recorded a total of 3 km of transects in an area of 30.2 km^2 so do not represent an overall recovery of populations in China.*

Gekko gecko was listed as threatened, vulnerable, and near threatened in the 2000, 2007 and 2015 versions of Viet Nam Red Data Book respectively. In the 2015 version, the population was estimated to have declined (by less than 30%) over the last 10 years, and possible declines in upland areas were noted as a result of small-scale harvesting for traditional medicines.

Studies in ***India*** *found* Gekko gecko *in commercial areas, residential areas and forested areas (Purkayastha* et al., 2011)*, with some studies showing that more individuals were found in mud houses than they were in trees (Singh & Choudhury, 2016). Although there is evidence showing that* G. gecko *occurs at lower densities in modified environments (Weterings* et al., *2018), it is clear that they are habitat generalists and are found widely in human modified environments.*

The only numerical estimates of decline that could be located were for Bangladesh and Viet Nam, though population decreases have also been noted in Indonesia, north-eastern Thailand, the Philippines and Myanmar, and "drastic" population decreases have been reported in southern China.

Although numerical estimates have been given for a 50% decline in Bangladesh and declines of 20% were reported in Viet Nam's 2007 Red Data Book, neither of these figures appear to be based on any empirical evidence. All of the other noted population decreases have been from media sources rather than based on any scientific population studies.

Deforestation rates in several range States (such as Indonesia and Malaysia) have historically been noted to be high.

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

National utilisation

Gekko gecko is principally used for medicinal purposes and is traded either preserved in alcohol or as dried specimens, but *G. gecko* meat was also reported to be served in restaurants in cities close to the Viet Nam/China border. Virtually all trade is thought to be in animals that were harvested from the wild. Although the species is used domestically in Indonesia for various medicinal purposes, the volumes of geckos involved were considered to be negligible compared to those involved in the international trade, whose consumption as traditional medicine is thought to be centred in China and Viet Nam. Geckos harvested in Viet Nam are mainly used domestically rather than exported. The CITES MA of Bangladesh noted that there was no record of local consumption of *G. gecko* in the country.

Gekko gecko has been used in Traditional Chinese Medicine for hundreds of years, where it has been reported to be effective in suppressing asthma, treating diabetes and erectile dysfunction, replenishing "the kidney essence", and relieving coughing. A novel demand emerged in 2009, for use as a cure for HIV/AIDS; this was centred in Peninsular Malaysia and Singapore and peaked in 2010/2011 but has since declined. The reasons for this are considered to be unclear but were thought to be related to a combination of improved enforcement, realisation that the claims were unfounded following World Health Organization statements outlining the lack of evidence for the effectiveness of their use, and the prevalence of scams. By 2013 the trade in Tokay Geckos for novel medicinal claims was considered to be relatively small and did not appear to pose a threat to the conservation of the species.

Trade

Levels of international trade in *Gekko gecko* are considered to be very high, with the principal destinations for dried exports of Tokay Geckos (i.e. those which are intended for consumption as traditional medicine) reported to be China and Viet Nam. The volumes of *G. gecko* imported into these countries annually were reported to be unknown but substantial. Taiwan Province of China alone, for example, was reported in 2013 to have imported ~15 million *G. gecko* individuals since 2004; 71% from Thailand with the remainder coming mostly from Indonesia (Taiwan Province of China import data). In China, reported annual trade levels of *G. gecko* in Pingxiang City were of 20,000–30,000 individuals, and Dongxing City of 2,400–6,000 individuals. Dried geckos have additionally been reported to be traded beyond Asia; between 1998–2002, for example, 8,503 kg of *G. gecko* were reported imported to the USA for medicinal purposes.

In Indonesia, export of dried individuals of *Gekko gecko* is not permitted; however, the country does have a quota system for Tokay Geckos to be consumed domestically and exported live for the pet industry. In 2006 this quota was reported to be 50,000 individuals, of which 5,000 were intended for domestic consumption and 45 000 were intended for export.

More recent reports from Indonesia suggest that in 2014 this quota was increased by the Indonesian Ministry of Forestry to over three million live captive-bred Gekko gecko*, intended to supply the international pet trade (Nijman & Shepherd, 2015). Research predicted that it was not an economically viable option to produce geckos on that scale and suggested that captive-breeding permits were being issued to launder wild-caught geckos for illegal export as dried specimens (Nijman & Shepherd, 2015). Previous research had shown that the quota of 45,000 live geckos (mentioned above) intended for the international pet trade was drastically exceeded and that 1.2 million dried* G. gecko *were exported to China outside of the quota system (Nijman, Shepherd, Mumpuni, & Sanders, 2012).*

The vast majority of global trade in *Gekko gecko* is thought to originate from Thailand and Java, Indonesia. However, Lao PDR, Myanmar, Peninsular Malaysia, Cambodia and the Philippines have also been highlighted as important source countries. It has been estimated that 2–5 million Tokay Geckos were exported from Thailand to China, Taiwan Province of China, Malaysia and the USA each year; whereas other reports state that, on average, the country exported 40 t of Tokay Geckos (equivalent to ca. 1,467,000 individuals) annually to Taiwan Province of China alone. Thailand reported exporting 1,099,178 live geckos between 2014–2018 (for unknown purposes), and 1,455,362 "live and dried" specimens in 2017 and 2018 combined. It was noted that export volumes had decreased, reportedly as a response to declining demand in destination countries and the increasing costs of export procedures. The main harvest areas in Thailand were reported to be in northern and north-eastern regions.

A one day survey at Mong La market in Myanmar counted 72 Gekko gecko *bodies (Shepherd & Nijman, 2007).*

Many of the Tokay Geckos destined for the **pet trade** have been reported to originate in Viet Nam and Java, Indonesia. It has been reported that the species was mainly exported to the EU and USA; however, *Gekko gecko* has also been noted to be present, among others, in the pet trade of Taiwan Province of China and Malaysia.

According to trade data from the U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS), direct imports of *Gekko gecko* to the USA 2007–2016 mainly comprised 179,681 live geckos, the vast majority of which were wild-sourced (96%) and the remainder reported as captive-bred. Live, wild-sourced geckos were imported from Viet Nam and Indonesia (56 and 44%, respectively). Imports of live geckos by the USA declined by over 50% between 2007 and 2016. Of the 173,275 live, wild-sourced geckos imported from Indonesia and Viet Nam, 12% were re-exported by the USA (information on destination of re-exports was not available). Additionally, imports of bodies (2,447 kg and 96 reported by number) and "medicinal parts or bodies" (8,929 kg and 476 reported by number) were reported 2007–2016, most of which were wild-sourced and imported from China, Hong Kong Special Administrative Region (SAR) and Thailand.

Illegal trade

The export of dead, dried specimens of *Gekko gecko* from Indonesia is not permitted; however, high levels of trade were still considered to take place. It was reported that in 2006 the country's three largest traders exported 1.2 million kiln-dried individuals to China, which would require a harvest of approximately 31,000 individuals per week. Illegal exports were thought to be facilitated by the laundering of wild-caught *G. gecko* individuals through legally registered captive-breeding facilities.

In 2011, 6.75 t (*equates to 225,000 dried geckos (Caillabet, 2013)*) of illegally harvested *Gekko gecko* were seized en route from Indonesia to Hong Kong SAR.

In India, *Gekko gecko* was listed under Schedule IV of the Wildlife (Protection) Act 1972 in 2014. This prohibited collection of the species from the wild; however, illegal collecting was considered to have continued. In 2014, The *Diplomat* newspaper reported that a crackdown by agencies in several Indian States had led to the arrest of over 300 gecko traffickers in the preceding year, and the seizure of over 1,000 Tokay Geckos. Seizures of *G. gecko* have been recorded at India's border with Bhutan (en route to China) and Bangladesh.

The Philippines has made comparatively large confiscations of *Gekko gecko* from 2010 to date; based on wildlife crime records, 12 confiscations were made involving 1,883 *G. gecko* individuals. The country considered it likely that more geckos are being illegally kept in captivity to supply both the local and international demand for the species.

In Bangladesh, the country's Wildlife Crime Control Unit seized 184 gekkos (species unspecified) from July 2012 to July 2018. Myanmar reported seizures of 96 Tokay Geckos between 2010 and 2013. There have been no recent records of *Gekko gecko* seizures in Nepal, aside from a single seizure of a few live animals in Kathmandu six or seven years ago.

In Malaysia, collection, selling, import, export, re-export and ownership of *Gekko gecko* is prohibited without a licence from the Department of Wildlife and National Parks (PERHILITAN). It has been noted that several dealers of *G. gecko* possessed these licences; however, according to PERHILITAN's head of enforcement, no licences to trade in Tokay Geckos have ever been issued. Rather than evidence of illegal activity, however, it has been considered that these contradictions are evidence of miscommunication or a lack of co-ordination between PERHILITAN's headquarters and its State offices.

Additional Information

Conservation, management and legislation

Bangladesh: *Gekko gecko* is listed in Schedule II of the Wildlife (Conservation and Security) Act 2012 (Government of Bangladesh, 2012). Under this Act, hunting, export and import of any wild animal requires a licence. Possession of wild animals as well as their parts or derivatives must be registered (Government of Bangladesh, 2012).

Cambodia: *Gekko gecko* is listed as a common species under the 2002 Forestry Law. Customary use is permitted, but trading or transporting Tokay Geckos above levels "necessary for customary use" is prohibited and can incur a fine of up to three times the market value of the species.

China: *Gekko gecko* is listed under a number of regulations that afford it protection. It is included in (a) the Regulations for the Conservation of Wild Terrestrial Animals, under which the sale of protected species and their derivatives is forbidden and (b) the Regulations on the Conservation and Management of Wild Resources of Medicinal Plants and Animals, under which it is listed as a Category II species (collection is "subject to the prior grant of a Medicine Collection Permit …approved by medicine departments at higher level"). The species is additionally listed in Appendix II of the 1988 Wild Animal Protection Law of the People's Republic of China, which stipulates that, as a Class 2 State protected animal, the species can only be collected, used and traded under licence (People's Republic of China, 1988). In the Guangxi Zhuang Autonomous region, it is listed as a Class 1 key species, as a result of which any collection and trade is prohibited unless it is "necessary for scientific research, domestication and breeding, exhibition or other special purposes". Use of the species for any of these exceptional circumstances requires approval from the Department of Wildlife Administration.

India: *Gekko gecko* was listed under Schedule IV of the Wildlife (Protection) Act, 1972 in 2014, which prohibits collection except for educational purposes and scientific research.

Indonesia: The species is not protected by Indonesian law, but wild specimens are subject to a national annual harvest and export quota. Breeders wishing to export *Gekko gecko* (whether captive-bred or wild-caught) must be registered with the Directorate General of Forest Protection and Nature Conservation (PHKA), and breeders supplying specimens to exporters must be registered with the Regional Natural Resource Management Office (BKSDA) offices at a provincial level. Indonesia has a quota for *G. gecko* for domestic consumption and live export for the pet industry, whose setting and regulation was reported to be managed by the CITES Scientific and Management Authorities respectively. Collection is only permitted from 23 designated sites, mainly in Java but also in Bali, Kalimantan, Sulawesi and Sumatra. There is no quota for the skin or medicinal trade. In 2006 the harvest quota for *G. gecko* was reported to be 50,000 live individuals for the pet industry, of which 5,000 were intended for domestic consumption and 45,000 were intended for export. Most of the quota (24,000 individuals) was for the harvest of *G. gecko* in Java. It is unclear whether the export of live individuals was within quota, but in the same year it was reported that 1.2 million dried individuals were exported from the country's three largest traders to China, in contravention of (a) the scale of the annual quota and (b) the nature of the quota (which is for the export of live specimens only). In March 2014, the Indonesian Ministry of Forestry gave permission to six companies to export *ca*. 3 million live, captive-bred individuals for the pet trade. However, potential issues have been noted relating to the laundering of wild-sourced dried individuals through legal captive breeding facilities.

It has been suggested that given the low wholesale price (USD 1.00–2.30) of a live individual, that it is not economically viable to produce captive-bred Gekko gecko *on an industrial scale to fulfil a quota of 3 million individuals annually (Nijman & Shepherd, 2015).*

Lao PDR: *Gekko gecko* is listed as a Category III species in the 2008 Wildlife and Aquatic Law ("wildlife…that are able to reproduce widely in nature and are very important for social-economic development and educational research). Hunting of species listed in Category III is only permitted in specified seasons and is only allowed "using tools and equipment not harmful to the animals' population". Hunting and capture for commercial purposes requires permission from the Ministry of Agriculture and Forestry. Export of *G. gecko* requires, among other things, an origin certificate, a certificate of health and permission from the Ministry of Agriculture and Forestry.

Malaysia: *Gekko gecko* was protected under the Wildlife Conservation Act 2010, where it is listed as Protected Wildlife under its First Schedule. Under the act, hunting, selling, import, export, re-export or ownership of Tokay Geckos (or any derivative of them) is only allowed with the possession of a licence issued by the Department of Wildlife and National Parks (PERHILITAN). Possession or hunting of the species without a licence can lead to penalties of up to MYR 50,000 (USD 16,000) (or MYR 200,000 (USD 63,957) for females and juveniles) and up to two years in prison. According to PERHILTAN (in 2012), no licences had ever been issued in Peninsular Malaysia to hunt, own or trade in Tokay Geckos. The species is not protected under the State law of Sabah and Sarawak.

Myanmar: No species-specific measures are in place, but all species within protected areas are protected from harvest in accordance with the Conservation of Biodiversity and Protected Areas Law 2018.

Nepal: Under the National Parks and Wildlife Conservation Act, 2029, wildlife (including *Gekko gecko*) cannot be hunted without a licence (Government of Nepal, 1973). The species is currently proposed for Schedule III listing in an upcoming amendment to the National Parks and Wildlife Conservation Act, 2029.

Philippines: *Gekko gecko* is nationally protected under the Republic Act 9147 or Wildlife Resources Conservation and Protection Act. In accordance with section 27 of the said law, hunting and trade of *G. gecko* in the Philippines can only be allowed subject to certain requirements and prior issuance of permits (e.g. a Wildlife Collector's Permit for commercial breeding, a Gratuitous Permit for scientific research, or a Wildlife Special Use Permit for direct trade purposes). No permits have been issued for *G. gecko* so far. A number of private individuals have been permitted legally to possess *G. gecko* provided they were acquired prior to the passage of the Wildlife Conservation and Protection Act (2001) and were registered with the government.

Viet Nam: Hunting of *Gekko gecko* is prohibited in protected areas. The species is listed under the "common forest animal" category whose exploitation is controlled by Circular No. 47/2012/TT-BNNPTNT of 2012. Collection outside of protected areas requires a permit from the country's Provincial Forest Protection Department, and those requesting a permit must carry out a species population assessment and submit a plan for exploitation. The legal origin of specimens must be certified according to the procedure outlined in Circular No. 01/2012/TT-BNNPTNT.

Thailand: The species is not nationally protected, but exports and imports of the species require a permit. Hunting and collection are prohibited in protected areas.

Artificial propagation/captive breeding
Tokay Geckos consumed for traditional medicine in South-east Asia were considered by TRAFFIC to be predominantly harvested from the wild. Individuals were reportedly bred in captivity in mainland China and Viet Nam; however, production was not considered to be capable of meeting demand.

Individuals were also reported to be bred in captivity for live export in Indonesia, and captive breeding was reported to be encouraged by the CITES MA of the country. However, the low sale price of Tokay Geckos (for both live export and as dried specimens) and the high financial cost of maintaining breeding facilities meant that large-scale captive breeding was thought to be financially unfeasible. In 2014, the Indonesian Ministry of Forestry granted permission to six companies to export a total of 3 million captive bred *Gekko gecko* for the pet trade; however, a TRAFFIC investigation noted that none of the companies was known to have ever bred the species in such considerable numbers and were known to supply wild-caught reptiles for the medicinal and meat trade. The production of such quantities of adult-sized geckos for export was noted to require facilities on a scale that was thought highly unlikely to be financially viable, and the TRAFFIC investigation raised suspicion that (a) wild-caught individuals were being laundered into trade described as captive-bred, and (b) geckos were being exported as dead specimens for the traditional medicine trade rather than as live specimens for the pet trade.

No captive breeding of the species was reported to take place in Myanmar.

In the Philippines less than 10 individuals have been bred in captivity, and no legal trade of captive-bred specimens has been recorded to date.

Implementation challenges (including similar species)
The genus *Gekko* consists of 45 named species. *Gekko gecko* is considered to be "easily identifiable" as a result of its orange-spotted, blue-grey skin (however, as previously noted, "black geckos" have a different colouration). Their vocalisations are considered to be unmistakable. It is unclear whether dried Tokay Geckos maintain a diagnostic feature that allows them to be easily identified from other species.

The only other species which could potentially be confused with Gecko gecko *is Smith's Green-eyed Gecko,* Gekko smithii*, but there is no evidence of this species being utilised in Traditional Chinese Medicine.*

Potential risk(s) of a listing

Given that some researchers have already found substitute species being sold as Gekko gecko *for use in TCM (Wagner & Dittmann, 2014), a listing could promote the use of substitute species, which may not be as commonly distributed as* G. gecko*. A major threat to* Tylototriton asperrimus *(subject to CoP18 Prop. 41) is said to be harvest in China for use in traditional medicine, where it is considered a substitute for* Gekko gecko *(van Dijk* et al., *2008).*

Potential benefit(s) of listing for trade regulation

Given the patchy nature of both trade and population data of Gekko gecko *a listing would certainly provide data in the future of the full scale of international legal trade of* G. gecko*. A listing may also promote the collection of empirical data on populations and densities of* G. gecko *throughout its range. This could alternatively be achieved through an Appendix III listing if enough populations were listed.*

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Inclusion of Grenadines Clawed Gecko *Gonatodes daudini* in Appendix I

Proponent: Saint Vincent and the Grenadines

Summary: The Grenadines Clawed Gecko *Gonatodes daudini* is a colourful gecko, which, when fully grown measures just 3cm. It was first discovered in 2005 and is endemic to Union Island of Saint Vincent and the Grenadines. It is only found in the mature forest on Chatham Bay, from near sea level to 300 meters above sea level. This species was classified as Critically Endangered on the IUCN Red List in 2011.

There is only one known population of *G. daudini,* currently tentatively estimated at almost 10,000 and inferred to be decreasing. A survey in 2017 found that densities have dropped by almost 80% since 2010 in some parts of the species range. It has a known extent of occurrence of 1 km^2 and an area of occupancy of 0.5 km^2.

No export permits for commercial purposes have ever been issued. Illegal trade in this species was first reported soon after the species was described in 2005 and exploitation is thought to have accelerated in recent years for the international pet trade. Collection of individuals damages the environment and exposes the remaining geckos to increased risk of predation and desiccation. Little quantitative data are available on numbers traded, but on the basis of online adverts in 2016 and 2017 that identified more than a dozen dealers operating from USA, UK, the Netherlands and Germany and significant microhabitat destruction caused by local harvesters, the proponents conclude that a significant number of geckos are being taken illegally from the wild population. Some captive-breeding appears to be taking place in non-range States.

In addition to threats from alien species and habitat destruction, a road constructed in 2005 has improved access to Chatham Bay. The proposed further development in this area would significantly impact the species' remaining habitat. The species is protected from harvest by existing legislation and is the subject of a Conservation Action Plan that seeks to protect the habitat of *G. daudini*, enhance population survival and incorporate local stakeholders.

Analysis: Population estimates for *Gonatodes daudini* indicate a relatively small population of just under 10,000 individuals, including both mature adults and juveniles. The density of the gecko population has fallen by nearly 80% in some parts of its limited range since 2010. The recorded extent of occupancy (1 km^2) and the area of occupancy (0.5 km^2) are very restricted, and it is only found in one location that is highly vulnerable to both intrinsic and extrinsic factors and has undergone declines. Therefore, the species meets the Appendix I biological criteria. Although the numbers in trade are unclear, illegal harvesting of specimens for international trade is impacting the microhabitat and by inference, also affecting the species. The species therefore meets the criteria for listing in Appendix I in Annex 1 of *Res. Conf. 9.24 (Rev. CoP17).*

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Range

Saint Vincent and the Grenadines

IUCN Global Category

Critically Endangered *B1ab(iii)+2ab(iii) Ver 3.1 (assessed in 2011) (Powell & Henderson, 2011).*

<u>Biological criteria for inclusion in Appendix I</u>

A) Small wild population

There is only a single population of *G. daudini. A survey conducted in 2010 (Bentz* et al.*, 2011) estimated the population to be 6,562 individuals.* A transect based survey in August 2018 estimated the total population to be 9,957. In both surveys the estimates were considered tentative because of the small sample size and patchy distribution of the species. Extensive surveys have failed to detect any other populations on Union Island.

B) Restricted area of distribution
The single population of *G. daudini* has a limited range found only in Chatham Bay on Union Island. It has a known extent of occurrence of approximately 1 km^2 and an area of occupancy of 0.5 km^2, and there is a continuing decline in the extent and quality of its habitat resulting from increasing development and reportedly from collection pressure. *This litter dwelling species relies on relatively mature forest, and this area of the island holds one of the last relatively intact areas of secondary forest (Bentz* et al*. 2011).*

C) Decline in number of wild individuals
The population of *G. daudini* is inferred to be decreasing. The 2018 survey replicated the methods of Bentz et al. (2011) and found that while the population density is stable on rocky outcrops, it has fallen by nearly 80% in the most accessible parts of the species range (from 87/ha in 2010 to 19/ha in 2018). *In 2011, when the species was assessed on the IUCN Red List as Critically Endangered the population was considered stable (Powell & Henderson, 2011). However, the 2018 survey found that population densities in some of the species range has decreased.*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade
Prior to *G. daudini* being formally described in 2005, this species was unknown and there is no evidence to suggest it was collected or exported under any other name. As a native reptile under the Wildlife Protection Act 1987 no person may export any wildlife from Saint Vincent and the Grenadines without the written permission of the Minister. No permit has ever been issued to collect or export *G. daudini* for commercial purposes. Only a small number of permits have been granted to scientists to catch the geckos for research purposes, but in most cases, this is on condition that the animals are released unharmed.

Illegal trade was first reported soon after the species was described and exploitation is said to have accelerated in recent years, driven by the commercial demand of the international pet trade. Desk-based research in 2016 and 2017 found more than a dozen dealers offering live *G. daudini* at prices upwards of USD 700 each from addresses in the USA, UK and the Netherlands and Germany.

This species was included in a review of species which may warrant further consideration in preparation for CoP17 (UNEP-WCMC, 2015). In this review a search of trade online found a wild-sourced pair in Germany for USD 1,450 and a single male for USD 500, these advertisements included shipping to the USA. More recent searches online found information on people selling and wishing to buy G. daudini *on forums such as Terraristik, suggesting there is continuing demand for this species in the international pet trade. A study conducted between September 2014 and December 2018 for online trade in* G. daudini *found a total of 19 advertisements with a total of 36 individuals offered for sale, the most frequent reported origin countries were Germany (39%), Netherlands (22%) and Austria (14%). A total of five individuals advertised during this period were reported to have been wild-sourced (Shepherd* et al.*, 2019).*

In addition, there is evidence online that hobbyists are claiming to be breeding G. daudini *in captivity, in some cases for commercial purposes, however the veracity of these claims cannot be verified.*

Collectors damage the gecko's environment and expose the remaining individuals to increased risk of predation and desiccation. *In 2016 the Forestry Department, Fauna & Flora International and Union Island Environmental Attackers, conducted a total of five days of surveys of the gecko population in Chatham Bay (Daltry* et al.*, 2016). A dozen geckos were found during this survey, a quarter of which had broken tails which could suggest either predator attacks or careless human captures. It was also clear from the survey that collectors had been active in the area, leaving many of the logs and rocks overturned from collecting specimens.*

Additional Information

Threats
Over-harvesting for commercial purposes is said to be a major threat to the continued survival of the *G. daudini* population in the wild. Trade is having a highly detrimental impact on the status of the species by both removing individuals from the population and causing critical damage to the species' microhabitat during the extraction process. The forests of Chatham Bay have also been degraded by several invasive alien plants and animals and are used for grazing domestic goats during the dry season.

A road constructed in 2005 improved access to Chatham Bay forest, opening this area to risk from further development. There is a proposal to extend the road through the heart of the Chatham Bay forest, which would

destroy much of the rock and litter cover that represents critical habitat for this gecko and facilitates the spread of invasive alien species.

Conservation, management and legislation
Gonatodes daudini is not yet listed as a Protected Species, but nonetheless has significant protection under the Wildlife Protection Act 1987, as a native reptile. Under this Act; no person may export any wildlife from St Vincent & the Grenadines without the written permission of the Minister.

In 2016, a Conservation Action Plan was developed by the Forestry Department, Fauna & Flora International and a diverse array of stakeholders. *This plan sets out clear objectives that aim to both enhance the survival of this species, the habitat and inclusion of the local stakeholders to benefit livelihoods (Daltry* et al*., 2016).* In accordance with the species' Conservation Action Plan, two forest wardens affiliated to the Forestry Department have been appointed on Union Island since May 2017 to assist in upholding the laws, protecting forests and wildlife; and a prosecution for illegal capture has taken place.

Artificial Propagation/captive breeding
No authorised captive breeding programmes exist. It is possible, but not confirmed that private collectors outside of the range States have bred this species in captivity for sale.

Implementation challenges (including similar species)
Gonatodes daudini is a distinctive miniature gecko, with a conspicuous white spot encircled first by black and then by red is also on the pineal eye. No other lizard has these distinctive markings. *Therefore, no implementation challenges in terms of lookalike species are anticipated.*

Potential risk(s) of a listing
There is the potential risk that if this species is included in CITES Appendix I that this could increase popularity amongst hobbyists. Advertisements of this species online do already mention that it is currently listed as Critically Endangered.

Potential benefit(s) of listing for trade regulation
The Forestry Department is currently investigating the status of lands in Chatham Bay and intends to propose that critical habitat is afforded stricter protection, ideally as a Wildlife Reserve under the Wildlife Protection Act. *It is hoped that the inclusion of* G. daudini *in CITES Appendix I would lead to better enforcement, including potentially the establishment of a protected area, and reduced chance of future development within its range (Daltry* et al*., 2016).*

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Inclusion of Grandidier's Madagascar Ground Gecko *Paroedura androyensis* in Appendix II

Proponents: Madagascar and the European Union

Summary: Grandidier's Madagascar Ground Gecko *Paroedura androyensis* is one of 21 species of Malagasy ground geckos in the genus *Paroedura*. It is endemic to southern Madagascar and can be found at elevations of up 120 m within dry-deciduous, spiny and gallery forests, but is not found in disturbed forests.

The IUCN Red List assessment of 2011 classified *P. androyensis* as Vulnerable with a declining population, but suggested that more research on the ecology and status of the species was needed. In 2011 the extent of occurrence was ca. 18,000 km^2, but it was noted that there is a continuing decline in the extent and quality of its habitat. There are very little quantitative data on population size or trend, but some studies have reported the species to be rare or infrequent based on the observations of just a few individuals during transect and pit-fall trap surveys. Breeding behavior is unknown for this species, but another species within the genus (*P. picta*) has been observed to lay two eggs per clutch with short interclutch intervals.

Deforestation caused by timber extraction for charcoal production and slash and burn agriculture are occurring throughout the species' range which is increasingly fragmenting its habitat. The species is sought after in the international pet trade. Madagascar reported exports of more than 6,000 individuals between 2013 and 2017 (ca. 1,200 per year) destined for North America, Europe and Asia.

Paroedura androyensis is protected as a category III species under Madagascar Law 2006-400, which allows for hunting and capture with a license during the hunting season and subsequent export.

Analysis: *Paroedura androyensis* does not have a restricted distribution but its habitat is fragmented and decreasing due to deforestation. There is no quantitative information on population size, although it is considered to be rare and its habitat is reportedly declining. It is apparently a sought-after species in the international pet trade and more than 6,000 individuals were reported as exported from Madagascar between 2013 and 2017, all of which are assumed to be wild. With no information on population size, densities or trends for this species it is not possible to determine what impact this level of trade might be having. Overall there is insufficient information to determine with any certainty that *P. androyensis* meets the Appendix II criteria, so the Parties can only consider the pros and cons of a precautionary listing.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS

Range
Madagascar

IUCN Global Category
Vulnerable *B1ab(iii) (2011) ver 3.1.*

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

Paroedura androyensis was described as "rare" with a decreasing population trend. Its relative abundance was found to be "infrequent" in a survey of Belomotse Forest in the Lower Onilahy River Valley Temporary Protected Area in southwest Madagascar in 2002. The species was assessed as Vulnerable in the IUCN Red List in 2011 on the basis of its extent of occurrence of 17,970 km^2, severely fragmented population, and a continuing decline in the extent and quality of its habitat. No information was found concerning the population size of *Paroedura androyensis*.

Paroedura androyensis has been observed at altitudes of 40 to 80 m above sea level in dry forests, on dry coastal rocks, in transitional, littoral and spiny forests, gallery forests, shrub land, and in riparian habitats. The species has not been found in heavily disturbed forest, and reportedly disappears after initial habitat modification. It was not reported to occur within forest edges during surveys of Malahelo nor at other nearby sites of Petriky and Andohahela.

Paroedura androyensis was recorded from spiny forest at an elevation of 120 m above sea level at a site 7.5 km northeast of Hazofotsy in central-southern Madagascar in 1995.

Paroedura androyensis is a nocturnal species which has been described as both arboreal and as being "mainly active on the ground". It is oviparous, and species of this genus usually produce a clutch of two eggs, which are buried in the ground. Feeding advice on a hobbyist website suggests that the species' diet consists of insects, mealworms and crickets.

Paroedura androyensis *has only ever been reported in studies that have been centred around more general herpetofauna assessments. From these studies, P. androyensis has been observed at very low levels in five protected areas (D'Cruze* et al., *2009; D'Cruze & Sabel, 2005; Gardner, 2012; Gardner* et al., *2016; Ramanamanjato* et al.*, 2002; Raselimanana* et al., *2012; Scherz* et al.*, 2012; Theisinger & Ratianarivo, 2015), but reports often suggest conflicting ecological descriptions of the species. In the Ifotaka North Protected Area,* P. androyensis *was described as being absent from disturbed areas of forest (Theisinger & Ratianarivo, 2015), whereas a separate study in the same protected area discovered* P. androyensis *in moderately disturbed areas (Scherz* et al.*, 2012).*

Although there is no population estimate for P. androyensis, *a density estimate of one individual per hectare was calculated during a study from western limits of the PK32 Ranobe protected area (Gardner, 2012). This figure should be treated with caution as it was calculated from observations of only two individuals from one study site. The vast majority of the information on observations of* P. androyensis *comes from presence/absence data where observations have been rare/infrequent. All of the data collected from field activities comes from transects and pit-fall traps and data were almost exclusively collected during the day. Given that* P. androyensis *is a small (max snout to vent length of 47mm) (Glaw* et al.*, 2001) nocturnal species which has been described as both terrestrial and arboreal, these general survey methods could underestimate the presence of the species.*

No specific reproductive behaviour was reported in the literature for P. androyensis*, but another species within the genus (*Paroedura picta*) has been reported to lay two hard-shelled eggs per clutch and interclutch intervals were observed to be short (Starostova* et al.*, 2010).*

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

Whilst the 2011 IUCN Red List assessment noted that the species "is not utilised or traded", evidence of trade in *Paroedura androyensis* in the EU was identified through an internet survey of non-CITES reptiles in 2009, with the species being listed on a range of online pet-trade websites in at least six EU Member States. Madagascar reported exports of 6,392 individuals of the species in the five years 2013-2017, with international trade increasing to over 1,000 specimens annually since 2015 (*these exports are assumed to wild as there are no known captive breeding facilities in Madagascar)*. The reported destinations of these exports were (in alphabetical order): Canada, Czech Republic, France Germany, Hong Kong SAR, Hungary, Japan, Malaysia, Netherlands, Russian Federation, Spain, Taiwan POC and USA.

According to the Law Enforcement Management Information System (LEMIS) database, managed by the U.S. Fish and Wildlife Service, imports of wild-caught *Paroedura androyensis* to the USA increased over the 10 years 2009-2018, with imports peaking in 2018 at around 600 individuals and a total of ca. 1900 imported over the ten year period. Low numbers of captive-bred specimens were reportedly imported from Madagascar to the US during the same period (mainly in 2010 and 2011), although captive breeding facilities for this species are thought not to exist in Madagascar.

Several websites in the EU priced an individual of this species at around EUR 30 *(USD 34)* in 2018. One online pet-trade website noted that *Paroedura androyensis* was not easily obtained, and that captive-bred specimens of this species were rare.

Two P. androyensis *specimens were also seen for sale during a physical market survey that took place in Japan in 2017 (Wakao* et al.*, 2018).*

The species is known to be collected in the south west region in the following places: Tulear II, Belalanda, Ankililoaka, Fiherenana, Antsoanabo, Mihary, Saint Augustin, Sept Lacs, and Mangily. Many of these collecting sites are located in proposed or newly created protected areas (Nouvelles Aires Protégées). Samples of the species from the pet trade were found in the Toliara region, but the original location of collection of these specimens was unknown. The species was included in a list of commercially traded exotic amphibian and reptile species in Texas in the USA 2002-2008, in which it was reported to be wild-sourced and exported [presumably re-exported] from the USA.

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

One species from the Paroedura *genus (*Paroedura masobe*) was listed in Appendix II following CoP17, but there are not thought to be any issues of resemblance between the two species.*

Additional Information

Threats

Paroedura androyensis is confined to intact forests, which are at risk from timber extraction for charcoal production and land clearance for slash and burn agriculture, both of which are ongoing throughout the species' range. *A recent study has shown that annual deforestation in Madagascar has reached 99,000 ha/year between 2010 and 2014, with 46% of forest located less than 100 m from the forest edge (Vieilledent* et al.*, 2018).*

Conservation, management and legislation

Paroedura androyensis is classified as a category III species under Madagascar Law 2006-400 on the classification of wildlife species. Category III includes game species for which hunting, and capture are only permitted with a hunting license and within the hunting season, from 1st February to 30th April. Madagascan law 2005-018 on the international trade of species of wild fauna and flora monitors and manages the country's wildlife trade with species listed Appendices I, II, and III which follow those of the Convention, and Appendix IV which includes non-CITES-listed species whose international trade is subject to national regulation and ensures Madagascar's compliance with CITES. The export of a species included in Appendices I, II, or III requires a formal export permit, whereas the export of a species in Appendix IV requires a less formal "leaving authorisation".

Reports suggest that no species-specific conservation measures were in place.

Artificial Propagation/captive breeding

There were reported to be international breeding stocks of *Paroedura androyensis* in captivity. Some hobbyist websites provide information on the breeding of the species, including that females lay clutches of two eggs every three to four weeks and that eggs should be removed and incubated at 25-30°C for 65-90 days. There are no known captive breeding facilities within Madagascar.

Implementation challenges (including similar species)

There are currently 18 species of the genus *Paroedura (endemic to Madagascar and the Comoros)*. *Paroedura vahiny* is described as similar in appearance to *Paroedura androyensis*, but has small dorsal tubercles, and is therefore smooth-skinned.

Additional, recent literature has now described a further three species within the Paroedura *genus* (*Glaw* et al., *2018*)*, bringing the total number of species in the genus to 21.*

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Inclusion of Spiny-tailed Iguanas *Ctenosaura spp.* in Appendix II

Proponents: El Salvador and Mexico

Summary: The genus *Ctenosaura*, known as Spiny-tailed Iguanas, are medium-sized omnivorous lizards that occur in lowland dry forests of Mexico and Central America. There are currently 18 recognised species, of which 11 are endemic to Mexico. Four species (*C. bakeri, C. melanosterna, C. oedirhina* and *C. palearis*) have been included in Appendix II since 2010; one additional species (*C. quinquecarinata*) has been included in Annex D of the EU Wildlife Trade Regulations since 2010.

Of the 14 species not listed in the Appendices, one is assessed as Critically Endangered (*C. oaxacana*), two Endangered (*C. flavidorsalis* and *C. quinquecarinata*), three Vulnerable (*C. clarki, C. defensor* and *C. nolascensis*), one Near Threatened (*C. alfredschmidti*), one Least Concern (*C. similis*), one Data Deficient (*C. praeocularis*), and five have not been assessed (*C. conspicuosa, C. macrolopha, C. hemilopha, C. pectinata* and *C. acanthura*). Two species are island endemics with both islands being less than 40 km^2 (*C. conspicuosa* and *C. nolascensis*), two have a range of less than 500 km^2 (*C. alfredschmidti* and *C. oaxacana*), six have ranges less than 5,000 km^2 (*C. clarki, C. flavidorsalis, C. hemilopha, C. macrolopha, C. praeocularis* and *C. quinquecarinata*), and four have ranges greater than 5,000 km^2 (*C. acanthura, C. defensor, C. pectinata* and *C. similis*). Habitats are generally fragmented for all species and the actual area of occupancy is considerably smaller than the overall range. However, several species also occur in human-dominated landscapes, and *C. similis* and C. *pectinata* are recorded as invasive species in some areas where they have been introduced.

There is very little population information for any of the proposed species, although IUCN Red List assessments for six species estimate populations to be likely less than 2,500 individuals (*C. alfredschmidti, C. clarki, C. defensor, C. oaxacana, C. nolascensis* and *C. quinquecarinata*).

Ctenosaura species are in trade for the exotic pet market, with 15 species recorded in international trade. Information on the global trade for most species is limited to imports into the USA. The USA reported imports totalling 30,000 live individuals from 1999-2012, of which 95% comprised *C. quinquecarinata* (10,000) and *C. similis* (17,000) (see below). These data indicate a shift from wild to captive-bred trade. Since 2007, the USA reported the import of ca. 700 wild-caught and ca. 7,000 captive-bred individuals. Almost all the captive-bred individuals were imported from Nicaragua and El Salvador (98%), while nearly all wild-sourced individuals were imported from Honduras and Guatemala (97%). A study in Japan found seven live individuals from four different species advertised online and 60 individuals from nine different species in a physical market survey.

Ctenosaura quinquecarinata: Native to Costa Rica and Nicaragua, and suggested to have an extent of occurrence of less than 5,000 km^2 and an area of occupancy of less than 500 km^2 when it was assessed as Endangered on the IUCN Red List in 2004. Its estimated population may be less than 2,500 individuals. In 2010, *C. quinquecarinata* were listed in Annex D of the EU Wildlife Trade Regulations to allow trade monitoring. According to the CITES Trade Database, 896 live *C. quinquecarinata* have been imported into the EU since 2010, with 592 exported from Nicaragua (of which 250 were captive-bred and the rest were from unspecified sources). USA import data shows imports into the USA between 1999–2012 totalling 10,000 live individuals. Of these, 7,000 were reported as captive-bred (all from Nicaragua) and just over 3,000 were reported as wild-caught (of which almost all were from Honduras, which is apparently not a range State).

***Ctenosaura similis*:** Native to Belize, Costa Rica, El Salvador, Guatemala, Honduras, Mexico, Nicaragua and Panama, this species is the most widespread in the genus. No population estimates exist, but it is said to be common, and was assessed as Least Concern on the IUCN Red List (2015). *Ctenosaura similis* was the *Ctenosaura* species imported into the USA in the highest numbers between 1999-2009; it accounted for 74% of all wild-caught *Ctenosaura* imports (12,323), and 22% of captive-bred imports (3,270). However, there have been no recorded imports of *C. similis* into the USA since 2009. The species can also be found for sale in Europe and Japan.

A number of other species were reportedly imported into the USA between 1999 and 2012 in smaller quantities, including *Ctenosaura alfredschmidti* (15), *C. clarki* (22), *C. conspicuosa* (50), *C. defensor* (49), *C. flavidorsalis* (6) and *C. pectinata* (205).

In addition to collection for international trade, *Ctenosaura* species are impacted by habitat loss, predation by domestic cats and dogs, and local consumption by people. *Ctenosaura* are protected by national legislation to varying degrees in six of their range States (Costa Rica, El Salvador, Guatemala, Honduras, Nicaragua and Mexico). Captive breeding is taking place in Costa Rica, El Salvador, Guatemala, Honduras, Mexico and Nicaragua, including for conservation purposes.

Differences between species in the genus were highlighted at CoP15 in relation to a proposal to list four *Ctenosaura* species, although more recent reports suggest that there are look-alike issues for all species in the genus, especially at the juvenile stage. High numbers have been exported as juveniles. An identification guide for the genus has been produced, but is intended to be a starting point rather than a conclusive document for identification.

Analysis: There is very little information on the wild populations of almost all *Ctenosaura* species, but some have been estimated to have small populations and/or limited ranges. Based on this, some species may already meet the biological criteria for inclusion in Appendix I (including but not limited to *C. conspicuosa* and *C. nolascensis)*, although reported international trade in wild-caught animals of these species is very limited.

The recorded international trade primarily comprises two species (*C. quinquecarinata* and *C. similis*) and trade in individuals reported as wild-caught appears to be decreasing. The only species not currently listed in Appendix II that appears to meet the criteria for inclusion under Annex 2a of *Res. Conf. 9.24 (Rev. CoP17)* is *C. quinquecarinata*. This species has a small population (2,500 mature individuals) and a relatively restricted and fragmented range. Although the majority of trade in this species appears to be in captive-bred individuals, given the possible small size of the population even low levels of trade may be of concern.

It is reportedly difficult for non-experts to distinguish between species of *Ctenosaura* and virtually impossible for juveniles, of which there are large numbers in trade. As some species in the genus are already included in the Appendices, the non-listed species therefore meet the criteria for inclusion in Annex 2b of *Res. Conf. 9.24 (Rev. CoP17)*.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

The CITES Standard Nomenclature Reference for species in the genus Ctenosaura already listed in Appendix II is Hollingsworth (2004) which includes all of the species listed in the SS. More recent research has suggested that C. alfredschmidti and C. defensor should be reclassified in the genus Cachryx (Malone et al., 2017). For the purpose of this analysis the aforementioned two species will be considered as being included in the genus Ctenosaura as per Hollingsworth (2004).

Range and IUCN Global Category

Species and year of description	*IUCN Global Category*	*Range*	*Appendix II-listed*
C. bakeri (1901)	*Critically Endangered (2004) B1ab(i,ii,iii,v) ver 3.1*	*Honduras*	*Yes (CoP15)*
C. oaxacana (2001)	*Critically Endangered (2004) B1ab(iii,v) ver 3.1*	*Mexico*	*No*
C. flavidorsalis(1994)	*Endangered (2004) B1ab(iii,v)+2ab(iii,v) ver 3.1*	*El Salvador, Guatemala, Honduras*	*No*
C. quinquecarinata (1842)	*Endangered (2004) B1ab(iii,v)+2ab(iii,v) ver 3.1*	*Costa Rica, Nicaragua*	*No (but included in EU Annex D since 2010)*
C. melanosterna (1997)	*Endangered (2012) B1ab(i,ii,iii,iv,v) ver 3.1*	*Honduras*	*Yes (CoP15)*

C. oedirhina (1987)	*Endangered (2010) B1ab(iii,v) ver 3.1*	*Honduras*	*Yes (CoP15)*
C. palearis (1899)	*Endangered (2011) A4acd; B1ab(i,ii,iii,iv,v); C2a(i) ver 3.1*	*Guatemala*	*Yes (CoP15)*
C. clarki (1928)	*Vulnerable (2004) B2ab(iii) ver 3.1*	*Mexico*	*No*
C. defensor (1866)	*Vulnerable (2004) A3cd; B1ab(iii,v)+2ab(iii,v) ver 3.1*	*Mexico*	*No*
C. nolascensis (1972)	*Vulnerable (2011) D2 ver 3.1*	*Mexico*	*No*
C. alfredschmidti (1995)	*Near Threatened (2004) ver 3.1*	*Mexico*	*No*
C. similis (1830)	*Least Concern (2015) ver 3.1*	*Belize, Costa Rica, El Salvador, Guatemala, Honduras, Mexico, Nicaragua, Panama*	*No*
C. praeocularis (2009)	*Data Deficient (2011) ver 3.1*	*Honduras*	*No*
C. acanthura (1802)	*Not assessed*	*Mexico*	*No*
C. conspicuosa (1919)	*Not assessed*	*Mexico*	*No*
C. hemilopha (1863)	*Not assessed*	*Mexico*	*No*
C. macrolopha (1972)	*Not assessed*	*Mexico*	*No*
C. pectinata (1834)	*Not assessed*	*Mexico*	*No*

Ctenosaura similis and Ctenosaura pectinata have also been recorded as invasive species in Florida, USA (Krysko et al., 2003). Populations of C. pectinata have also been established in Texas (USA), Honduras, Colombia and Venezuela (IUCN SSC & ISG, 2017).

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

For most species in the genus, the population size is unknown. Estimates indicate populations of less than 2,500 individuals for *C. alfredschmidti, C. defensor, C. flavidorsalis, C. nolascensis, C. oedirhina, C. palearis and C quinquecarinata*. For *C. bakeri* and *C. melanosterna* it is estimated that they have less than 5,000 individuals. Species with broad distributions (e.g. *C. acanthura, C. hemilopha, C. pectinata, C. similis*), have populations structured in small metapopulations: for example, *C. pectinata* is structured in at least five genetically isolated groups each with small distributions, and *C. similis* is separated into three groups, of which only one of these is considered to have a broad distribution (from southern Mexico to Panama), while the other two are more limited.

Studies on abundance and density of iguana populations are very scarce. Reported densities of:
- 0.93 individuals / ha for *C. quinquecarinata* in the Rio Wildlife Refuge Escalante-Chaconcete, Nicaragua.
- for *C. pectinata*, 1.06 individuals / ha for the municipality of Buenavista, Michoacán (Mexico) and 0.013 individuals / ha in Oaxaca (Mexico).
- *C. similis* was reported to be 3.1 individuals / ha in the Zamorano region, Honduras.
- an uncommonly high density was reported for *C. oaxacana* in a study conducted in an isolated subpopulation in 9 ha in Nizanda, Oaxaca, Mexico, where a density of 21 iguanas / ha was reported.

Ctenosaura species have experienced rapid population declines in recent years and have disappeared in many areas where they were previously abundant. No population increases have been reported for any of the species within the *Ctenosaura* genus. According to IUCN Red List assessments (*13 species have been assessed*), nine species have been assessed as having decreasing populations: *C. bakeri, C. clarki, C. flavidorsalis, C. oaxacana, C. oedirhina, C. palearis, C. pectinata C. quinquecarinata and C. alfredschmidti*; the population trend is unknown for three species: *C. macrolopha, C. defensor and C. praeocularis;* and five are stable: *C. acanthura, C. conspicuosa, C. hemilopha, C. nolascensis, C. similis*. Many *Ctenosaura* species are targeted for human consumption (hunting is often aimed at gravid females). Studies show that *Ctenosaura pectinata* is affected by hunting in Guerrero and Oaxaca (Mexico), and *C. similis* faces the same pressure in Honduras and Nicaragua.

*Population status and trade information for specific species is summarised below (*species already listed in Appendix II).*

C. acanthura: *Endemic to Mexico and widely distributed along the Eastern coast of Mexico (ITWG, 2016). There is no IUCN Red List assessment for this species, but a recent study suggested that populations are stable (Morales-Mavil et al., 2016).*

There is very limited evidence of C. acanthura in international trade. No evidence of trade was found in Japan and data from the U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS)

shows that only one scientific specimen was imported from Mexico and less than 50 kg of meat were imported between 1999 and 2012.

C. alfredschmidti: *Endemic to Mexico and known only from the Yucatan Peninsula in the state of Campeche (ITWG, 2016). The IUCN Red List assessment estimated the population to be fewer than 2,500 individuals in an area of less than 100 km^2 (Köhler, 2004a). Further studies have estimated the distribution of C. alfredschmidti to be 5.1 individuals/ha, but further examination suggested a density of 0.2 individuals/ha (Morales-Mavil et al., 2016). Extrapolating the data using an area of occupancy of 100 km^2 gives a population estimate of 1,964 individuals, but the author states that the density estimate is assumed to be low due to the secretive nature of the species (Morales-Mavil et al., 2016).*

There is limited evidence of trade in this species. The US LEMIS database reports that from 1999-2012, 15 live C. alfredschmidti were imported to the USA with six of these having been collected from the wild.

****C. bakeri:*** *This species was listed on CITES Appendix II in 2010. CITES trade data since the listing show that only four live C. bakeri have been exported for zoological purposes from Honduras (reported by the importer), three of which were born in captivity (source code F) and one of which was a pre-Convention live individual (source code O). Online survey data (from the SS) shows C. bakeri advertised for sale online in the USA on two websites (number of individuals not reported). Physical survey data from Japan showed two C. bakeri for sale at a reptile fair (unpublished data in Wakao et al., 2018).*

C. clarki: *Endemic to Mexico, specifically the Balsas-Tepalcatepec basin in the states of Michoacán and Guerrero (ITWG, 2016). The Red List assessment estimated a population of less than 2,500 individuals (but no population data appear to be available) with isolated subpopulations and an area of occupancy of less than 2,000 km^2 (Köhler, 2004b). No further population data could be found.*

There is limited evidence of this species in trade. The US LEMIS database reports that between 1999-2012, 22 live C. clarki were imported into the USA, with 21 of these exported from Germany. Five individuals were observed on sale during a survey of reptile fairs in Japan in 2017 (unpublished data in Wakao et al., 2018) and the SS shows online advertisements in France and Germany (no quantities reported).

C. conspicuosa: *This species is a Mexican island endemic (range limited to San Esteban Island (40 km^2) and has also been introduced to Cholludo Island) (ITWG, 2016). There is no Red List assessment for this species and no population data could be found in the literature.*

There is some evidence of C. conspicuosa in international trade. The US LEMIS database reports a total of 50 live imports of this species to the USA between 1999-2012, of which all were captive-bred from Nicaragua. In 2017, six C. conspicuosa were found to be available at reptile fairs in Japan (unpublished data in Wakao et al., 2018).

C. defensor: *Endemic to Mexico, specifically the Yucatán Peninsula in the Mexican states of Campeche and Yucatán (ITWG, 2016). The IUCN Red List estimates a population of less than 2,500 individuals and area of occupancy less than 2,000 km² (but no population data appear to be available) (Köhler, 2004c).*

There is limited evidence of this species in trade. The US LEMIS database reports that between 1999-2012, 49 live C. defensor were imported to the USA with three having been collected from the wild. During a survey in Japan in 2017, one live individual was found for sale online and nine were observed at reptile fairs (unpublished data in Wakao et al., 2018). The SS shows online adverts for C. defensor in Austria and Germany.

C. flavidorsalis: *Species described in 1994 and has a range from eastern Guatemala to eastern El Salvador and to south-western and south-central Honduras (Salvador et al., 2001). The only population estimate for this species is from the IUCN Red List assessment which reports that the population is perhaps less than 2,500 mature individuals (Köhler, 2004d), but this is not based on any empirical evidence. No further population data could be found and it was reported in 2012 that there were no research or conservation activities ongoing related to C. flavidorsalis (Stephen et al., 2012).*

Very limited evidence of this species in trade was found. US LEMIS data shows that between 1999–2012, six live, captive-bred C. flavidorsalis were imported into the USA. There was no evidence of C. flavidorsalis in trade in either Europe or Japan.

There is no current evidence of this species being significantly consumed in range States, but there is anecdotal evidence that in El Salvador the fat of C. flavidorsalis is used medicinally to treat sprains and cure young children of illness (Stephen et al., 2012).

C. hemilopha: *Endemic to Mexico and is found in the southern part of the Baja California peninsula and on Isla Cerralvo (Davy et al., 2011; ITWG, 2016). There is no IUCN Red List assessment, but a study in 1989 suggested a density of 2 individuals per hectare (Alvarez et al., 1989) and from the distribution presented by the Iguana Taxonomy Working Group (ITWG, 2016), C. hemilopha has a range much larger than 10,000 km^2.*

There is almost no evidence of C. hemilopha in trade, with only one scientific specimen of C. hemilopha being imported to the USA from Mexico in 2010 (based on US LEMIS data). No evidence of trade in either Europe or Japan.

C. hemilopha is consumed for its meat and is thought to have medicinal properties to cure whooping cough (Grismer, 2002).

C. macrolopha: *Was regarded as a subspecies of C. hemilopha until 1999, and is endemic to north-western Mexico (ITWG, 2016). There is no Red List assessment for this species and no population data could be found in the literature.*

There is limited evidence of C. macrolopha in trade; no live C. macrolopha were imported into the USA between 1999-2012 according to the US LEMIS database. Two live C. macrolopha were observed at a Japanese reptile fair in 2017 (unpublished data in Wakao et al., 2018).

****C. melanosterna:*** *This species was listed on CITES Appendix II in 2010. The only trade recorded in the CITES Trade Database was of 318 wild-caught C. melanosterna imported into the USA from Honduras for scientific purposes.*

C. nolascensis*: This species is an island endemic with a range limited to San Pedro Nolasco Island (5 km^2) (ITWG, 2016). C. nolascensis was considered as a subspecies of C. hemilopha until 1999 (Grismer, 1999). The IUCN Red List assessment states that the population is unknown but estimates it to be less than 2,500 individuals, and cites encounter rates of 80 individuals/km which suggests that C. nolascensis is found at high densities on the island (Reynoso & Pasachnik, 2012).*

No evidence could be found of C. nolascensis in international trade, but care guides with photo evidence of the species in captivity can be found (Köhler et al., 2004). The extent of the numbers being kept in captivity is unclear.

C. oaxacana*: Endemic to the state of Oaxaca in Mexico (Köhler, 2004e; Valenzuela-Ceballos et al., 2015). The species was separated from C. quinquecarinata in 2001 (Kohler & Hasbun, 2001 in Hasbun & Kohler, 2009). The IUCN Red List assessment states that the population is unknown but is perhaps less than 2,500 mature individuals with an area of occupancy of less than 100 km^2 and is thought to occur in severely fragmented populations (Köhler, 2004e). A study in 2012 estimated a population density of 33.7 individuals/ha (Rioja et al., 2012).*

There is no evidence of this species being imported to the USA (based on US LEMIS data), limited evidence of this species being in trade in Europe (one advert on a Spanish website in SS), and one observation of a wild-caught individual at a reptile fair in Japan (unpublished data in Wakao et al., 2018). Habitat loss and hunting for food appear to be the main threats (Diaz-Juarez & Reynoso, 2015; Rioja et al., 2012).

****C. oedirhina:*** *This species was listed on CITES Appendix II in 2010. The only trade recorded in the CITES Trade Database was of 399 wild-caught C. oedirhina that were imported into the USA from Honduras for scientific purposes.*

****C. palearis:*** *This species was listed on CITES Appendix II in 2010. The only trade recorded in the CITES Trade Database was of five live C. palearis exported from the USA to Canada that that had reportedly been seized/confiscated (source code I).*

C. pectinata: *Endemic to Mexico, with a distribution covering western Mexico (ITWG, 2016). There is no IUCN Red List assessment for C. pectinata, but it does not have a limited area of distribution and densities of up to 12.3 individuals/hectare have been recently recorded (Gómez-mora & Alvarado-díaz, 2012).*

A total of 205 live individuals were imported into the USA between 1999 and 2012 according to US LEMIS data. Of these imports 80 were wild-caught and 125 were captive bred (all from from Mexico). The USA also reported exports of live individuals totalling 529 individuals over the same time period with 98 of these reportedly wild-caught. Data from a market survey in Japan in 2017 observed one C. pectinata for sale online and 30 individuals for sale at reptile fairs (unpublished data from Wakao et al., 2018). The SS shows evidence of online adverts in Germany, France and the UK advertising C. pectinata for sale.

C. praeocularis: *Described in 2009 and has a range that is limited to south-eastern Honduras (Carlos. Roberto. Hasbun & Kohler, 2009; ITWG, 2016). Due to the recent description of the species, it is assessed on the IUCN Red List as Data Deficient and has no population estimate but an estimated range of 1,200 km^2 (C. R. Hasbun & Pasachnik, 2013). In 2012 it was reported that there were no research or in situ conservation programs aimed at C. praeocularis (Stephen et al., 2012). No evidence of trade in C. praeocularis could be found.*

C. quinquecarinata: *Present in Nicaragua and Costa Rica and suggested to have an extent of occurrence of less than 5,000 km^2, an area of occupancy of less than 500 km^2 and fewer than 2,500 mature individuals remaining in severely fragmented populations (Köhler, 2004f). A study by Chamorro estimated a population density of 0.93 individuals/ha (Chamorro, 2010).*

In 2010, C. quinquecarinata were being imported in such numbers to the EU that they were added to Annex D to allow trade monitoring (European Commission, 2010). According to the CITES Trade Database, 896 live C. quinquecarinata have been imported into the EU since 2010, with 592 exported from Nicaragua (of which 250 were captive-bred and the rest were from unspecified sources). US LEMIS data shows imports into the USA between 1999 – 2012 totalling 10,660 live individuals. Of the imports into the USA, 7,179 were reported as being from captive-bred sources (all from Nicaragua) and 3,431 were reported as wild-caught (of which 3,296 were from Honduras (not a range State)). The general trend in imports of this species into the USA has been a reduction in the number of wild-caught Ctenosaura and an increase in the number of captive-bred individuals. For example, from 2007-2012, 155 wild-caught C. quinquecarinata were imported, versus 6,783 captive-bred individuals over the same time period. In Japan, a survey in 2017 showed three individuals for sale online and one individual for sale at a reptile fair (unpublished data from Wakao et al., 2018).

it has been reported that C. quinquecarinata are locally persecuted out of fear that they are poisonous (Stephen et al., 2012). Also, like all species within the genus, it is believed to be affected by habitat destruction/degradation (Köhler, 2004f; Stephen et al., 2012).

C. similis: *The most widespread of the Ctenosaura species with a range throughout Central America, and has also been introduced to the USA and the Bahamas (ITWG, 2016). Although no population estimate exists for this species, it was reported to be common throughout its range in the IUCN Red List assessment (Pasachnik, 2015). It was also reported in 2012 that although there was continuing international and domestic trade, this had not endangered populations (Stephen et al., 2012).*

C. similis was the Ctenosaura species imported into the USA in the highest numbers between 1999-2012 according to US LEMIS data; it accounted for 74% of all wild-caught Ctenosaura imports (12,323), but only 22% of captive-bred imports (3270). However, there have been no recorded imports of C. similis into the USA since 2009. Evidence of trade in Europe can also be found in the SS and seven individuals were observed in an online and physical market survey in Japan (unpublished data in Wakao et al., 2018).

The main species in international trade appear to be C. quinquecarinata and C. similis, both of which have reported total levels of trade of more than 10,000 live individuals since 1999. Ctenosaura pectinata has reported levels of international trade of more than 200 individuals since 1999; all other species within the genus have reported levels of international trade of less than 100 individuals since 1999. These figures should be treated with some caution as the online and physical market surveys in Japan and Europe only give a snapshot of the trade during a limited time period, while CITES Trade Database data only shows reported figures from 2010 onwards (and will not be complete for non-listed species).

National use

The iguanas of the genus Ctenosaura are commonly hunted locally for the consumption of their meat and eggs. Gravid females are consumed more frequently. In some regions they are affected by collection for medicinal use; it is believed that they can cure colds and are used for the treatment of sprains or headaches. Some are sold as handicrafts and skins, such as *C. pectinata*. It has been reported that *C. similis* and *C. quinquecarinata* are used as feed for domestic animals.

International trade
International trade in *Ctenosaura* spp. has been documented. In addition to the species already listed in Appendix II, trade in *C. quinquecarinata* is also recorded in the CITES Trade Database (*records are for imports into the EU as the species has been listed in Annex D of the EU Wildlife Trade Regulations since 2010)*, and in the greatest numbers. Of these records, 5% were bred in captivity and the same percentage were wild-sourced (the source of the remaining were unspecified). *According to the CITES Trade Database, 896 live individuals of C. quinquecarinata were imported into the EU since 2010, with 592 exported from Nicaragua (of which 250 were captive-bred, the remainder were from an unspecified source). There is also evidence of nine Mexican Ctenosaura spp. having been advertised as for sale within the EU (Auliya et al., 2016), but the specific species and numbers advertised were not reported.*

In 2018, Mexico carried out consultations with 23 countries regarding international trade in *Ctenosaura* spp. Responses were received from 11 countries/territories, of which five reported no known trade in *Ctenosaura* spp. (Canada, China, France, Hong Kong SAR and Malaysia) and six reported trade in one or more species (Austria, El Salvador, Germany, Netherlands, Spain and UK). Mexico also carried out a trade analysis on trade in *Ctenosaura spp.* using data from the US Fish & Wildlife Service's LEMIS database (date range 1999-2018). Further information from these analyses can be found in the Supporting Statement.

According to the US Fish & Wildlife Service's LEMIS database, 95% of all Ctenosaura imports from 1999-2012 were of two species: C. similis and C. quinquecarinata. Over the period 1999-2013, Ctenosaura imports to the USA peaked in 2004 and subsequently declined considerably, although there was an overall increasing trend from 2008 to 2012; there was also a shift from wild-caught to captive-bred imports over this period (Figure 1). 98% of captive bred individuals were imported from Nicaragua and El Salvador, while 97% of wild-caught individuals were imported from Honduras and Guatemala.

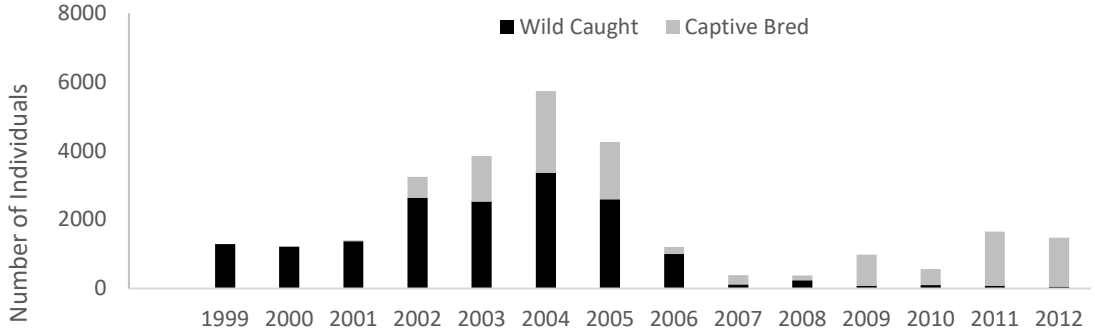

Figure 1: *Imports to the USA of wild and captive Ctenosaura spp. according to U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS), 1999-2012.*

A 2018 web search found 30 sites with *Ctenosaura* advertised for sale (from websites in Europe and North America). Ten species were advertised: *C. acanthura*, *C. bakeri*, *C. clarki*, *C. conspicuosa*, *C. defensor*, *C. melanosterna*, *C. oedirhina*, *C. pectinata*, *C. quinquecarinata* and *C. similis*. Mainly juveniles were offered for sale and prices ranges from USD 20 – 3,500. Nine species were also advertised for sale on Facebook from 2013–2018: *C. alfredschmidti*, *C. bakeri*, *C. defensor*, *C. oaxacana*, *C. melanosterna*, *C. quinquecarinata*, *C. palearis*, *C. pectinata* and *C. similis*. Prices on Facebook ranged from USD 15 – 1,200.

In Japan, a market survey in 2017 found a total of seven live Ctenosaura spp. from four species (C. defensor, C. pectinata, C. quinquecarinata and C. similis) available online, and physical visits to pet shops and reptile fairs observed 61 live indiviuduals from a total of nine species (C. bakeri, C. clarki, C. conspicuosa, C. defensor, C. macrolopha, C. oaxacana, C. pectinata, C. quinquecarinata and C. similis) (unpublished data in Wakao et al., 2018).

Illegal trade
Mexico: *Ctenosaura* iguanas have been illegally captured in the country for their meat and eggs for a long time. According to the Federal Attorney's Office for the Environment and Natural Resources, seizures registered for the genus *Ctenosaura* from 2008 to 2018 included a total of 159 confiscated individuals (131 *C. pectinata*, 16 *C. quinquecarinata*, 8 *C. defensor* and 4 *C. similis)*.
Guatemala: Some of the largest seizures that occur in the country are illegal shipments of *C. similis* and *Iguana iguana*, the species most consumed as food, some of which are transported to El Salvador. It has been reported that *C. acanthura* is exploited in the country and can be acquired illegally.

Honduras: Between 2005 and 2008, Honduran authorities confiscated 220 *Ctenosaura* individuals, which was the third most confiscated genus.
Nicaragua: In 2008, the Nicaraguan authorities confiscated a total of 707 illegal wild *Ctenosaura* and *Iguana iguana,* which made them the most confiscated taxa in that year. From 2005 to 2010, more than 1,561 *Ctenosaura* were seized in Nicaragua.
El Salvador: Illegal shipments of *Ctenosaura* from Honduras and Guatemala have been recorded, which were confiscated by the authorities. From 2006 to 2008, 3,611 *Ctenosaura spp*. and *Iguana iguana* were seized according to official data. In 2009, 241 *Ctenosaura were* seized in El Salvador, exported from Nicaragua, and in 2010, 45 black iguanas (*C. similis*) were reported as being seized by the National Police. The CITES Authorities of El Salvador stated that there have been six seizures of *C. similis* in the last 10 years, totalling 226 individuals.

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

There is similarity between species of the genus, and it is difficult to differentiate species even by expert biologists. It is common for confiscated specimens to have been assigned the wrong species name, generally of more common species. The offspring of large species of the genus *Ctenosaura* may become confused with those of *Iguana iguana (which is currently listed in Appendix II along with other species of the genus Iguana). High numbers have been exported as juveniles (Stephen et al., 2012).*

Although the Supporting Statements for C. bakeri, C. melanosterna, C. oedirhina and C. palearis at CoP15 (CoP15 Props 11 and 12), highlighted the differences between these species and other species in the genus, recent reports suggest that there are look-alike issues for all members of the Ctenosaura genus with species being difficult to tell apart, especially at the juvenile stage (Stephen et al., 2012). An identification guide has been produced, but it has been designed to be a starting point for authorities rather than a conclusive document for identification (in Stephen et al., 2012).

Additional Information

Threats

Habitat in the original range of all *Ctenosaura* species has been modified by degradation, fragmentation and destruction, with the exception of *C. conspicuosa* and *C. nolascensis*, which are occur on islands that do not authorize human settlements or changes in land use. Species that inhabit very remote places (e.g. *C. hemilopha*, *C. macrolopha*, *C. melanosterna*) suffer the effects of habitat destruction less than species such as *C. acanthura*, *C. pectinata* and *C. similis* that are highly affected by agriculture and livestock. Some species like *C. bakeri* and *C. oedirhina* are being seriously affected by tourism development. Despite the change in land use, many species can coexist with humans (but are affected by hunting). Species such as *C. defensor* and *C. alfredschmidti* are the exception, since they need pristine habitat for their survival.

The main threats to the genus are habitat destruction, hunting for meat and skin, the illegal national and international pet trade, hybridisation with similar species, elimination by invasive species (e.g. cats), and hunting due to being thought to be a poisonous species. The dry forests where smaller species live consistently suffer changes in land use due to introduction of livestock; this is important for *C. alfredschmidti*, *C. defensor*, *C. clarki, C. flavidorsalis, C. quinquecarinata and C. defensor*.

The illegal sale and consumption of *Ctenosaura* iguanas in Mexico and Central America is common, mainly for food consumption, including traditional consumption (e.g. *C. pectinata* during Easter and during the Guelaguetza, in Oaxaca, Mexico), and to a lesser extent for the pet trade.

Conservation, management and legislation

The species *C. palearis, C. melanosterna*, *C. bakeri* and *C. oedirhina* were listed in CITES Appendix II in 2010. *Ctenosaura quinquecarinata* was listed in Annex D of the EU Wildlife Trade Regulations in 2010.

Mexico: The capture, reproduction and export of the 11 Mexican species of *Ctenosaura* are regulated by the General Law of Wildlife, which establishes strict criteria for the use and conservation of endangered wild species. NOM- 059-2010-SEMARNAT classifies 7 species in the following categories: endangered (*C. defensor*); threatened (*C. clarki, C. oaxacana, C. similis, C. pectinata*); and under special protection (*C. acanthura and C. hemilopha,* which includes *C. conspicuosa, C. macrolopha and C. nolascensis*). Those classified as endangered can only be used for conservation purposes and breeding in captivity. There are authorisations for the capture of *C. pectinata,* and the breeding in captivity and use of *C. pectinata* and *C. defensor.*

Costa Rica: The Wildlife Conservation Law prohibits the hunting and extraction of threatened wildlife species except those from registered sustainable captive breeding facilities. All imports and exports of wild fauna are forbidden.

Guatemala: The Law on Protected Areas prohibits the collection, hunting trade and export of endangered fauna. Guatemala has an official list of endangered species that has three categories: *C. palearis* is included in Category 2 that only allows its use for scientific purposes and for reproduction with conservation purposes; *C. similis* is in Category 3 that restricts use including trade.

Honduras: The General Law on the Environment regulates the use and conservation of wildlife. Exports of *C. melanosterna*, *C. bakeri* and *C. oedirhina* have been restricted, and captive breeding facilities for *C. melanosterna* and *C. bakeri* have been created for conservation projects and reintroduction.

Nicaragua: This is the only country with a law that focuses on iguanas (Decree 547), which prohibits the capture or trade and prohibits the export of *C. similis* and its products such as eggs. The conservation and trade of other species of iguanas such as *C. quinquecarinata* is regulated by the Mandatory Nicaraguan Technical Standard for Domestic Wildlife Trade (NTON 05011-01), Law 217, General Law of the Environment and Natural Resources, which allows for the allocation of hunting closures. Nicaragua only allows the export of *C. quinquecarinata* from two captive breeding facilities.

El Salvador: The Wildlife Conservation Law regulates the hunting, collection and marketing of wild species.

Artificial propagation/captive breeding
There are few captive breeding facilities for *Ctenosaura* spp. in the Central American region because these species are reportedly very aggressive and difficult to maintain, and also because there is a much greater demand for Green Iguanas *Iguana iguana*.
Mexico: Eight captive breeding facilities have been created for *C. pectinata* and one for *C. defensor* since 2000. In the period 2000-2015 there were 6,016 individuals of *C. pectinata* and seven *C. defensor* produced.
Honduras: There is a captive breeding facility for *C. bakeri* used to increase wild populations and a further captive breeding facility has been proposed for *C. melanosterna* for reintroduction purposes.
Nicaragua: There are two captive breeding facilities for *C. quinquecarinata,* one of which claims to produce more than 6,000 individuals per year. A captive breeding facility for *C. similis* is used exclusively for the purposes of research.
Guatemala: Has a program of reproduction in semi-captivity of *C. palearis* for conservation in a private reserve.

Further captive breeding facilities to supply the pet trade have been reported in El Salvador and Costa Rica (Stephen et al., 2012).

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Inclusion of Spider-tailed Horned Viper *Pseudocerastes urarachnoides* in Appendix II

Proponent: Islamic Republic of Iran

Summary: The Spider-tailed Horned Viper *Pseudocerastes urarachnoides* is a recently described (2006) viper species, known only from a few locations in the Zagros Mountains of western Islamic Republic of Iran (Iran). The species may also be present in suitable habitat in adjacent areas of Iraq. Its unique tail resembles a spider and is used to lure insectivorous birds. Little information is available on its biology, but it is mainly found in hilly areas and is associated with deep cracks in limestone sediments.

There are no population size estimates and population trends are lacking, although it is considered to be rare based on field observations.

In Iran the hunting, killing or catching of all wild animals (including reptiles) is prohibited. Any export of live wild animals without a licence or approval from the Department of Environment is also prohibited. Despite this there is some evidence of international trade in *Pseudocerastes urarachnoides,* although it is limited to photographic evidence of the species in captivity on social media, a survey of German pet keepers for the German Government in 2018, and a single conversation on social media in 2017. Only described in 2006, the uniqueness of the species may create increasing demand in the future in the pet trade. Similar species do not appear to be traded in large volumes.

The species is said to be impacted by illegal collection for the pet trade, habitat destruction and future climate change. It is reported that the species is sometimes killed when encountered by local communities.

Analysis: *Pseudocerastes urarachnoides* has a small reported range in western Iran, based on observations of a few specimens. There are no population estimates and population trends are not available, although it is considered to be rare. Evidence of trade is limited although there are concerns that the species' uniqueness may attract demand in the future. All current trade from Iran is illegal. Given limited evidence of trade, it seems unlikely that the species meets the criteria for inclusion in Appendix II in *Res. Conf. 9.24 (Rev. CoP17)*. Iran could consider an Appendix III listing.

Other Considerations: If the proposal is accepted (or Iran lists the species in Appendix III), as it is nationally protected and therefore trade is illegal, Iran may wish to reflect this by putting in place a voluntary zero export quota that would be listed on the CITES website and empower re-exporting and importing countries to assist law enforcement.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Pseudocerastes urarachnoides is one of three species in the genus *Pseudocerastes (Pseudocerastes urarachnoides, Pseudocerastes persicus* and *Pseudocerastes fieldi).*

Range

Islamic Republic of Iran.

IUCN Global Category

Data Deficient (2008) ver 3.1.

Biological and trade criteria for inclusion in Appendix II (*Res. Conf. 9.24* (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

Pseudocerastes urarachnoides is known only from a few localities in the Zagros Mountains of western Iran. *Based on the extent of potentially suitable habitat, the species is presumably more widely distributed within the Zagros Mountains and is likely to be present in adjacent areas of Iraq (Fathinia* et al., *2009).* It is mainly found in hilly areas and is associated with deep cracks in limestone sediments.

The political instability in Iraq and its border with Iran, together with low detection of these vipers hampers the precise knowledge on the distribution of this species (del Marmol et al., *2016).*

Biology

Described in 2006, there is little information on the biology of the snake. It is an oviparous, primarily diurnal species with sexual activities occurring nocturnally. It emerges from hibernation in early April and enters hibernaculum in late November. It has a bimodal activity pattern with peaks in spring and late summer/early autumn.

Adult specimens have a total length of 55–86 cm and a short tail (TL/T=9.65), few pairs of subcaudals (15 in the known specimens), the distal pair forming an oval knob-like structure; lateral dorsal caudal scales projected to form elongate “appendages” along the sides of the terminal knob. The scales are more prominently rugose than in any other snake found in Iran. The tail of *Pseudocerastes urarachnoides* resembles a spider to attract small birds.

Population size and trend

There is no estimate available for population size of this species from recorded sites. Evidence based on field observations indicates that the species is rare, and the lack of reliable data prevents estimation of the total population *and population trends (Anderson & Papenfuss, 2009). A complete decline of a 10 individual population was observed after exposure to people (B. Fathinia,* in litt., *2019).*

Trade

The SS reports that the principal and immediate threat to the species is the illegal collection to meet the demand for national and international trade. There is no use of the species for traditional medicines or other purposes in Iran and there have been no legal captures and exports of this species for commercial purposes. No permits have been issued in this regard.

There is a rumoured use of this species in the pet trade—detailed records for the volume of illegal trade are lacking, the SS suggests there are strong indications that this species is illegally collected and smuggled out of the country. The SS provides photos from Germany of *Pseudocerastes urarachnoides* on community networks (possibly six individuals seen in the photo). It is reported that the species is kept in European terrariums *(this is based on a personal communication*). The SS states *Pseudocerastes urarachnoides* was confirmed to be kept in private households based on a recent survey among German pet keepers, on behalf of the German Government. The SS also states that based on a post in a closed Facebook group, several specimens are kept in captivity, with one pair sold in 2017.

A viper breeder in El Paso, Texas, USA appears to have four males and six females of the species based on social media posts (Instagram), while reporting the presence of four or five specimens in Europe. Their origin is unclear, as is the reliability of social media posts.

The SS states that the collection for the pet trade has already resulted in the local extinction of some populations of the other two species of the *Pseudocerastes* genus, *Pseudocerastes persicus* and *Pseudocerastes fieldi*. *These are based on IUCN Red List assessments. For* Pseudocerastes persicus *the IUCN Red List assessment states “over-collection is a serious threat and is causing significant declines in local populations. Continued over-exploitation could cause this species to be threatened, hence ongoing monitoring of its status is required” (Ananjeva* et al., *2010). For* Pseudocerastes fieldi *the IUCN Red List assessment mentions that over-collection for the pet trade in Egypt is a threat (Amr* et al., *2012).*

Between 1999 and 2014, 58 live specimens of Pseudocerastes persicus *were imported into the USA from Egypt according to USA import data, while 14 live specimens of* Pseudocerastes *species reported as False Horned Vipers were also imported (US Fish & Wildlife Service Law Enforcement Management Information System (LEMIS) data).*

Additional Information

Threats

The principal and immediate threat to the species is believed to be illegal collection to meet demand for national and international trade, while habitat destruction and global warming effects such as drought may affect survival of the species. It is presumed that animals are also killed when encountered by villagers because of fear and superstitions. *Fathinia (*in litt., *2019) reports that when talking to local communities, they report the killing of this viper. The IUCN Red List assessment states that threats to the species are not known, although because it is an unusual snake a threat may be over-collection for the international pet trade in the future (Anderson & Papenfuss, 2009).*

Conservation, management and legislation

In accordance with the Environmental Protection and Enhancement Act (1974) and the Executive By-Law on the Game and Fish Law (1967) any hunting, killing or catching of all wild animals, birds and reptiles as well as fishing, killing or catching aquatic animals is prohibited. Any export of live wild animals without a licence or approval from the Department of Environment (DoE) is also prohibited. The DoE has recently listed *Pseudocerastes urarachnoides* in the "nationally endangered species" category. Any illegal and not permitted collecting of this snake is subjected to a fine of IRR 50 million (more than USD 500).

The species lacks a formal "conservation action plan" (CAP) from the DoE in Iran. Some projects on basic information collection have been started and the DoE plans to prepare a CAP as soon as possible.

Implementation challenges (including similar species)

Pseudocerastes urarachnoides closely resembles *Pseudocerastes persicus* in the dorsal scale characters, which distinguish that species from *Pseudocerastes fieldi*, apart from the greatly shortened tail and the elaborate caudal appendage, which set it apart from both.

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Transfer of Bourret's Box Turtle *Cuora bourreti* from Appendix II to Appendix I

Proponent: Viet Nam

Summary: Bourret's Box Turtle *Cuora bourreti* is a medium-sized terrestrial forest turtle that can reach 18 cm shell length. *Cuora bourreti* has historically been consumed for food, however in the last decade large-scale consumption has largely ceased with most animals collected now sold into the pet trade or for traditional medicine. Based on observations in captivity, the species matures at 10–15 years and lays a single clutch of one to three eggs annually.

Cuora bourreti has been listed in CITES Appendix II since 2000 under a genus-level listing. *Cuora bourreti* was previously considered a subspecies of *C. galbinifrons,* however it was recognised as a species in the standard nomenclature reference adopted at CoP17 in 2016. Before this split was recognised under CITES, *C. galbinifrons* was included in the Periodic Review after CoP16 and the Animals Committee recommended the transfer of *C. galbinifrons* (at that time including *C. bourreti* and *C. picturata)* to Appendix I.

Cuora bourreti is known from central Viet Nam and the adjoining province of Savannakhet in Lao People's Democratic Republic (PDR), although confusion with *C. galbinifrons* means its range could be smaller than currently believed. It inhabits upland, moist, closed-canopy evergreen forest, and habitat loss and degradation are considered a significant but predominantly localised threat to the species. Unsustainable collection is considered to be the main threat given that *C. bourreti* is a long-lived, late-maturing species with limited annual reproductive output and high juvenile mortality.

The species was assessed as Critically Endangered in 2015 on the basis that documented market volumes in China and Hong Kong Special Administrative Region (SAR) indicated a collapse of populations of over 90% over the past 60 years (three generations) and decline was predicted to continue for the next 20 years. This was inferred from market observations reported as predominantly *C. galbinifrons,* which at the time was also considered to include what are now accepted as *C. bourreti* and *C. picturata.* The population of *C. bourreti* is estimated to be between 10,000–20,000 individuals in the wild and the species is considered rare.
In 2013 a zero export quota for wild specimens for commercial purposes was adopted for the listing of *C. galbinifrons* (including *C. bourreti*) and there has been no legal trade in *C. bourreti* reported in the CITES Trade Database since then. Live specimens are observed for sale online, often stating that they are from captive-bred populations. However, it continues to be regarded as a difficult, sensitive species to breed successfully in captivity, reproducing slowly with small clutch sizes. Low numbers have been observed in farms in China, and juvenile animals are said to be raised in villages in range States for sale into trade, although they suffer high mortality rates.

The species is legally protected from exploitation in both range States. Illegal trade is considered to continue to the main destination markets of China and Hong Kong SAR. A small number of captive-bred specimens were observed in Hong Kong SAR markets between 2014 and 2018.

Analysis: No legal trade in *Cuora bourreti* has been reported since a zero export quota was put in place in 2013. Illegal trade is thought to occur, but it is not clear on what scale. The population size was estimated to be between 10,000 and 20,000 individuals, which could be considered small for a low productivity species such as this. It does not appear to have a restricted range, although issues with misidentification with *C. galbinifrons* could mean that the distribution is smaller than previously thought. The species was categorised as Critically Endangered in 2015 based on a decline of 90% in the past three generations. It would therefore appear to meet the criteria for inclusion in Appendix I.

Other Considerations: At CoP16 a zero export quota for wild specimens for commercial purposes was adopted with the listing for *Cuora galbinifrons* and therefore all trade in wild specimens of *C. bourreti* is already illegal. Benefits of an Appendix I listing are not likely to be realised unless enforcement efforts are increased.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS), text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Cuora bourreti was traditionally considered a subspecies of *Cuora galbinifrons;* however, more recent research has treated *bourreti* as a full species, including the nomenclature standard reference for the *Cuora galbinifrons* group adopted at CoP17, which recognises *Cuora bourreti* (and *C. picturata*) as full valid species.

Range

Lao PDR, Viet Nam

IUCN Global Category

Critically Endangered A2bd+4bd (2015) ver. 3.1.

Biological criteria for inclusion in Appendix I

A) Small wild population

It was estimated that the population of Cuora bourreti *is between 10,000–20,000 individuals at most due to its large geographic distribution, and it is one of the turtle species at the highest risk of extinction (TCC, 2018).*

The SS notes that no absolute population size numbers or estimates are available for *Cuora bourreti,* and only anecdotal relative population density data. All recent indications are that the species requires an extensive search effort to encounter. During field surveys in Lao PDR in 1993–1999, encounter rates were of the order of one turtle per three months in the field for a herpetologist, and one *Cuora galbinifrons* (or *C. bourreti*) per day when working with a trained turtle-hunting dog in prime turtle habitat.

B) Restricted area of distribution

Cuora bourreti is known from central Viet Nam as well as from adjoining Savannakhet Province in Lao PDR. *Cuora bourreti* inhabits upland, moist, closed-canopy evergreen forest. The species is predominately terrestrial and is not specifically associated with forest streams. In general, forest cover in Viet Nam declined between 1943 (43% land area) and 1973 (29% land area). *Reforestation attempts have been made but efforts have been focussed of monocultures of fast-growing exotic species (McNamara,* et al., *2006) which cannot be considered suitable habitat for this species (R. Struijk,* in litt., *2019).*

In Viet Nam the species' range is from Ha Tinh Province southward to Kon Tum Province, however details of its distribution are not certain due to misidentifications with Cuora galbinifrons *(Stuart* et al., *2011). Stuart* et al., *(2011) consider that the extended range of* C. bourreti *northward to Nghe An Province is likely an error as the authors listed this discovery in the same distribution of* C. galbinifrons *and field-collected specimens from this area are* C. galbinifrons*. Therefore, the distribution of this species may be smaller than previously reported.*

In 2009, Stuart et al., *(2011) documented* C. bourreti *in Lao PDR for the first time. During their survey the authors made two observations of* C. bourreti*, one was in the possession of a turtle hunter who was camping at Laving-Laveun within the Laving-Laveun Provincial Protected Area in Savannakhet Province and the second was observed by a village resident in the Vilabouli District, Savannakhet Province.*

C) Decline in number of wild individuals

Cuora bourreti has been subject to intensive exploitation since the 1990s across its range, primarily for consumption and secondarily for the pet and farming/aquaculture trades. Trade volumes have collapsed in recent years and field surveys indicate the species to be rare, with an estimated population collapse of over 90% over the past 60 years (three generations, at 20 years per generation time), and predicted to continue for the next 20 years, is likely an underestimate (McCormack & Stuart, 2016).

Extensive survey work has been undertaken in Viet Nam between 2009–2012 focused on determining the range and priority habitat for taxa of the *Cuora galbinifrons* group, with a focus on *C. bourreti* and *C. picturata.* Anecdotal information from interviews throughout the range has found that historic quantities of *C. bourreti* available for collection in the forest have been greatly reduced, with many hunters stating that while the species was common 7–15 years ago, it is now increasingly difficult to find. During surveys in 2006 in and around Song Thanh Nature Reserve, Quang Nam Province, local hunters claimed that numbers of *C. bourreti* had already been seriously depleted; from being able to catch 20 animals a day in the mid-1990s they were able to obtain only a few animals in a week.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Cuora bourreti *has historically been consumed locally for food as part of a subsistence diet, however, in the last decade consumption has largely ceased, with most animals now sold into the pet trade (McCormack & Stuart, 2016).* Collection efforts include both targeted searches for turtles involving trained dogs, or occasionally pitfall traps, as well as capitalising on casual turtle encounters when collecting other forest products. Turtles, of any species, are collected whenever or wherever they are encountered in the region, regardless of legal protection status or location inside protected areas. Collected turtles are traded, mostly illegally, through a network of middlemen before being exported or consumed locally. Increasing economic value has ensured that hunting pressure is sustained despite the increasing rarity of the species (local prices have increased from an estimated USD 9 in 2006 to USD 15 more recently and in the Chinese markets from USD 20 in 2005 to USD 150). Available information indicates that most *Cuora* traded in Viet Nam were exported to East Asian markets, mainly in Hong Kong SAR and southern mainland China.

There are no records of *Cuora bourreti* in the CITES Trade Database. Records prior to 2016 would have been recorded as part of the trade in *C. galbinifrons,* for which records of legal trade of 3,372 live specimens and 35 specimens were recorded during 1999–2013; only a minority of specimens were likely to have been C. bourreti*, making it difficult to determine the level of trade for* C. bourreti.

The population trend data strongly suggest that *Cuora bourreti* has been subject to unsustainable collection for the past 15–20 years resulting in depletion if not collapse of each population that has been surveyed. Turtle farms are considered to be the primary purchasers of wild-collected turtles and driving the collection of the last remaining wild animals through increased trade prices. *Commercial turtle farms in China are not interested in breeding* Cuora bourreti *in captivity due to high mortality rates, low egg rate production, difficult maintenance and lower value compared to other turtle species in the pet trade (T. Blanck,* in litt., *2019), therefore the provenance of "captive-bred" specimens available in trade could be considered questionable. In more recent years, Viet Nam and China have made increasing efforts to produce specimens with eggs taken from wild-collected females (presumably this meant the female was killed) that are hatched in captivity. These captive-born stocks are considered hardier and yield more money in Viet Nam and China (R. Struijk,* in litt., *2019).*

Cuora galbinifrons (and likely C. bourreti) were present in nearly every reported market survey of turtle trade in China and Hong Kong SAR since recording began in 1993, with *C. bourreti* usually recorded if precise taxonomy was known. All these animals appeared wild-caught and most were offered in food markets. *Cuora bourreti* was not evident in visible retail trade in China or Hong Kong SAR in the early 2010s. It is occasionally offered for sale on internet fora and bulletin boards often claimed to be from captivity: the veracity of such claims is often difficult to confirm. *Searches for* C. bourreti *online found advertisements for both captive-bred and wild-sourced animals although they were sometimes described and traded as* Cuora *spp., (Y. Wang,* in litt., *2019), making it difficult to determine the level of trade.*

Reported seizures of Cuora bourreti *appear to be relatively limited:* in 2018, 18 individuals were seized in Kon Tum and Quang Nam provinces in Viet Nam. *It appears that most of the trade is illegal and there are limited seizures that report this species, which could be a result of weak enforcement in this region and which is unable to regulate trade in this species (T. Blanck,* in litt., *2019).*

Additional Information

Threats

Long-lived late-maturing species with limited annual reproductive output and high juvenile mortality, as exemplified by *Cuora bourreti,* have proven to be highly susceptible to over-exploitation, particularly of adult animals. The primary threat to *C. bourreti* is collection for trade. Habitat loss and degradation are considered a significant but more localised threat to the species.

Conservation, management and legislation

Cuora bourreti was included as a subspecies of *C. galbinifrons,* in CITES Appendix II at CoP11 (2000) as part of the genus level listing. Cuora galbinifrons *was proposed for inclusion in Appendix I at CoP16 (2013) but due to another proposal proposing* C. galbinifrons *and a number of other species already included in Appendix II for a zero-export quota for wild specimens (CoP16 Prop. 32), Viet Nam withdrew their proposal*. A zero-export quota for wild specimens traded for commercial purposes was adopted for *C. galbinifrons* (including *bourreti* and *picturata* as subspecies) at CoP16 (2013). The CITES Standard nomenclature adopted at CoP17 in 2016 recognises *C. bourreti* and *C. picturata* as full separate species.

Cuora bourreti is legally protected from exploitation in both range States, but enforcement may be insufficient.

The genus *Cuora,* including *C. bourreti* is included in Annex B of the EU Wildlife Trade Regulations.

Populations of *Cuora bourreti* are not known to be managed in any part of its range, although a conservation initiative to focus on *C. bourreti* through improved protected area management, community engagement and enforcement capacity building has been initiated at Song Thanh Nature Reserve, Quang Nam Province in central Viet Nam.

Artificial propagation/captive breeding
The Turtle Conservation Centre at Cuc Phuong National Park has been breeding *Cuora bourreti* with limited success; survival of eggs is low and long-term survival of hatchlings is lower. *Cuora bourreti* is maintained in modest numbers in captivity by hobbyists in Asia, Europe, the USA and elsewhere, but continues to be regarded as a difficult, sensitive species that is challenging to establish and reproduce consistently in captivity. A European studbook has existed for the taxon since the late 1990s, with 22 registered animals maintained in 2009. *This species is considered to be delicate in captivity and is easily stressed, however successful breeding is increasing. It is estimated that fewer than 1,000 specimens are believed to be in captivity (TCC, 2018). One expert noted that the climate in southern China is not warm enough so they do not do well in farms—during farm surveys there only a handful of* C. bourreti *(and* C. picturata*) were observed compared to a few dozen* C. galbinifrons*, tens of thousands of* C. flavomarginata *and thousands of* C. amboinensis*,* C. trifasciata *etc. (T. Blanck,* in litt., *2019). The same expert also believed that captive-bred individuals were destined for the pet trade as they were hardier, and wild-caught individuals were consumed as food or for Traditional Asian Medicine. Juvenile animals are often kept at the village level in attempts to raise them to sell on into the trade (often unsuccessful, with animals dying) (McCormack & Stuart, 2016).*

Extremely little is known of the biology of C. bourreti *in the wild; most observations on diet, growth, and reproduction derive from animals maintained in captivity. Slow growth to maturity (10–15 years) is combined with low fecundity (in captivity a single clutch of 1–3 large eggs) (McCormack & Stuart, 2016).*

Implementation challenges (including similar species)
Cuora can be separated from all other turtles by the combination of possessing a single hinge on the plastron allowing them to close their shell effectively, their distinctly domed shell, and generally bright facial coloration. The species *C. bourreti* can most easily be separated from *C. galbinifrons* by its plastron colouration, which is solid black in *galbinifrons* and bony yellow with a large black blotch on each scute in *bourreti* and *picturata. This group of species is easy to identify from the plastral pattern, and easy for specialists to identify from the carapace side (R. Struijk,* in litt., *2019).*

Potential risk(s) of a transfer from Appendix II to I
Information from hobbyists in China indicates that the inclusion of Cuora picturata *in Appendix I would have little impact on trade and animals would continue to be wild-sourced and the value may go up as a result of increased demand (D. Gaillard,* in litt., *2019). Although the current CITES Appendix II listing with a zero-export quota has meant no legal exports have occurred from Viet Nam, enforcement is weak with illegal trade between Lao PDR, Viet Nam and China occurring. If current enforcement efforts are not sufficient, inclusion of this species in Appendix I may not have any significant effect on regulating this trade (T. Blanck,* in litt., *2019).*

One expert considered that ex-situ captive populations of this species were crucial for its long-term survival, and this requires international exchange to maintain genetic diversity of captive populations so they are viable for reintroduction; inclusion of the species in Appendix I, may hinder these efforts (T. Blanck, in litt., *2019).*

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Transfer of Vietnamese Box Turtle *Cuora picturata* from Appendix II to Appendix I

Proponent: Viet Nam

Summary: The Vietnamese Box Turtle *Cuora picturata* is a medium-sized terrestrial turtle, reaching up to 19 cm carapace length. *Cuora picturata* has historically been consumed for food, however in the last decade large-scale consumption has largely ceased with most collected animals now sold into the pet trade or for traditional medicine. Based on observations in captivity, the species lays a single clutch of one to three eggs annually.

Cuora picturata has been listed in CITES Appendix II since 2000 under a genus level listing. *Cuora picturata* was previously considered a subspecies of *C. galbinifrons,* however *C. picturata* was recognised as a species in the standard nomenclature reference adopted at CoP17. Before this split was recognised under CITES, *C. galbinifron*s was included in the Periodic Review after CoP16 and the Animals Committee recommended the transfer of *C. galbinifrons* (at that time including *C. picturata* and *C. bourreti*) to Appendix I.

The species is endemic to Viet Nam and thought to be limited to the eastern slopes of the Langbian Plateau. Only one of the three localities where it is confirmed to occur is currently protected. Large areas of the plateau are being rapidly converted to coffee plantations and other agricultural lands and the remaining area of suitable habitat is estimated to be around 3,000 km² in extent. Unsustainable collection is considered to be the main threat given that *Cuora picturata* is a long-lived, late maturing species with limited annual reproductive output and high juvenile mortality.

The species was assessed as Critically Endangered in 2015 on the basis that documented market volumes in China and Hong Kong Special Administrative Region (SAR) indicated a collapse of populations of over 90% in the past 60 years (three generations), and that collection pressure for the last remaining individuals was likely to continue if not increase in the next 20 years. This was inferred from market observations reported as predominantly *C. galbinifrons,* which at the time was also considered to include what are now accepted as *C. picturata* and *C. bourreti.* The global wild population of *C. picturata* is estimated at below 25,000 individuals, and likely no more than 3,000–10,000. Many hunters state that while *C. picturata* was common 7–15 years ago, it is now increasingly difficult to find.

In 2013 a zero export quota for wild specimens for commercial purposes was adopted for the listing of *C. galbinifrons* (including *C. picturata*) and there has been no legal trade in *C. picturata* reported in the CITES Trade Database since then. Live specimens are observed for sale online, often stating that they are from captive-bred populations. However, *C. picturata* continues to be regarded as a difficult, sensitive species to breed successfully in captivity, reproducing slowly with small clutch sizes. Low numbers have been observed in farms in China, and juvenile animals are said to be raised in villages in Viet Nam for sale into trade, although suffer high mortality rates.

Cuora picturata is protected from commercial exploitation in Viet Nam. Illegal trade is considered to continue to the main destination markets in China and Hong Kong SAR. However, no observations of the species were reported in Hong Kong SAR markets between 2014 and 2018.

Analysis: No legal trade has been reported since a zero export quota was put in place in 2013. Illegal international trade is thought to occur, but it is not clear on what scale. The population size is estimated at fewer than 25,000 (more likely between 3,000 and 10,000) which could be considered small for a low productivity species such as this. Due to habitat loss only a small part of its range is now believed to be suitable (3,000 km^2). The species was categorised as Critically Endangered in 2015 based on a decline of 90% in the past three generations, and this decline was predicted to continue. It would therefore appear to meet the criteria for inclusion in Appendix I.

Other Considerations: At CoP16 a zero export quota in wild specimens for commercial purposes was adopted with the listing for *Cuora galbinifrons* and therefore all trade in wild specimens of

C. picturata is already illegal. Benefits of an Appendix I listing are unlikely to be realised unless enforcement efforts are increased.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Cuora picturata was traditionally considered a subspecies of *Cuora galbinifrons;* however, more recent research has treated *C. picturata* as a full species, including the nomenclature standard reference for the *Cuora galbinifrons* group adopted at CoP17, which recognises *C. picturata* (and *C. bourreti*) as full valid species for CITES purposes.

Range

Viet Nam

IUCN Global Category

Critically Endangered A2bd+4bd (2015) ver. 3.1.

Biological criteria for inclusion in Appendix I

A) Small wild population

The global surviving wild population is likely to be well below 25,000 individuals, and likely much less than this. Cuora picturata *was included in a recent publication by the Turtle Conservation Coalition, highlighting species that are considered to be at the highest risk of extinction. It was estimated that the remaining natural habitat for this species is likely to be 3,000 km^2 and that the population is between 3,000–10,000 individuals in the wild (TCC, 2018).*

Population density of *Cuora picturata* is considered to be low:

- Ly *et al.* (2011) carried out field surveys and required 15 days of a four-person team with three trained hunting dogs (60 man-days/480 man-hours; 45 dog-days/360 dog-hours) to encounter eight turtles, translating to 60 man-hours and/or 45 dog-hours per turtle.
- During a field survey in 2012 in Deo Ca–Hon Nua Special Use Forest (a traditional collecting area in Phu Yen Province, Viet Nam), a team of five dogs (four local hunting dogs and one trained survey dog) found only a single *Cuora picturata* and two *Cuora mouhotii* during a week of searching in which dogs were actively used over 22 km of transects, meaning an estimated density of less than one *C. picturata* per square km, or one turtle per 280 dog hours.

B) Restricted area of distribution

The estimated area of occurrence is 25,000 km^2 and the estimated area of occupancy is 3,000 km^2 (McCormack et al., *2016).* Large areas of suitable habitat at suitable elevation have been converted for agricultural purposes.

Cuora picturata is apparently limited to the eastern slopes of the Langbian Plateau, being known only from Khanh Hoa and Phu Yen provinces of southern Viet Nam. *A survey found one male specimen of this species in Binh Dinh Province in 2013 but as this specimen was observed in the house of a trader it is still unverified if this species naturally occurs in this area (Duong* et al., *2014).*

Cuora picturata was originally (at the time of its description, late 1990s) speculated possibly to occur in eastern Cambodia. However, surveys in Cambodia have failed to find any indication of the species there.

C) Decline in number of wild individuals

Documented trade volumes indicate a collapse of populations over the past three decades of over 90% which is extrapolated back to cover the past 60 years (three generations), and collection pressure for the last remaining individuals is likely to continue if not increase in the next 20 years (McCormack et al., *2016).* Available interview and trade observation data strongly suggest that populations have been severely depleted in recent decades.

Sustained survey work has been undertaken in Viet Nam between 2009–2012 focused on determining the range and priority habitat for the *Cuora galbinifrons* group, focusing on *C. picturata* and *C. bourreti.* Anecdotal information from interviews concluded that historic quantities of the species available for collection in the forest have been greatly reduced, with many hunters stating that while these box turtles were common 7–15 years ago, they are now increasingly difficult to find.

There have been recent reports of a new hybrid species described as Cuora serrata *which was believed to be a hybrid of* C. picturata *and* C. mouhotii obsti*, found in the possession of a local trader in Phu Yen Province who sourced the specimen from a locality within* C. picturata *range (Struijk & Blanck, 2016). One possible outcome for natural hybrids of these species is that population densities for both parental species has become so low that is has reduced the availability of compatible mates (Struijk & Blanck, 2016).*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Cuora picturata has historically been consumed locally for food as part of a subsistence diet. However, in the last decade consumption for food has largely ceased, with most animals now sold into the pet trade (McCormack et al., *2016).* Collection efforts are targeted and occur mainly as searches for turtles involving trained dogs. Turtles, of any species are collected whenever and wherever encountered in the region, regardless of legal protection status or location inside protected areas. Collected turtles are traded, mostly illegally, through a network of local middlemen before being exported or consumed locally. In May 2012, local hunters claimed they still caught 50 individuals of the species annually (during a six-month hunting season).

There are no records of *Cuora picturata* in the CITES Trade Database. Previous records would have been recorded as part of the trade in *C. galbinifrons,* for which records of legal trade of 3,372 live specimens and 35 specimens are recorded during 1999–2013; only a small minority of these specimens were likely to have been *C. picturata, making it difficult to determine the level of trade.*

The population trend data strongly suggests that *Cuora picturata* has been subject to unsustainable collection for the past 15–20 years with over 15,000 specimens of *C. galbinifrons* (including *C. picturata*) recorded in Hong Kong SAR markets alone during the period 2000–2003 and this has resulted in depletion if not collapse of each population that has been surveyed (McCormack *et al.,* 2016). Turtle farms are considered to be the primary purchasers of wild collected turtles and are driving the collection of the last remaining wild animals through increased trade prices. *One expert noted that commercial turtle farms in China are not interested in breeding* C. picturata *in captivity due to high mortality rates, low egg rate production, difficult maintenance and their lower value compared to other turtle species in the pet trade (T. Blanck,* in litt., *2019) therefore, the provenance of "captive-bred" specimens available in trade could be considered questionable. However, it was noted by a different expert that in more recent years, Viet Nam and China have made increasing efforts to produce captive-born species with eggs taken from wild-collected females (which are presumably killed) that hatch in captivity, these captive-born stocks are hardier and yield more money in Viet Nam and China (R. Struijk,* in litt., *2019).*

Cuora picturata was not evident in visible retail trade in China or Hong Kong SAR in the early 2010s. It is occasionally offered for sale on internet fora and bulletin boards, often claimed to be from captivity: the veracity of such claims is difficult to confirm. *Searches for* C. picturata *online found advertisements for both captive-bred and wild-sourced animals. However, they were sometimes described and traded as* Cuora *spp., (Y. Wang,* in litt., *2019), making it difficult to determine the level of trade. Surveys conducted of markets in Hong Kong SAR between 2014–2018 found no* C. picturata *(T. Blanck.,* in litt., *2019).*

An evaluation of reported seizures of tortoises and freshwater turtles between 2000–2015 did not record any *Cuora picturata.* Illegal trade in *C. picturata* may have been recorded as part of *C. galbinifrons,* which was reported in nearly every market survey that looked at turtle trade in China and Hong Kong SAR since recording began in 1993. All these animals appeared wild caught and most were offered in food markets. *It appears that most of the trade is illegal and there are limited seizures that report this species, which could be a result of weak enforcement in this region and which is unable to regulate trade in this species (T. Blanck,* in litt., *2019).*

Additional Information

Threats

The primary threat to *Cuora picturata* is collection for trade. Long-lived, late maturing species with limited annual reproductive output and high juvenile mortality, such as exemplified by *C. picturata,* have proven to be highly susceptible to over-exploitation, particularly of adults. *Extremely little is known of the biology of C. picturata in the wild; most observations on diet, growth, and reproduction derive from animals maintained in captivity—the species has low fecundity (in captivity a single clutch of 1–3 large eggs) (McCormack* et al., *2016).*

Habitat loss and degradation are considered a significant but more localised threat to the species, large areas of forest on the Langbian Plateau are being rapidly converted to coffee plantations and other agricultural lands and only one of the three localities where Cuora picturata was found (Deo Ca Protected Forest) was protected at the time (McCormack et al., *2016).*

Conservation, management and legislation
Cuora picturata was included as a subspecies of *C. galbinifrons,* in CITES Appendix II at CoP11 (2000) under a genus level listing. Cuora galbinifrons *was proposed for inclusion in Appendix I at CoP16 (2013) but due to another proposal proposing* C. galbinifrons *and a number of other species already included in Appendix II for a zero quota for wild specimens (CoP16 Prop. 32), Viet Nam withdrew their proposal*. A zero-export quota for wild specimens traded for commercial purposes was therefore adopted for *Cuora galbinifrons* (including *bourreti* and *picturata* as subspecies) at CoP16. The CITES standard nomenclature adopted at CoP17 in 2016 recognises *C. bourreti* and *C. picturata* as full separate species.

Cuora picturata is protected from commercial exploitation in Viet Nam as a Priority Protected Rare, Precious and Endangered Species under the revised Decree 160/2013/ND-CP of the Government.

The genus *Cuora,* including *C. picturata,* is included in Annex B of the EU Wildlife Trade Regulations.

Only one of the three localities where *Cuora picturata* is confirmed to occur is currently protected (Deo Ca Protected Forest), *however indiscriminate extraction of wildlife still occurs in the protected area as does intensive logging (T. Blanck,* in litt., *2019).* No population monitoring programmes are in place for *C. picturata* anywhere in its limited range.

Artificial propagation/captive breeding
Cuora picturata is maintained in small numbers in captivity by hobbyists and institutions in Asia, Europe, USA and elsewhere, and has been bred in captivity, but continues to be regarded as a difficult, sensitive species that is challenging (but not impossible) to establish and reproduce consistently in captivity. A European studbook exists for *C. picturata* since the late 1990s, with 57 registered animals maintained at three institutions and private keepers in 2009; of these 31 were bred in captivity. *It is estimated that fewer than 500 specimens exist in captivity, with only about 20% serving for assurance purposes (TCC, 2018).*

Cuora galbinifrons was recorded among the stock kept at commercial turtle farms in China in the early 2000s, and likely included some specimens of *picturata.* The *galbinifrons* group is understood not to breed successfully in commercial captive conditions and is no longer included in inventories of turtle farms in recent years. *This species is hard to keep and breed in captivity, producing small clutches, which makes them unattractive to commercial Chinese turtle farms (T. Blanck,* in litt., *2019). The climate in southern China is not warm enough so they do not do well in farms—during farm surveys there only a handful of* C. picturata *(and* C. bourreti*) were observed compared to a few dozen* C. galbinifrons*, tens of thousands of* C. flavomarginata *and thousands of* C. amboinensis*,* C. trifasciata *etc. (T. Blanck,* in litt., *2019). The same expert also believed that captive-bred individuals were destined for the pet trade as they were hardier, and wild-caught individuals were consumed as food or for Traditional Asian Medicine. Juvenile animals are often kept at the village level in attempts to raise them to sell on into the trade (often unsuccessful, with animals dying) (McCormack* et al., *2016).*

Implementation challenges (including similar species)
Cuora can be separated from all other turtles by the combination of possessing a single hinge on the plastron allowing them to close their shell effectively, their distinctly domed shell, and generally bright facial coloration. The species *picturata* can most easily be separated from *galbinifrons* by its plastron coloration, which is solid black in *galbinifrons* and bony yellow with a large black blotch on each scute in *bourreti* and *picturata. This group of species is easy to identify from the plastral pattern, and easy for specialists to identify from the carapace side (R. Struijk,* in litt., *2019).*

Potential risk(s) of a transfer from Appendix II to I
Enforcement is weak with illegal trade between Lao People's Democratic Republic (PDR), Viet Nam and China and if current enforcement efforts are not sufficient, inclusion of this species in Appendix I may not have any significant effect on regulating this trade (T. Blanck, in litt., *2019).*

One expert considered that ex situ captive populations of this species were crucial for its long-term survival, and this requires international exchange to maintain genetic diversity of captive populations so they are viable for reintroduction, inclusion of the species in Appendix I, may hinder these efforts (T. Blanck, in litt., *2019).*

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Transfer of Annam Leaf Turtle *Mauremys annamensis* from Appendix II to Appendix I

Proponent: Viet Nam

Summary: The Annam Leaf Turtle *Mauremys annamensis* is a medium-sized freshwater turtle. Historically *Mauremys annamensis* was consumed as food as part of a subsistence diet, however in the last decade most animals were sold into the higher value international pet and traditional medicine trade.

Mauremys annamensis occurs in central Viet Nam in five provinces (possibly seven) where it is found in the marshes and slow-flowing streams of the lowlands. Within this area, the species is now limited to scattered occurrence within isolated wetlands. Conversion of wetlands to agriculture, such as rice paddies and irrigation canals has led to extensive collection of *M. annamensis* through incidental encounters.

The current or historic population size of *M. annamensis* is unknown. In the late 1930s the species was considered to be abundant, and this remained the case in the 1980s and early 1990s according to anecdotal accounts. *Mauremys annamensis* was assessed as Critically Endangered in 2000 based on a known or inferred population reduction of at least 80% over the past three generations due to actual or potential levels of trade, and a similar projected future decline over the same time period. The main threats were/are over-collection and habitat loss. In recent years, field surveys have found very small numbers of animals in the wild, despite targeted surveys, and the species is rarely observed in market surveys or seizures, indicating it is now extremely rare. Some experts consider that this species is now functionally extinct in the wild.

Mauremys annamensis is legally protected from exploitation in Viet Nam, although enforcement is considered to be weak. The species was included in CITES Appendix II in 2003, and since 2013 there has been a zero-export quota for wild specimens for commercial purposes. Captive breeding is known to occur in Viet Nam and non-range States (including China, the USA and Europe) and it is now thought most specimens in trade are captive-bred, although wild individuals may be being used as parental stock. Animals take about seven years to mature, and recruitment is slow.
Prior to 2013, live exports were approximately five individuals per year, whereas from 2013 this increased to ca. 300 per year totaling ca. 2,000 (predominantly reported as captive-bred/born exported from the USA for commercial purposes to Hong Kong Special Administrative Region (SAR) (1,100). Extreme price fluctuations have been observed in trade in China and Viet Nam, and the current price of around USD 30 per juvenile has been inferred to suggest that the demand for this species is decreasing or that there was now an abundance of captive-bred hatchlings available which reduced the market value.

Analysis: The ongoing international trade in *Mauremys annamensis* consists mainly of individuals reported as captive-bred and captive-born. Its range is limited to scattered isolated occurrences in five provinces (possibly seven) in central Viet Nam. In 2000 the species was assessed as Critically Endangered. Reports from local people and market observations indicate that the species was considerably more abundant in the 1980s and 1990s, suggesting a marked population decline. It is very rare and could be functionally extinct in the wild. All indications suggest that this species' life history traits make it intrinsically vulnerable to over-exploitation. Following a Periodic Review after CoP16 the Animals Committee recommended the inclusion of *M. annamensis* in Appendix I. It would appear *M. annamensis* meets the criteria for inclusion in Appendix I.

Other Considerations: At CoP16 a zero-export quota in wild-sourced specimens for commercial purposes was adopted with the listing for this species and therefore all trade in wild specimens is already illegal. Any additional benefits of an Appendix I listing are not clear.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Range
Viet Nam

IUCN Global Category
Critically Endangered *A1d+2d* (*2000) ver 2.3.*

Biological criteria for inclusion in Appendix I

A) Small wild population

The actual or historic population size of *Mauremys annamensis* is unknown. In recent years, field surveys have found very small numbers of animals in the wild, indicating that the species is now extremely rare; the first animal trapped by scientists in natural habitat since 1941 was in 2006, over 65 years later.

Recently, seven animals found in the wild were identified as *"Ocadia glyphistoma",* a name now believed to be based on specimens of a hybrid origin with *Mauremys annamensis* and *Mauremys sinensis* as parents. The occurrence of these animals in the wild led to the hypothesis that population levels of both species have fallen so low that no separate breeding populations exist anymore, leading to hybridisation.

B) Restricted area of distribution

Mauremys annamensis *is endemic to five provinces (possibly seven) of central Viet Nam where it appears limited to the marshes and slow-flowing streams of lowlands of the Boung River drainage and geographic barriers limit the species' range (TCC, 2018).* Within this geographic distribution area, the species is now limited to scattered occurrence in isolated wetlands.

Habitat for this species has been greatly reduced in recent decades: lowland wetlands are considered prime agricultural land, the rapid population growth in Viet Nam of recent decades has been accompanied by demand for such land. Nearly all lowland wetland areas throughout the historic range of the species have been converted to agriculture, largely for rice cultivation. At one site for the species in Binh Son district, Quang Nam province, some natural boggy wetlands remain, which may represent some of the last historic habitat for the species.

C) **Decline in number of wild individuals**

In the late 1930s, the species was considered abundant in its localised area of occurrence. Interviews conducted by the *Mauremys annamensis* Project staff in recent years found local interviewees often reported that when the species was common in the 1980s and early 1990s, animals would often wander into local houses, or were even considered pests as large animals would trample rice crops.

Mauremys annamensis was assessed in 2000 as Critically Endangered on the IUCN Red List based on a known or inferred population reduction of at least 80% over the past three generations due to actual or potential levels of trade, and a similar projected future decline over the same time period. A reassessment (currently in progress) indicates that the species will be retained as Critically Endangered. The actual or historic population size of *M. annamensis* is unknown. *Since its description there have been a limited number of observations of this species in its native range as a result of conflict and political isolation, it has therefore been difficult to determine the rate of decline (McCormack et al., 2014).*

Mauremys annamensis take about seven years to mature and females may produce one or two clutches of 5–8 eggs per year; egg and hatchling mortality rates are high, and recruitment is slow. These life history attributes make the species intrinsically vulnerable to over-exploitation, particularly of adults.

During April and May 2006 comprehensive interview-based surveys were conducted in Quang Nam province, focusing on *Mauremys annamensis.* A total of 397 local people were interviewed, 93 of whom provided information on *M. annamensis.* Information was received from a boy who was in possession of a specimen of *M. annamensis* he claimed to have caught in a lake known locally as Ha Tre Lake. In November 2006, a team from the *Mauremys annamensis* Project (MAP) investigated this area, which resulted in the first capture of a wild single sub-adult since 1941. A full-time monitoring team was then deployed in the area in September 2007 in the Dien Phong Commune in addition to further interviews with locals. A total of 110 days of trapping were carried out at three sites in Duy Xuyen and Dien Ban districts and resulted in no further field records of *M. annamensis.*

Habitat loss, the use of pesticides and habitat degradation combined with collection for Vietnamese farming demand and trafficking to China have resulted in dramatic declines in numbers and experts now consider M. annamensis *functionally extinct in the wild (T. Blanck,* in litt., *2019).*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Historically, *Mauremys annamensis* has been consumed locally for food as part of a subsistence diet, however in the last decade consumption as food has largely ceased with most animals now sold into the pet trade (and Traditional Asian Medicine *(T. Blanck,* in litt., *2019))* due to the high economic incentive.

Mauremys annamensis does have specific local medicinal uses: throughout much of its range, its use in local traditional medicines remains and is possibly being promoted. Blood from the turtles is mixed with strong rice wine and drunk with the belief that it is a cure for heart disease. Soups and other tonics are also made from the species. The alleged heart disease cure gives it a higher local price than other, closely related local species such as *Mauremys sinensis.*

According to the CITES Trade Database, between 2003 and 2017 exporters reported exporting a total of 1,685 live specimens (importers reported 1,558):

- *1,246 were reported to be from captive-bred sources ("C"), 435 were captive-born ("F") and there were no reported exports of wild-sourced specimens ("W"). Exports of "F" could be the offspring of at least one wild parent.*
- *The largest exporter of captive-bred and captive-born specimens was the USA (1,410 live specimens).*
- *The largest importer was Hong Kong SAR (1,211 live specimens).*
- *Most exports were for the purpose of commercial trade (1,548 live specimens).*
- *Only four individuals, exported from the USA to Taiwan Province of China were reported to have been confiscated in 2009.*
- *Prior to 2013, annual average live exports were approximately five, whereas from 2013 this increased to 326 per year. Trade volumes were particularly high between 2015–2017 and comprised mainly live specimens from captive-bred and captive-born sources (Figure 1).*

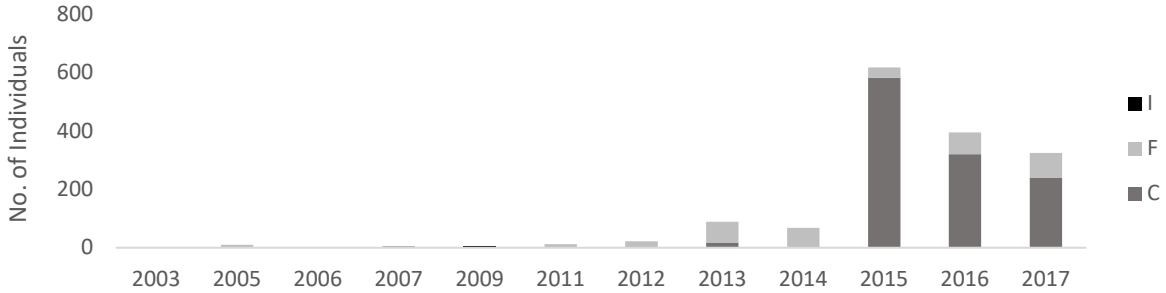

Figure 1: Reported exports of live specimens between 2003–2018 according to the CITES Trade Database

Evidence online suggests that there is still demand for Mauremys annamensis*, and a recent post in 2019 claimed to have stock of over 400 individuals (gui999, 2019). It appears that the market is currently dominated by captive-bred stock (Y. Wang,* in litt., *2019).*

There have been significant price fluctuations. The wholesale price in central Viet Nam has varied around USD50 during 2009–2011, but in August 2011 the wholesale price for *Mauremys annamensis* spiked to USD1,200–1,440 per kg, before collapsing below its previous level within weeks, and hovered around USD25 per kg in 2012. *Similarly, its price increased dramatically in China from over USD12 per individual in 2008 to over USD1,700 per individual in 2014. The market appeared to collapse soon after due to a large quantity of captive-bred specimens entering the market (Y. Wang,* in litt., *2019). Since 2017, the interest of Chinese farmers for* M. annamensis *was said to be ceasing, while Western breeders were selling hatchlings for over USD100, the current price is around USD30 per juvenile suggesting that the demand for this species is decreasing (T. Blanck,* in litt., *2019).*

While *Mauremys annamensis* was common in local trade in 1996, the species is rarely seen in wildlife trade shipments in recent years, with only modest numbers (fewer than 10) specimens seen annually in local trade at sites since 2007, and the last wild collected specimens recorded in trade in 2013.

Additional Information

Threats

Mauremys annamensis is affected by direct collection and habitat degradation. Direct collection of adults and juveniles, historically for occasional subsistence consumption, but mainly for the international pet trade in recent years, is highly likely significantly to impact populations of a species whose life history has evolved to accept moderate losses of juveniles but whose reproductive adults are of great population value. *With very limited numbers of wild-sourced individuals seized and the rarity of* Mauremys annamensis *in the wild, there may be no trade in wild-sourced specimens occurring (T. Blanck,* in litt., *2019).*

Conversion of natural lowland to wetlands to agricultural land use, such as rice paddies and irrigation canals leads to extensive collection of animals that are encountered incidentally while tending crops etc. In addition, pollution from herbicides used during conflict, agrochemicals, industrial pollution, and sewage effluent, likely all represent additional impacts on the species and its habitat.

Conservation, management and legislation

Mauremys annamensis is legally protected in Viet Nam from any form of exploitation by inclusion in Schedule IIB of Decree 32/2006/ND-CP, though enforcement at local jurisdictions may be insufficient. Schedule II includes species whose utilisation is restricted to scientific research, establishing breeding populations, and international exchange; any such activities require a collection permit from the Ministry of Agriculture and Rural Development.

Mauremys annamensis was listed in Appendix II in 2003, and a zero-export quota for wild-sourced specimens for commercial purposes was adopted in 2013, and it is included in Annex B of the EU Wildlife Trade Regulations. *It was proposed for inclusion in Appendix I at CoP16 but due to another proposal proposing* Mauremys annamensis *and a number of other species already included in Appendix II for a zero-export quota for wild specimens (CoP16 Prop 32.) Viet Nam withdrew their proposal*. *Mauremys annamensis* was included in the Periodic Review between CoP16 and CoP17 and the Animals Committee agreed with the recommendation in the review to transfer *M. annamensis* to Appendix I.

No population management measures have taken place or are in preparation in the species' range beyond the establishment of a legal framework for sustainable development and conservation of freshwater turtles and tortoises. An international programme has taken shape since 2006 to reintroduce and strengthen a viable population of *M. annamensis* in its native range. *There have been attempts to release captive-bred specimens back into the wild, but this has been hindered by the lack of remaining suitable and protectable habitat (T. Blanck,* in litt., *2019).* Surveys in central Viet Nam have not confirmed the species' presence in any existing protected areas; it is likely that due to its habitat niche such areas do not exist. Work is ongoing to establish a small Species Habitat Conservation Area (SHCA) for *M. annamensis (roughly 100 ha (McCormack* et al., *2014)).*

Artificial Propagation/captive breeding

The species is easily bred and can produce high numbers in Europe, USA and Asia. Chinese farmers can produce thousands of hatchlings and this has influenced market prices which have dropped down from over USD 500 for a hatchling in 2015 to only USD 25 in 2018 (T. Blanck, in litt., *2019).*

Some turtle breeding farms have been established in Bac Ninh, Binh Dinh and Phu Yen provinces of Viet Nam, with the largest observed in 2009 holding approximately 40 animals of *Mauremys annamensis*; the current number held is unknown. The Asian Turtle Program, at its facilities at Cuc Phuong National Park, has a long-established successful breeding programme for the species for conservation purposes. *Mauremys annamensis* is one of about 30 species of freshwater turtles that is commercially bred at a turtle farm at Tun Chan, Hainan Island.

Breeding of *Mauremys annamensis* in Europe and the USA is primarily a non-commercial, conservation-focused effort, and a commercial interest for only a very limited number of breeders. Successful captive breeding at European institutions occurred from 2006 and quickly became so successful, and placement options so limited, that some institutions have resorted to only incubating some of the eggs produced.

Implementation challenges (including similar species)

Mauremys annamensis can be easily recognised by its clear pattern of creamy yellow stripes on its otherwise deep olive-green face, head and neck.

Potential risk(s) of a transfer from Appendix II to I

Some experts now consider Mauremys annamensis *to be functionally extinct in the wild (T. Blanck,* in litt., *2019) and the long-term survival of this species relies on ex-situ conservation programmes that are captive breeding this species for reintroduction purposes (D. Gaillard,* in litt., *2019 & T. Blanck,* in litt., *2019). Therefore, inclusion*

of Mauremys annamensis *in Appendix I could prevent or impede possible reintroductions from ex-situ captive bred populations (D. Gaillard,* in litt., *2019 and T. Blanck,* in litt., *2019).*

Enforcement appears to be weak in this area with illegal trade occurring between Viet Nam and China, allowing turtles to pass across borders that are not controlled. Therefore, if current enforcement efforts are not sufficient, inclusion of this species in Appendix I may not have any significant effect on regulating this trade (T. Blanck, in litt., *2019).*

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Transfer of Star Tortoise *Geochelone elegans* from Appendix II to Appendix I

Proponents: Bangladesh, India, Senegal and Sri Lanka

Summary: The Star Tortoise *Geochelone elegans* is very popular in the pet trade. It is found in north-western and south-eastern India, eastern Pakistan, and northern and eastern Sri Lanka. Its current area of occupancy is greater than 2,000 km^2 and extent of occurrence greater than 20,000 km^2, with both reported to be declining.

Geochelone elegans is found in a variety of dry vegetation types including scrubland, grassland and desert edge. It is a relatively adaptable species and tolerant of change, being found in agricultural landscapes including fields, hedgerows and plantations.

Geochelone elegans faces two main threats; habitat loss, primarily in the form of conversion of preferred habitat to agriculture, and illegal harvesting for the pet trade, particularly the collection of juvenile specimens. Other threats include accidental mortalities from road kills, agricultural equipment and discarded fishing nets, and deliberate killing to protect crops.

There is a lack of quantitative population data for this species. *Geochelone elegans* is categorised by IUCN as Vulnerable having been assessed in 2015; based on past and inferred future declines, a decline of greater than 30% (over a three-generation period) was predicted to occur by 2025 (from a start point of 1995), if exploitation continued or expanded. However, as *G. elegans* was assessed as Vulnerable and not Endangered, declines of greater than 50% were not indicated. Estimated densities of 4.0–2.5 animals/ha were recorded in 1991.

Geochelone elegans has been listed in CITES Appendix II since 1975; and is fully protected by law from commercial exploitation, trade or possession in each of its three range States. However, it is the single most confiscated species of tortoise or freshwater turtle worldwide. Seizures of large numbers of *G. elegans* are well documented. At least 34,000 live individuals were seized between 2000 and 2015, with a further ca. 14,400 individuals seized between 2016 and 2018. Observations of 55,000 individuals being removed from the wild from one location over one year (2015), in India, suggest illegal harvest and trade levels could be considerably higher than the observed seizures.

The CITES Trade Database shows high numbers of *Geochelone elegans* in trade; almost 63,000 live specimens were reported between 2000 and 2015, over half of which were reported as captive-born or bred individuals (ca. 37,000). Over a third had no source code reported (ca. 24,000) and were exported from Jordan, a non-range State, and the largest global exporter of *G. elegans*. Jordan also reported exporting almost 31,000 captive-bred individuals, as well as re-exporting just over 1,900 wild-sourced with no origin specified. Afghanistan was also a significant exporter of wild-caught individuals (5,000). Uncertainty over the size of captive breeding populations and numbers exported from non-range States suggest a large portion of the legally permitted trade is likely to include illegally harvested and misreported wild specimens from range States. Trade from Jordan of *G. elegans* has been subject to the "Review of trade in animal specimens reported as produced in captivity", with recommendations for Jordan including a zero-export quota.

Analysis: *Geochelone elegans* does not have a restricted range, nor does it appear to have a small population. The *G. elegans* population has been reported to be declining, there is strong evidence of large-scale illegal international trade, which along with other factors is believed to be driving this decline. Scant quantitative population trend data are available, but it has been estimated that if threats continue, declines of greater than 30% (but less than 50%) are likely to occur during the three generation period from 1995–2025. This is less than the guideline figure given in *Res. Conf. 9.24 (Rev. CoP17)* for a marked recent rate of decline. However, illegal trade appears to present a constant pressure on the population and given concerns raised through the "Review of trade in animal specimens reported as produced in captivity", it may be precautionary to list the species in Appendix I. *Res. Conf. 12.10* (*Rev. CoP15*) outlines that inclusion in Appendix I would mean commercial captive

breeding operations would need to meet the provisions of *Res. Conf. 10.16* (Rev.) to be registered with the CITES Secretariat, and that registered operations shall ensure an appropriate and secure marking system to identify all breeding stock and specimens in trade. This enhanced oversight could help allay concerns over fraudulent claims of captive breeding and continued wild offtake for breeding stock.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS

Range
India, Pakistan and Sri Lanka

IUCN Global Category
Vulnerable A4cd (2015) ver 3.1

Biological criteria for inclusion in Appendix I

A) Small wild population
The status of specific geographic populations in the wild is poorly known. Estimated densities of 4–12.5 animals/ha were recorded in 1991.

Available information indicates that this species maintains relatively large populations of >10,000, although it is not clear where this value originates (D'Cruze et al., *2016). However, observations of 55,000 individuals poached from the wild, in one trade hub in India, in one year, suggest this estimate could be conservative (D'Cruze* et al., *2015).*

B) Restricted area of distribution
Geochelone elegans is found in the wild in three range countries; widespread in India and Sri Lanka, with a smaller subpopulation in Pakistan.

Available information indicates that this species has a global extent of occurrence (EOO) >20,000 km^2 and an area of occupancy (AOO) of more than 2,000 km^2, although both are reported to be declining (D'Cruze et al., *2016).*

The SS reports that spatial occupancy of the species is shrinking at a fast rate. Of the 16 protected areas surveyed in the State of Gujarat, five of them lost *Geochelone elegans* during 1989–1998. *However, the original paper only reports the presence and absence of the species in the different protected areas. It does not appear to refer to the species being lost.*

C) Decline in number of wild individuals
A marked decline in the population size in the wild which has been observed as ongoing or as having occurred in the past (but with potential to resume):
Based on recent past and predicted future declines, a listing of Vulnerable A4cd is proposed given concerns that population reductions of >30% are likely to occur *by 2025 (over a three-generation period from 1995–2025)*, if this exploitation continues or expands. As *Geochelone elegans* was assessed as Vulnerable *and not Endangered, declines of >50% were not expected.* This echoes separate national Conservation Assessment Management Plan (CAMP) workshop assessments of Indian (assessed 1998) and Sri Lankan (assessed 2000) populations.

Wild populations of this species are present in both protected areas and in agricultural landscapes in range States. Substantial wild populations appear still to exist in India and Sri Lanka, but recent detailed field surveys regarding their exact status are currently lacking. Any populations in Pakistan appear to be extremely localised and small, with recent detailed field surveys also lacking.

A marked decline in the population size in the wild which has been inferred or projected on the basis of: Recruitment
The SS suggests that a marked decline in the population size in the wild can be inferred based on an inferred decrease in recruitment due to the indiscriminate removal of juvenile and adult *Geochelone elegans* from the wild over multiple generations for exploitation in domestic and international trade.

Primary collectors tend to pick up specimens that are a year old or older, but collection of sub-adults and adults also takes place (D'Cruze et al., *2015). Most of the pet trade involves small to medium-sized animals, few exceeding 10 cm CL (assumed to mean carapace length) (D'Cruze* et al.*, 2016).*

Due to the long life expectancy of this species, a consequent lack of recruitment may not manifest as a population reduction for several years, masking the impact of offtake (*this reference is currently in press*).

Intrinsic vulnerability
The SS suggests that a marked decline in the population size in the wild can be inferred based on a high intrinsic vulnerability of the species to over-exploitation due to late onset of reproduction and slow reproductive rate, behavioural traits that allow ease of capture, and specialised niche requirements (i.e. diet and habitat).

Geochelone elegans is long-lived with a low reproductive rate. Data on key life-history parameters in wild populations are incomplete, however mature females (*female sexual maturity reported at the age of 10 years (Frazier, 1987), while Das (1991) and Vyas (1997) report sexual maturity at the age of 6–7 years*) are known to typically produce two clutches (exceptionally up to four clutches) per year, each comprising an average between 2–10 eggs. Generation length has been estimated as approximately 10 years.

Geochelone elegans is most active during the monsoon season when the majority of wild collection for the international pet trade appears to occur (*in D'Cruze* et al., *2015 it states that, "Collection is predominantly seasonal, taking place after the local monsoon seasons (March and April; and September, October and November) when tortoises tend to emerge out of hiding to feed on fresh sprouting vegetation"*).

Extrinsic vulnerability
The SS suggests that a marked decline in the population size in the wild can be inferred based on a high vulnerability to extrinsic factors, specifically a decrease in the area and quality of habitat due to deforestation and land conversion for agriculture, and a high threat of accidental mortalities, via road kills, agricultural equipment and deliberate mortalities to protect crops in converted habitats.

The SS notes that the species is found in a region with one of the highest human populations in the world and projected to increase, presumably inferring greater habitat loss. Estimates of change in the availability of preferred and/or suitable habitat for *Geochelone elegans* have not been made.

Habitat loss, destruction and/or degradation threaten the continued survival of *Geochelone elegans* populations in the wild. Scrub forest habitat is being converted to orchards and cash crop agriculture leading to a reduction of available area of the preferred habitat type. *The habitat is not reported as being severely fragmented (D'Cruze* et al., *2016).* It is a relatively adaptable species, able to tolerate anthropogenically altered habitat, continued habitat loss is likely further to impact wild numbers negatively.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade
Legal trade
A total of 62,085 live *Geochelone elegans* direct exports were reported in the CITES Trade Database exported by 28 different non-range States between the years 2000 and 2015. *The majority that had a source code were recorded as captive-born or bred (61%). Jordan accounted for 89% of specimens in trade: 30,922 as captive-bred and a further 24,276 with no source code. Jordon also re-exported 1,916 tortoises: 1,915 of which were wild but the origin was not stated.* However, given that Jordan is a non-range State that has only received one shipment of 20 live *G. elegans* of unknown origin, there are concerns that its founderstock was not bred in captivity in conformity with *Res. Conf. 10.16 (Rev.)*. A total of 8,599 re-exports were reported.

Consequently, Jordan has been asked to report on how this founder stock was sourced and how much time was required to build up the breeding population. *In a report commissioned by CITES Secretariat, Outhwaite* et al., *2014 reported that at the time there was just one authorised breeding facility in Jordan producing captive-bred* Geochelone elegans*: the facility acquired its original stock of 22 females and nine males (all adults) from the local market in Jordan in 1985, all of which were sold in 1995. The then current breeding stock consisted of 185 females and 62 males, though an additional 72 females and 28 males were added in 2009 and the first clutch from these new animals was expected in 2014. Annual production for the period 2008 to 2012 (around 2,000 offspring per year) was far lower than exports from Jordan of around 4,650 captive-bred specimens per year (2008 to 2011) reported in the CITES Trade Database.* The Animals Committee recommended at its 70th meeting in July 2018 that Jordan establish a zero-export quota for the species from all sources and provide evidence of the legal origin of its breeding stock following a "Review of trade in animal specimens reported as produced in captivity". Ukraine (n=3,962), Hong Kong Special Administrative Region (SAR) (n=1,797) and Slovenia (n=1,783)

were the three largest exporters after Jordan. Ukraine and Slovenia reported no legal import records for *G. elegans* that also bring the legitimacy of their founding into question (*although not all intra-EU trade is reported in the CITES Trade Database so Slovenia may have imported tortoises from another EU Member State*). Hong Kong SAR reported the import of 7,117 live *G. elegans* since 2000: 97% (6,876) of these specimens were imported from Jordan.

According to (re-) exporters, a total of 2,541 wild live specimens were traded between 2000 and 2015, most were re-exports with no origin specified re-exported by Jordon (1,915). Importers reported a total of 5,395 wild live specimens, >99% from Afghanistan and imported by Japan.

Trade was also reported from range States between 2000 and 2015: India (582 exported to Malaysia, source code "I"), Pakistan (5 specimens and 1 live "W" for scientific purposes) and Sri Lanka (9 live "C" for personal/zoo purposes).

From 2016 to 2017 7,929 live Geochelone elegans *were exported, with 7,619 being reported as being bred or born in captivity. Jordan was the largest exporter, responsible for 96% (n=7,332) of these specimens. These data are likely to be incomplete as countries are unlikely to have reported their trade to date.*

There are also concerns that "legal loopholes" are being exploited to sell illegally sourced *Geochelone elegans* in non-range States such as Thailand. Historically, Kazakhstan is reported to have been the main supplier into Thailand even though it is not a range country for this species and has a complete lack of import records for any captive breeding stock. The significant involvement of Lebanon (a non-CITES Party until 2013) also calls the legitimacy of Thailand's founderstock into question. Previous calls for CITES Management Authorities to investigate this trade route may be partly responsible for the observed lack of *Geochelone elegans* imports into Thailand since 2010.

Illegal trade

Numerically, *Geochelone elegans* was said to be the single most confiscated species of tortoise or freshwater turtle worldwide and is thought to represent around 11% of global seizures involving chelonians. Between 2000 and 2015, at least 34,080 live individual *Geochelone elegans* were recorded as seized by wildlife and customs authorities during 118 different enforcement actions internationally. Noteworthy is that during this time nearly two-thirds of all seized live *G. elegans* (21,316 animals) were detected and seized within India.

The illegal collection of at least 55,000 (mostly juvenile) specimens from just one location (comprising 16 villages) from the state of Andhra Pradesh in India, over a period of one year has been reported. This figure is three to six times larger than the 10,000–20,000 individuals previously estimated to be poached throughout the entire range of this species each year.

Data from TRAFFIC's wildlife trade information system reported 33,925 Geochelone elegans *seized between 2007 and 2018, with 14,399 reported between 2016 and 2018.*

A total of 8,825 individual live specimens (reported to have an estimated value of USD 3,530,000, based on one live specimen having an estimated market value of USD 400) seized during 2016 and 2017 has been documented.

There are concerns that the species is being smuggled from India and Sri Lanka into pet markets in Asia, Europe and the USA. However, the majority appear to be destined for use as exotic pets in Asian countries, such as Thailand and China. *Large volume illegal trade pathways for* Geochelone elegans *for the pet trade are reported from South Asia to Southeast Asia (CITES, 2017).*

Country specific information is provided below:

- In Sri Lanka unpublished data provided by the Customs Department and other enforcement officials (including navy, police and flying squad) state that at least 3,130 individual specimens were seized between 2015 and 2017 alone.

- In Thailand, *Geochelone elegans* was the most frequent illegally traded tortoise seized by enforcement authorities between 2008 and 2013 (5,966 individuals during 15 cases). *Geochelone elegans* is the most commonly observed tortoise at Chatuchak Market in Thailand (*the largest market in Thailand*).

- Additional seizures of *Geochelone elegans* occurred in Germany, Indonesia, the Netherlands, the Philippines, Slovakia, Spain, the UK and the USA, in most cases from air travellers arriving from Asia,

as well as some from express mail parcels sent from Asia. The price per specimen in the European pet market may be up to EUR 400–800 (USD 453-906).

- Seizure events in India, reported by the media between 2011 and 2015 revealed that at least 8,533 individual live specimens of *Geochelone elegans* were seized and that it appeared in at least 23% of all such seizure events.

- Geochelone elegans *was the most commonly sighted tortoise and freshwater turtle species observed in Jakarta, Indonesia, during a three-month study of pet shops and markets in 2015. 937 animals were observed with a median of 74 per week (Morgan, 2018).*

- *Sy (2018) report a minimum of 56* Geochelone elegans *observed in 11 posts for sale in Philippine Facebook groups from June–August 2016. This was the third most recorded tortoise and freshwater turtle species in the study.*

- *Sung and Fong 2018 report 486* Geochelone elegans *were advertised for sale in 322 "posts" on an internet forum in Hong Kong SAR between September 2013 and August 2016. The average price was USD 274.*

Additional Information

Threats

Locally in rural areas, *Geochelone elegans* is sometimes eaten for subsistence.

This species is also collected for use as pets (with owners believing they bring good luck and fortune) and for spiritual use *nationally (in some societies they are thought to represent a reincarnation of the Hindu God "Vishnu")*, although trade for these purposes is likely to be negligible compared with the high volume of the international pet trade. Over 100 have been observed in one urban household in India alone and researchers observed a total of 22 animals at three different Shiva temples in the State of Gujarat, India. *In Sri Lanka, some societies consider* Geochelone elegans *to be a bad omen and unlucky (reported in D'Cruze* et al., *2018).*

Habitat loss is occurring throughout the species' range; scrub forest habitat is being converted to orchards and cash crop agriculture, leading to a reduction of available area of the preferred habitat type. Although this is a relatively adaptable species, able to tolerate anthropogenically altered habitat, continued habitat loss is likely to impact wild numbers further (D'Cruze et al., *2016).*

Conservation, management and legislation

Legislation

Geochelone elegans is fully protected by law from commercial exploitation, trade or possession in each of its three range States.

In India, *Geochelone elegans* was placed in Schedule IV of the Wildlife (Protection) Act 1972 in 1980 and for over 38 years it has been illegal to hunt (including take from the wild) this species or trade it domestically without a licence (Sections 9 and 44 of the Wildlife (protection) Act, 1972). *The SS reports that to date no permissions to trade the species domestically appear ever to have been granted.* International trade of the species from India is prohibited.

In Pakistan, the forest, environment and wildlife department of the Government of Sindh, through a notification issued in September 2014, included *Geochelone elegans* along with other chelonian species of Pakistan in Schedule II (Protected Animals) of the Sindh Wildlife Protection Ordinance 1972.

In Sri Lanka, *Geochelone elegans* is protected under the Sri Lanka Fauna and Flora Ordinance (1993).

Internationally, *Geochelone elegans* has been included in CITES Appendix II, as part of the listing of the genus *Geochelone* when the Convention came into force on 1st July 1975. *Geochelone elegans* is listed in Annex A of the EU Council Regulation 338/97, providing the highest protection level to this species within the EU. It is not specifically covered by other Conventions or multilateral environmental agreements.

Management

No official management measures are known in range States for the protection and specific study of *Geochelone elegans*.

There are several challenges associated with the reactive management and repatriation involving large numbers of seized and confiscated live *Geochelone elegans* specimens. These include but are not limited to: (1) non-existent national action plans for effective seizure and disposal of live specimens; (2) incomplete understanding of which national agencies are responsible for effective seizure of live specimens; (3) a lack of financial resources for effective disposal of live specimens; and (4) lack of skilled staff for effective seizure and disposal of live specimens.

No official in-country population monitoring programmes have yet been established for *Geochelone elegans* by any of the three range States.

Artificial propagation/captive breeding
The SS reports that Geochelone elegans does not appear to reproduce regularly or in great numbers even at the best captive facilities. While some small-scale captive breeding may be occurring at some zoos and with some private keepers, few of the offspring are traded internationally, and no large-scale commercial captive production facilities have been documented.

Information provided by the global online database Species 360 (2014) confirmed that a total of 765 *Geochelone elegans* were being held in captivity at 78 different zoos and aquaria in four different geographical regions (Asia, Europe, Oceania and North America).

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Transfer of Pancake Tortoise *Malacochersus tornieri* from Appendix II to Appendix I

Proponents: Kenya and the USA

Summary: The Pancake Tortoise *Malacochersus tornieri* has a unique appearance of a flat and flexible shell allowing it to wedge itself tightly into rock crevices. The species inhabits rocky outcrops in Kenya, United Republic of Tanzania (Tanzania) and northern Zambia, and due to its very specific microhabitat requirements it is discontinuously distributed across its range. While its calculated area of occupancy has been estimated at 72,000 km^2, due to the species' specific requirements the actual area of suitable habitat is thought to be less than 5% of this (and even less to have suitable crevices in terms of dimensions and orientation). The species has low productivity in the wild: maturing at over five years and laying one (sometimes two) eggs per year. The species is in demand for the international pet trade, the largest markets are within Asia, and the USA.

It is thought the Pancake Tortoise spends most of its time inactive in the rocky crevices which provide a thermal buffer; a behaviour which makes it difficult to survey. However, an extrapolation based on population density studies in Kenya suggests a global population of between 4,000 and 32,000 in 2001/2002. Presence in Zambia was confirmed in 2006, and based on a mark-recapture study, the Zambian population was estimated at just over 500 individuals. Opportunities for recolonization of over-exploited areas are limited due to the species' limited movement (limited home range and high site fidelity).

A recent assessment accepted for publication in the March 2019 Red List update categorises the species as Critically Endangered due to observed, estimated and projected population reductions of about 80% over three generations (45 years in total) that will be reached in the next 15 years. The population is believed to be declining: the international pet trade has been identified as the main factor but habitat degradation and loss, particularly from rock destruction and farming, are also significant threats to the species. Low population densities have been observed in otherwise seemingly suitable habitat: surveys in Kenya found the location Voo to have the highest density in 2001/2002 (9/km^2) but no tortoises were observed in 2014 during a repeat survey, and this was attributed to the establishment of a commercial farming operation nearby using wild specimens for breeding stock. Other areas in Kenya surveyed in 2001/2002 that had good Pancake Tortoise populations at that time were also noted to be depleted in 2014. In Tanzania, in the early 1990s the average number of tortoises encountered per hour was approximately 90% lower in areas where they have been exploited.

The Pancake Tortoise was listed in Appendix II in 1975. Kenya does not permit wild exports, and following inclusion in the Review of Significant Trade in the late 1980s, the Standing Committee recommended a trade suspension for Tanzania for wild specimens which remained in place between 1993 and 2018: this has now been lifted on the condition that Tanzania implements a zero export quota for wild specimens. It is not clear what legal provisions there are for the species in Zambia. All three countries have licensed captive breeding facilities. Concerns have been raised about the ability of these farms to produce the fluctuating numbers reported in trade, particularly for Zambia, which started exporting captive-bred tortoises in 2006 (the year the wild population was confirmed in Zambia, although some wild/ranched specimens had been exported before that time). Fluctuations could indicate ongoing capture from the wild (within Zambia or neighbouring Tanzania) for export and/or parental stock. Zambia's exports totalled around 23,000 between 2006 and 2016. Significant exports reported as wild have taken place from non-range States, and illegal trade is highlighted as an issue.

Analysis: The population size in 2001/2002 was estimated at between 4,000–32,000, but on the basis of results from more recent surveys, the slow breeding potential of the species and inferred ongoing wild collection, it seems possible that the current population now meets the definition of being a small wild population that is declining. An IUCN Red List assessment due for publication in March

2019 has categorised the species as Critically Endangered as it is estimated that the population will have declined by 80% over three generations (two past, one future—through to 2033). If the rate of decline is equal across all three generations this will have meant the species has suffered a marked recent decline of over 50% in the last two generations with the decline predicted to continue. The species is affected by trade, although most exported specimens are reported to be captive-bred, results from surveys indicate wild harvest has continued, which may be being used as parental stock in farms or being exported. Therefore, it seems that the Pancake Tortoise meets the criteria for inclusion in Appendix I.

Other Considerations: Wild exports are not permitted from Kenya, and Tanzania has stated that it does not intend to permit export of wild specimens. *Res. Conf. 12.10* (*Rev. CoP15*) outlines that inclusion in Appendix I would mean commercial captive breeding operations would need to meet the provisions of *Res. Conf. 10.16* (Rev.) to be registered with the CITES Secretariat, and that registered operations shall ensure an appropriate and secure marking system to identify all breeding stock and specimens in trade. This enhanced oversight could help allay concerns over fraudulent claims of captive breeding and wild offtake for breeding stock.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS

Taxonomy

There are no other species in the genus *Malacochersus.*

Range

Kenya, United Republic of Tanzania, Zambia.

IUCN Global Category

Vulnerable A1bd ver (1996 needs updating) 2.3.

An updated categorisation of Critically Endangered has been accepted for publication in March 2019 (Mwaya et al.*, 2019).*

Biological criteria for inclusion in Appendix I

A) Small wild population

The abundance of this species is generally low. Due to the cryptic nature of this rock-crevice dwelling species, reliable estimates of its population size are difficult to achieve and therefore poorly understood. However, relative population data (such as abundance and the number of specimens per person-hour of searching time) have been obtained from various populations, and comparisons of harvested with non-harvested sites clearly document reduced density at exploited sites. *Using the global estimated distribution of 3,600 km^2 (see calculation below) and the lowest and highest recorded densities for Kenya, a very approximate estimation of the global population in 2001/2002 was between 4,000 and 32,000. The current population is likely to have decreased as wild harvest appears to have continued and due to the species' life history characteristics (slow breeder, unable to colonise new areas) it is likely to have been unsustainable.*

Kenya: Population surveys in 2001/2002 recorded densities in appropriate habitats ranging from 8.86 tortoises/km^2 to as low as 1.2 tortoises/km^2.

Tanzania: In Tanzania abundance was estimated in 1995 (*surveys undertaken in 1992*) based on the number of specimens per person-hour (ph) of searching time. Encounter rates varied between 8.5 tortoises/ph to 0.23 tortoises/ph.

Zambia: In Zambia, 68 animals were found at eight sites in Nakonde District with 2–25 individuals per site. Based upon a mark-recapture study, a population size estimate of 518 animals was obtained and a density of 11 individuals/km^2.

B) Restricted area of distribution

Pancake Tortoises have extremely rigid microhabitat requirements and only live in rock crevices of suitable dimensions in small rocky hills (kopjes) in dry savannah in parts of Kenya, Tanzania and marginally in northern Zambia. These microhabitats are sparse and few and separated by large areas of unsuitable habitat. The

species' distribution is determined by four limiting factors, namely: geology, climate, vegetation and altitude. Local population size depends on the number of suitable crevices the outcrops provide, the crevices provide buffering against temperature and humidity fluctuations. Suitable rock crevices for the species are easily identifiable by collectors, making populations vulnerable to depletion.

Much of the range appears to be comprised of discontinuous and disjunct populations occurring in isolated patches of suitable habitat. Isolation of suitable habitats, coupled with limited dispersal abilities and low recruitment rates, make recovery of depleted populations unlikely.

While the calculated area of occupancy (AOO) and extent of occurrence (EOO) appear moderate to large, the actual suitable micro-habitat (rocky areas with appropriate crevices) constitute only a very small proportion of the calculated AOO: *estimated less than 5% of the AOO contained the required rocky outcrops, and even within that the area with suitable dimensions is much smaller (P. Malonza,* in litt., *2019).The estimated AOO given in the IUCN Red List assessment is 72,000 km^2 (Mwaya* et al., *2019) using the estimate of 5% being suitable this would equate to 3,600 km^2. While the AOO should relate to the area that the taxon is present in, in reality due to the coarse nature used when mapping and calculating the AOO, the actual AOO can be smaller.*

Zambia: Has only been recorded from a single location in the hilly areas of the northern Nakonde District, an unprotected area bordering Tanzania. *It is possible the species used to occur at a lower altitude but may have been pushed out due to habitat destruction (Chansa & Wagner, 2006).*

C) Decline in number of wild individuals

The status of M. tornieri *populations in the wild is poorly known, but many populations are decreasing (Ngwava, 2015). Low population densities in otherwise seemingly suitable habitat result from removal and offtake by commercial collectors (Mwaya* et al.*, 2018).* An IUCN Red List assessment due for publication in March 2019 has categorized the species as Critically Endangered as it is estimated that the population will have declined by 80% over three generations (two past, one future) (Mwaya *et al.,* 2019*). If these estimates are reliable the species may have lost over 50% (a marked recent decline) in the past two generations (based on an 80% decline split evenly over time, as it would equate to 27% per generation so 53% over two generations), although no information was available on whether the decline is calculated to remain constant over the entire three generations.*

Habitat for this species is rapidly deteriorating in extent and quality. Habitat degradation and loss from the destruction of its rock crevice micro-habitat, often as a result of collection, is an additional threat to the species. The species is slow to mature; age at maturity in captivity is reported at 5–9 years. Life expectancy was estimated at 25 years, and it has very low fecundity, with only one or occasionally two eggs laid annually in the wild at the onset of the wet season (Mwaya *et al*., 2018). Information on relative declines in ranges States are detailed below:

Kenya: It was reported in 2008 that collection of specimens for international trade has been identified as a major threat to populations in a number of sites.

Surveys in 2001/2002 took in 98% of the species' range (P. Malonza, in litt., *2019) (presumably only 98% of the range within Kenya).* Densities were recorded at eight sites during these surveys ranged from 8.86 tortoises/km^2 in Voo, Kitui County to 1.2 tortoises/km^2 at Nguni—the low population density in the Nguni area was reported to be due to past collection. *Tortoise collectors have visited most of the areas surveyed in Mwingi district and have had depleting effects in some areas (Malonza, 2003).*

A repeat survey was undertaken in 2014 *which covered at least 50% of the 2001/2002 sites. The total number of tortoises seen (62) (Ngwava, 2015) equates to a very low encounter rate per person hour (4 x researchers, 8 hour days, 52 days): 0.04 tortoises/ph (similar to the encounter rate in exploited areas in Tanzania (see below)).* The 2014 survey found no Pancake Tortoises in Voo, but a commercial farming operation for Pancake Tortoises had been established and likely specimens from the local wild population have been collected to provide breeding stock for the farm. Other areas in Kenya surveyed in 2001/2002 that had good Pancake Tortoise populations at that time were also noted to be depleted in 2014*, although it was a particularly dry year which may have contributed to low encounter rates (Ngwava, 2015).*

Tanzania: Results of preliminary surveys indicated that in less than 10 years of intensive collection (1985–1995), the Pancake Tortoise had become severely threatened throughout its range in Tanzania.

Recent survey data from Tanzania is lacking. Surveys in 1992 compared six unexploited areas with five exploited areas, finding the latter on average was significantly lower (2.42 tortoises/ph vs. 0.27 tortoises/ph) (Klemens &

Moll, 1995) (which equates to around 90% lower). At the time collectors said it was becoming increasingly difficult to find tortoises because of intensive collection, some collectors noted harvesting large numbers (e.g. 400 in one year for one dealer). Physical evidence of crevice destruction due to collection was observed (Klemens & Moll, 1995).

Studies in Tanzania have demonstrated that collection for trade decreased population densities and changed age-class compositions towards a larger proportion of juveniles in wild populations (as adults were specifically targeted), as compared to undisturbed populations. *This may suggest that the wild specimens are being used for breeding stock in farms, rather than being directly exported as international trade tends to favour juveniles.*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Information provided in the section above refers to harvest for trade impacting populations.

Between 2000–2016, exporting countries reported the trade of 34,336 live Pancake Tortoises (CITES Trade Database). The largest exporters were:

- *Zambia = 23,310 (21,830 "C", 350 "W")*
- *Tanzania = 5,038 (4,988 "F", 50 "W") (United Arab Emirates reported importing 300 wild in 2009)*
- *Democratic Republic of the Congo = 3,000 (all "W" – DRC has not reported any imports since 1975)*

Over this time period there has been a clear shift to exports declared as captive-bred ("C") or captive-born ("F") (Figure 1). Of the 24,694 reported as "C", 88% were exported from Zambia, 8% from Kenya and the rest from non-range States (all captive exports from Tanzania were "F"). Exports of "C" individuals have fluctuated; peaking at 6,366 in 2011 before dropping to just 100 two years later (Figure 1). Trade appears to fluctuate with a large peak in 2011, and a decline since 2014. Exporter data for Zambia was not yet available for 2017, but according to importers, trade in 2017 was approximately 1,170 live tortoises (nearly all from Zambia).

Zambia and Tanzania reported wild exports (350 and 50 respectively). Non-range States exported 3,193 tortoises declared as wild (94% from Democratic Republic of the Congo).

Japan, Hong Kong SAR, USA, Taiwan Province of China, Thailand and Costa Rica were the destination of nearly 90% of reported exports.

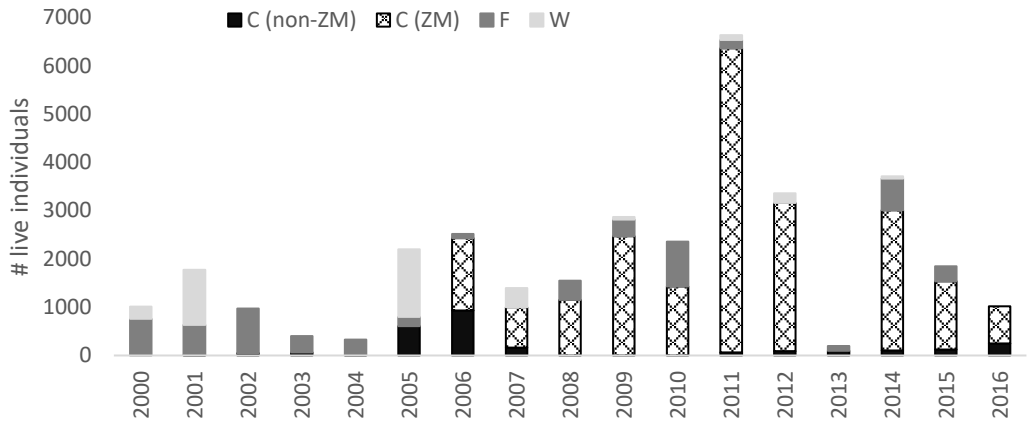

***Figure 1**. Source code reported by exporters of Pancake Tortoise (Source: CITES Trade Database). Source codes "I", "R", "U" not included, direct exports only.*

The species is frequently seen for sale in small quantities in South-east Asian pet markets and online (Malaysia, Indonesia, Thailand, Philippines) (S. Chng, in litt.*, 2019), online trade has also been observed in China (L. Chou,* in litt., *2019) and around 20 individuals were recorded in surveys in Japan (Wakao* et al., *2018).*

Illegal trade has been reported for this species:

- The CITES Secretariat reported the seizure of 370 specimens allocated to 13 seizure cases between 2000 and 2015.
- *EU Member States reported seizures of 15 live tortoises between 2008 and 2017 (EU-TWIX, 2019*).
- In 2006, officials in Hungary seized 55 *M. tornieri* and other tortoises from a lorry arriving from Serbia and destined to Rotterdam (the Netherlands).

- The Czech authorities are reported to have seized 888 *M. tornieri* in 2000.

Additional Information

Threats

Overexploitation for commercial trade is considered the single most important threat to the species in all range States. *An additional related factor threatening the species is the destruction of rock crevices by collectors allowing freer access to the crevice inhabitants (Malonza, 2003).*

Habitat degradation from rock destruction is also an important factor affecting the Pancake Tortoise, especially as related to illegal tortoise collection (range-wide), construction purposes and slab and ballast extraction in Kenya, and kiln building for charcoal production in Zambia. Vegetation removal through slash-and-burn cultivation, wildfires and charcoal burning are also detrimental. In Zambia's Nakonde District, 98% of land area has been reported to be deforested through land clearance for agriculture, firewood collection and charcoal production.

Conservation, management and legislation

The genus *Malacochersus* was listed in Appendix II in 1975. A proposal was submitted by Kenya and the USA at CoP11 (2000) to transfer the Pancake Tortoise from Appendix II to I, which was withdrawn *following the results of findings of the Animals Committee mission to Tanzania in 1998 (although it is not clear on what basis) and further due to recommendations for development of strict management measures for captive breeding and trade of the species (Kyalo, 2008). The species has been listed on Annex A of the EU Wildlife Trade Regulations since 1997 (Species+, 2019).*

The species was included in the 1988 Significant Trade in Wildlife: Review of Selected Species (IUCN and CITES Secretariat,1988) and subsequently the SC recommended a trade suspension for Tanzania which was in place 1993–2018. Since 1998 the suspension was restricted to wild exports, but not to specimens produced from ranching or captive breeding operations, for which the annual export quota has to be agreed between the Management Authority and the Secretariat. *The quota increased from 719 in 2000 to 940 in 2017 (Species+, 2019). At SC70 (2018), the Standing Committee agreed to remove the trade suspension subject to the publication of a zero export quota on the CITES website, although it not clear if this relates only to wild or all specimens (SC70 SR, 2018).*

In Kenya hunting and trade has been prohibited since the late 1970s. However, there is an exception which permits captive breeding. *In Tanzania, the species is listed in the Wildlife Conservation (National Game) Order which means it is protected under the Wildlife Conservation Act, 1974 (W. Crosmary,* in litt., *2019).*

It has been estimated that only 23% of *M. tornieri* habitat is currently protected across its entire range. *It was reported in 2003 that the stronghold areas for the Pancake Tortoise in Kenya occur on private lands (>95%) (Malonza, 2003).*

Artificial Propagation/captive breeding

Very little information is publicly available on the operation of breeding facilities (such as the number and origin of breeding stock, offspring produced, ratio of adults to hatchlings, marking of animals etc.). Given the very low reproductive rate of the species concerns have been expressed about the accuracy of source codes used in trade. While the species has been successfully bred in captivity, it is highly unlikely that the large numbers of animals exported as “R”, “F” and “C” do indeed originate from captive production. *One expert has previously noted that the species is considered difficult to breed in captivity, particularly on a large scale due to its low reproductive rate, and because it is aggressive, territorial and stress-sensitive (AC27 Doc. 17 (Rev. 1), 2014).*

Kenya legally discontinued exporting Pancake Tortoises from the wild in 1981, and since then at least three breeding farms have been licensed. At least four farms have been licensed in Tanzania. *The Tanzanian Management Authority reported that there are currently two zoos that hold the species in the country, and six breeders (SC70 Doc 29.2, 2018). In Tanzania, commercial Pancake Tortoise farms inspected in 2001 appeared to be conduits for wild-caught animals (Mwaya* et al.*, 2019).*

Zambia started to regularly export the species as captive-bred in 2006, soon after a study had found the species to exist in one location in Zambia. With no public information available on the operation of breeding facilities in Zambia and only a single Zambian population described in an isolated, unprotected area and an estimated population of 518 animals, concerns have been raised that tortoises may actually originate from other countries. *The CITES Secretariat contracted TRAFFIC to visit Zambia in 2014 and inspect three captive breeding facilities, and while no obvious signs of any wild-caught tortoises were found, in some cases the quantity produced was rather high in relation to the known biological capacity of the species (SC66 Doc 41.1, 2016).*

In captivity, breeding and egg laying have occurred almost all year round, provided there is a good food supply. Reproductive potential in captivity is low: from one to six clutches per year have been reported, with clutches usually contained only a single egg, occasionally two eggs. Larger clutches are reported to have low fertility. Low fertility and hatch rate seem common in captive *M. tornieri* and was usually far below 50%. In contrast, survival rate of hatchlings appeared rather high between 32 and 100%. There is no consensus on whether it is challenging to raise the animals to sexual maturity in captivity.

Potential benefit(s) for trade regulation of a transfer from Appendix II to I

Listing in Appendix-I would mean captive breeding operations intending to export tortoises will need to be registered with the CITES Secretariat which may increase transparency.

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Inclusion of Glass Frogs of the genera *Hyalinobatrachium, Centrolene, Cochranella* and *Sachatamia* in Appendix II

Proponents: Costa Rica, El Salvador and Honduras

Summary: Glass frogs (family Centrolenidae) are so called because of their unique transparent abdominal skin. Glass frogs are distributed throughout the Neotropics, from Mexico to Bolivia, with an isolated group of species occurring in southeastern Brazil and northeastern Argentina. After a taxonomic revision of Centrolenidae in 2009, there are currently considered to be 11 (possibly 12) genera included within the family. Only four genera are included in this proposal (*Hyalinobatrachium, Centrolene, Cochranella* and *Sachatamia*) comprising over 100 species.

Information on the population sizes and trends of many species is scarce, although 30 are considered to be declining and 17 stable. Among the 104 species listed by the proponents, four are assessed on the IUCN Red List as Critically Endangered, 11 Endangered, 13 Vulnerable, four Near Threatened, 27 Least Concern and 23 Data Deficient. The main threats appear to be habitat loss and fragmentation, along with pollution, disease and climate change. It was estimated that the habitat for only seven of the species that have been assessed was stable or undisturbed.

Global trade data are not available for any glass frog species. Based on available data on imports into the USA from 2004-2017, international trade appears to mainly be in:
- live animals for commercial purposes (1,147 (260 wild))
- bodies for scientific or educational purposes (389 (all wild))
- specimens for scientific or educational purposes (1,408 (all wild); not possible to equate to number of frogs so not detailed further).

Live glass frogs have also been observed for sale online in Europe, and some illegal trade has been reported. The following species have been identified as being traded:

- ***Hyalinobatrachium fleischmanni*** was assessed as Least Concern in 2010. The primary threats were identified as deforestation and agricultural pollution. Reported imports to the USA from 2004-2017 included 842 live (203 wild) and six bodies (all wild).

- ***Espadarana prosoblepon*** (reported as *Centrolene prosoblepon*) and ***Teratohyla spinosa*** (reported as *Cochranella spinosa*) were both reported in trade, however, following the taxonomy of Guayasamin *et al.* (2009) (reflected in the current version of the CITES Standard Reference for amphibians (Frost, 2015)), both species would fall outside the scope of this proposal. They were both assessed as Least Concern in 2008. The major threats were identified as deforestation and agricultural pollution. Small quantities of both species were reportedly imported to the USA from 2004-2017: 57 live and 304 bodies (all wild) of *E. prosoblepon*, and six live and six bodies (all wild) of *T. spinosa*.

A number of species were assessed on the IUCN Red List as Least Concern between 2008 and 2010 (the main threats identified at the time were habitat loss and degradation), and only relatively small amounts of trade in live individuals and bodies into the USA between 2004 and 2017 were reported. This includes:
- ***Cochranella granulosa*** (12 live and 11 bodies, all wild);
- ***Hyalinobatrachium valerioi*** (50 live, all captive-bred);
- ***Sachatamia ilex*** (reported as *Centrolene ilex*) (20 bodies, all wild).

No trade into the USA was reported for most other species, although for a total of 201 live individuals (198 captive-bred and 3 wild-sourced) the specific species was not reported.

In most range States, harvest of these genera from the wild is prohibited or requires a permit.

The distinction of the species within these four genera is said to be difficult for non-experts, however some identification guides have been developed in recent years. Glass frogs have conserved morphology, and preserved specimens can lose color and distinctive features which poses challenges for identification.

Analysis: There is little information on the historical and current size of wild populations for most species of glass frogs. Some species within these four genera appear to have limited ranges while others are considered common. The main threats have been identified to be habitat loss and fragmentation, along with pollution and disease. Some species are known to be in international trade, with demand mainly for live individuals, bodies and scientific specimens. Although few trade data are available, reported levels of international trade are relatively low and there is no evidence that trade presents a threat to any of the species concerned. Based on the information available it therefore does not appear that any species in the four genera subject to this proposal meets the criteria for inclusion in Appendix II.

Other considerations: Identification guides have been developed to distinguish live individuals within the family Centrolenidae, however bodies and specimens are thought to be more difficult to identify. Under the current Standard Nomenclature Reference for amphibians, a total of seven (possibly eight) other genera of glass frogs, including some species that appear in trade, are not subject to this proposal and therefore would be excluded from an Appendix II listing should this proposal be adopted. The continual taxonomic changes within the family Centrolenidae may therefore pose implementation challenges, as species may be moved between genera.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

According to Guayasamin et al., *(2009), within the family* Centrolenidae *there are 11/12 genera and 146 species, including two subfamilies (* = new genus described in 2009; genera subject to this proposal are underlined):*
Centroleninae – Centrolene *(23 spp.),* Chimerella *(1 spp.)*,* Cochranella *(7 spp.),* Espadarana *(3 spp.)*,* Nymphargus *(34 spp.),* Rulyrana *(8 spp.)*,* Sachatamia *(3 spp.)*,* Teratohyla *(4 spp.),* Vitreorana *(8 spp.)* = Total 91 species*
Hyalinobatrachinae – Celsiella *(2 spp.)*,* Hyalinobatrachium *(28 spp.)* = Total 30 species*
*The placement of the 12th genus (*Ikakogi *(1 spp.)*) has not yet been resolved.*
The placement of further 24 species was not resolved because molecular data were not available, and morphological and behavioural characters did not provide unambiguous evidence for their placement (Guayasamin et al.*, 2009).*

Guayasamin et al.*, (2009) stated that the number of species in the family was unstable with new species and/or synonyms added each year. Their taxonomic revision added seven new genera to the family.*

*Four genera (*Centrolene, Cochranella, Hyalinobatrachium *and* Sachatamia*) are subject to this proposal, which according to Guayasamin* et al.*, (2009) totalled 61 species.*

The proponents consider 104 species to be covered by the proposal: *Centrolene* (39 spp.), *Cochranella* (24 spp.), *Hyalinobatrachium* (36 spp.) and *Sachatamia* (5 spp.), *including species that Guayasamin* et al., *(2009) had assigned to genera outside the scope of this proposal (e.g.* Cochranella armata -> Nymphargus armatus) *or that they considered to be unresolved (*Cochranella riveroi*).*

The 2019 version of the CITES Standard Nomenclature Reference for amphibians (Frost, 2015) notes that the subfamily Centroleninae *follows the taxonomy of Guayasamin* et al., *(2009), although gives a total of 120 species rather than 91 (and includes some but not all of the unresolved species).*

Range

Centrolene spp: Colombia, Ecuador, Peru, Venezuela
Cochranella spp.: Bolivia, Colombia, Costa Rica, Ecuador, Honduras, Nicaragua, Panama, Peru
Hyalinobatrachium spp.: Brazil, Colombia, Costa Rica, Ecuador, French Guiana, Guyana, Mexico, Peru, Venezuela

Sachatamia spp.: Colombia, Costa Rica, Ecuador, Honduras, Nicaragua, Panama (According to the IUCN Red List and Guayasamin et al., *2009).*

IUCN Global Category

Among the 104 species included in this proposal, four are assessed on the IUCN Red List as globally Critically Endangered, *11 Endangered, 13 Vulnerable, four Near Threatened, 27 Least Concern and 23 Data Deficient (giving a total of 82 that have been assessed). The remainder of the species have not been assessed.*

IUCN Red List Global Category	*Centrolene*	*Cochranella*	*Hyalinobatrachium*	*Sachatamia*
Critically Endangered	*Centrolene ballux* (2004) *Centrolene gemmatum* (2004) *Centrolene heloderma* (2004)	*Cochranella euhystrix* (2018)		
Endangered	*Centrolene azulae* (2004) *Centrolene hesperium* (2018) *Centrolene lynchi* (2004) *Centrolene petrophilum* (2016) *Centrolene pipilatum* (2004)	*Cochranella mache* (2006) *Cochranella megistra* (2016)	*Hyalinobatrachium anachoretus* (2018) *Hyalinobatrachium Esmeralda* (2016) *Hyalinobatrachium guairarepanense* (2004) *Hyalinobatrachium pallidum* (2004)	
Vulnerable	*Centrolene buckleyi* (2008) *Centrolene daidaleum* (2004) *Centrolene geckoideum* (2004) *Centrolene quindianum* (2016) *Centrolene peristictum* (2004) *Centrolene sabini* (2017)	*Cochranella balionota* (2004) *Cochranella riveroi* (2004) *Cochranella xanthocheridia* (2017)	*Hyalinobatrachium fragile* (2004) *Hyalinobatrachium ibama* (2004) *Hyalinobatrachium orientale* (2004)	*Sachatamia punctulata* (2016)
Near Threatened	*Centrolene antioquiense* (2016)	*Cochranella nola* (2004)	*Hyalinobatrachium aureoguttatum* (2008) *Hyalinobatrachium pellucidum* (2013)	
Least Concern	*Centrolene hybrida* (2004) *Centrolene notostictum* (2004) *Centrolene robledoi* (2014) *Centrolene savage* (2016)	*Cochranella erminea* (2017) *Cochranella euknemos* (2008) *Cochranella geijskesi* (2017) *Cochranella granulosa* (2008) *Cochranella guayasamini* (2018) *Cochranella resplendens* (2017)	*Hyalinobatrachium bergeri* (2004) *Hyalinobatrachium chirripoi* (2017) *Hyalinobatrachium colymbiphyllum* (2008) *Hyalinobatrachium crurifasciatum* (2004) *Hyalinobatrachium fleischmanni* (2009) *Hyalinobatrachium kawense* (2017) *Hyalinobatrachium mondolfii* (2004) *Hyalinobatrachium munozorum* (2008) *Hyalinobatrachium nouraguense* (2008) *Hyalinobatrachium ruedai* (2008) *Hyalinobatrachium talamancae* (2014)	*Sachatamia albomaculata* (2008) *Sachatamia ilex* (2008)

			Hyalinobatrachium tatayoi (2008) *Hyalinobatrachium taylori* (2004) *Hyalinobatrachium tricolor* (2017) *Hyalinobatrachium valerioi* (2008)	
Data Deficient	*Centrolene acanthidiocephalum* (2004) *Centrolene altitudinale* (2004) *Centrolene condor* (2009) *Centrolene guanacarum* (2004) *Centrolene huilense* (2004) *Centrolene lema* (2004) *Centrolene medemi* (2008) *Centrolene muelleri* (2004) *Centrolene ocellifera* (2004) *Centrolene paezorum* (2004) *Centrolene papillahallicum* (2004) *Centrolene scirtetes* (2004) *Centrolene solitaria* (2004)	*Cochranella duidaeana* (2004) *Cochranella litoralis* (2008) *Cochranella phryxa* (2008) *Cochranella ramirezi* (2004)	*Hyalinobatrachium duranti* (2004) *Hyalinobatrachium eccentricum* (2004) *Hyalinobatrachium iaspidiense* (2004) *Hyalinobatrachium ignioculus* (2004) *Hyalinobatrachium vireovittatum* (2008)	*Sachatamia orejuela* (2008)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

Population Status and Distribution

Information on population trends is scarce. It is estimated that the habitat for only seven species (*Hyalinobatrachium crurifasciatum, Hyalinobatrachium eccentricum, Hyalinobatrachium nouraguense, Cochranella riveroi, Centrolene charapita, Centrolene hybrida, Centrolene notostictum*) in the four genera is stable or undisturbed; the habitat of all other species is subject to deterioration and degradation. According to the IUCN Red List most species have unknown population trends, 30 are declining and only 17 are stable. The loss of forests in Central and South America reached a rate of more than nine percent between 1990 and 2015, and consequently, the habitat of most species of these genera has experienced a significant decline. *Most of the species of glass frogs have only been described in the last 40 years as they are often difficult to find in the wild, since much of their habitat is deemed inaccessible (Kubicki, 2003). Although recent progress has been made in determining the taxonomy, large knowledge gaps on the distributions, natural histories and status of glass frogs remain (Thompson* et al., *2018).*

Of the species estimated to be experiencing population declines, three are listed as Critically Endangered, nine Endangered, 11 Vulnerable, two Near Threatened and 16 Least Concern. For only four species does the assessment include an estimate of decline:

- Centrolene ballux*: at the time of assessment drastic population decline estimated to be more than 80% over the last three generations (Bolivar* et al., *2004a).*
- Centrolene buckleyi*: at the time of assessment population decline projected to be more than 30% over the next 10 years in much of its Ecuadorian range (Guayasamin, 2010).*
- Centrolene geckoideum*: at the time of assessment population decline estimated to be more than 30% over the next 10 years (Bolivar* et al.*, 2004b).*

- Centrolene heloderma*: at the time of assessment drastic population decline estimated to be more than 80% over the last three generations, inferred from the apparent disappearance of most of the population (Coloma* et al.*, 2004).*

Of the four genera, *Hyalinobatrachium* has the widest distribution and is found in tropical forests of Central America, the tropical Andes, the Cordillera de la Costa de Venezuela, Tobago, the Upper Amazon Basin and the Guyana Shield. A total of 20 of the 36 species of *Hyalinobatrachium* are endemic (*according to taxonomy in SS*). Ten are native to Venezuela; Costa Rica, Colombia and French Guyana each have two endemic species; and Brazil, Ecuador, Guyana and Peru have one endemic species each.

The genus *Centrolene* is distributed from the Cordillera de Merida in Venezuela, through the Andes of Colombia and Ecuador, to the Cordillera de Huancabamba in northern Peru. A total of 29 of the 41 species of *Centrolene* are endemic: 14 in Colombia, six in Peru, five in Ecuador, three in Venezuela and one in Guyana.

The genus *Cochranella* is found at elevations below 1,750 metres above sea level in Central America, the lowlands of the Pacific, the cloud forests of Colombia and Ecuador, on the Amazonian slopes of the Andes and in the Amazonian lowlands in Ecuador, Peru and Bolivia. Of the 24 species, 15 are endemics: Peru and Colombia have four endemics each, Bolivia has three, Venezuela has two, and both Ecuador and Suriname have one.

The genus *Sachatamia* is found at elevations below 1,500 metres above sea level in Central America (Honduras, Nicaragua, Costa Rica, Panama) and South America (Colombia and Ecuador). One of the three species is endemic to Colombia.

None of the Red List assessments report trade as being a threat, however most species were last assessed in 2008 or before so the situation may have changed.

International Trade
Imports into the USA totalled 2,138 specimens (*assumed all commodity types*) across all four genera between 2004 and 2016. *According to U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS), the USA imported a total of 1,178 live individuals between 2004-2017 (Table 1). A total of 1,147 live frogs were imported for commercial purposes and the remainder were for scientific and zoological purposes. A total of 389 bodies were also imported into the USA during this period, all for scientific purposes, as well as 1,408 specimens of which 1,183 were for scientific purposes, 162 for educational purposes and 63 for commercial purposes. It is not clear why there is a difference between the quantity and species reported in the SS and in the data obtained by IUCN/TRAFFIC as they are both derived from LEMIS data.*

Table 1: *Imports into the USA based on US LEMIS Data in Supporting Statement and US LEMIS data obtained by IUCN/TRAFFIC (all purposes).*

Species	IUCN Red List Global Category (assessment date)	USA LEMIS data from Supporting Statement 2004-2016 ("specimens")	USA LEMIS data obtained by IUCN/TRAFFIC 2004-2017 (No. live individuals)	USA LEMIS data obtained by IUCN/TRAFFIC 2004-2017 (No. bodies individuals)	USA LEMIS data obtained by IUCN/TRAFFIC 2004-2017 (No. specimens)
Sachatamia ilex (reported as *Centrolene ilex*)	*Least Concern (2008)*	891	0	20 (wild)	222 (wild)
Centrolene spp.	-	355	14 (captive-bred)	4 (wild)	171 (wild)
Espadarana prosoblepon (reported as *Centrolene prosoblepon*)	*Least Concern (2008)*	288	57 (wild)	304 (wild)	576 (wild)
Hyalinobatrachium spp.	-	222	187 (184 captive-bred, 3 wild)	10 (wild)	142 (wild)
Hyalinobatrachium fleischmanni	*Least Concern (2009)*	194	842 (693 captive-bred, 203 wild)	6 (wild)	66 (wild)
Cochranella granulosa	*Least Concern (2008)*	178	12 (wild)	11 (wild)	14 (wild)
Cochranella spp.	-	155	0	23 (wild)	121 (wild)

Teratohyla spinosa (reported as *Cochranella spinosa*)	*Least Concern (2008)*	41	6 (wild)	6 (wild)	45 (wild)
Hyalinobatrachium colymbiphyllum	*Least Concern (2008)*	16	0	5 (wild)	41 (wild)
Hyalinobatrachium vireovittatum	*Data Deficient (2008)*	8	0	0	8 (wild)
Hyalinobatrachium valerioi	*Least Concern (2008)*	0	50 (captive-bred)	0	0
Hyalinobatrachium aureoguttatum	*Near Threatened (2008)*	0	10 (wild)	0	0
Vitreorana helenae (reported as Cochranella helenae)	*Data Deficient (2004)*	0	0	0	1 (wild)
Sachatamia albomaculata	*Least Concern (2008)*	0	0	0	1 (wild)
Total		**2,348**	**1,178**	**389**	**1,406**

The SS reports that only live individuals are known in international trade. *Other commodities also reportedly imported into the USA include bodies, eggs and specimens, the majority for scientific purposes (US LEMIS Database).*

Of the 1,178 live specimens imported into the USA between 2004-2017, a total of 291 individuals were reported to be wild-sourced. Exporters of wild-sourced frogs included Nicaragua (69%), Suriname (21%), Panama (4%), Costa Rica (3%) and Ecuador (3%).

Of the 389 bodies imported into the USA between 2004-2017, all were reported to be wild-sourced. Exporters of wild-sourced bodies included Panama (388 bodies) and Honduras (1 bodies).

One farm in Ecuador (Wikiri) was authorized to begin commercial breeding of Hyalinobatrachium aureoguttatum in 2014 and Sachatamia albomaculata in 2017 (L. Guarderas, in litt., *2019), and between 2014 and 2019 exported a total of:*

- *94 live* H. aureoguttatum *and 22 live* S. albomaculata *to the USA*
- *61 live* H. aureoguttatum *and 9 live* S. albomaculata *to Europe*
- *21 live* H. aureoguttatum *and 5 live* S. albomaculata *to Canada*
- *Three live* H. aureoguttatum *to Ecuador.*

Glass frogs are regularly offered via the internet and at European fairs of reptiles and amphibians, mainly the fair in Hamm, Germany, which takes place several times a year. *It appears that online trade of* Centrolene *spp. is difficult to find even on rare amphibian trade platforms. Advertisements for* Cochranella *spp. and* Sachatamia *spp. can be found (however they do not appear to be common). Of the four genera it appears that advertisements for* Hyalinobatrachium *spp. are most commonly seen but advertise relatively low numbers of individuals (J. Janssen,* in litt.*, 2019). Online searches for other genera of glass frogs excluded from this proposal revealed small numbers of* Teratohyla *spp. and Espadarana spp. advertised for sale (Terraristik, 2019).*

There is reported to be illegal trade in some of the species from certain countries. *There have been no reported cases of legal or illegal trade from El Salvador but there is reported to be illegal trade in Costa Rica, and this could potentially pose a threat if it extends to other range States within the region (V. Henríquez in litt., 2019).* In 2017, a Dutch trader through the website Terraristik (www.terraristik.com) offered a large number of specimens of *Teratohyla spinosa* (reported as *Cochranella spinosa*), specifying them as "farm-raised" from Costa Rica. However, the Costa Rican authorities confirmed that there are no registered breeding facilities for this species and that all export of specimens captured from the wild is illegal.

Species known to be in trade and/or threatened include:

Sachatamia (Centrolene) ilex: *Assessed as Least Concern by IUCN in 2008. Native to Colombia, Costa Rica, Ecuador, Nicaragua and Panama, the species is known from scattered localities in humid lowland and premontane areas. At the time of the assessment the population size is unknown, and although the population was said to be declining and the species difficult to find, it could still be regularly recorded in suitable habitat. Reportedly threatened by habitat destruction but has been recorded in a number of protected areas (Solís* et al.*,*

2010). Imports were reported as both Centrolene ilex *and* Sachatamia ilex *and included specimens, bodies and eggs for educational and scientific purposes (US LEMIS Database).* In 2014, a German citizen was captured in Costa Rica attempting to smuggle species including 20 *S. ilex*: prior to the seizure the smuggler's partner announced online several species of glass frogs to be sold at a fair in Germany.

Espadarana (Centrolene) prosoblepon: *If the taxonomy of Guayasamin* et al., *(2009) is accepted this species would fall outside the scope of this proposal.* Espadarana prosoblepon *was assessed as Least Concern in 2008 due to its wide distribution, presumed large population and because the population was thought to be stable. The species occurs in Colombia, Costa Rica, Ecuador, Honduras, Nicaragua and Panama. At the time of the assessment the species was considered common in some sites, and in others there had been declines. The major threats are deforestation for agricultural development, illegal crops, logging, and human settlement, and pollution resulting from the spraying of illegal crops. Recorded declines in Costa Rica might be due to the disease chytridiomycosis (Kubicki* et al.*, 2010*). *A total of 57 wild-sourced specimens were imported into the USA between 2004-2017 for the purpose of commercial trade. Other imports included specimens and bodies for scientific and educational purposes (US LEMIS Database).*

Teratohyla (Cochranella) spinosa: *If the taxonomy of Guayasamin* et al.*, (2009) is accepted this species would fall outside the scope of this proposal.* Teratohyla spinosa *was assessed as Least Concern in 2008 and was considered at this time to have a wide distribution and presumed large population. The species occurs in Colombia, Costa Rica, Ecuador, Honduras, Nicaragua and Panama. In Colombia this species is considered common and it is regularly encountered in Costa Rica and Panama, whereas in Ecuador and Honduras it is considered uncommon. The major threats are deforestation for agricultural development, illegal crops, logging and human settlement and pollution resulting from the spraying of illegal crops (Coloma* et al., *2010a). A total of six live wild-sourced specimens were imported into the USA between 2004-2017 for scientific and zoological purposes. Other imports included bodies and specimens for scientific purposes (US LEMIS Database).*

Cochranella granulosa: *Assessed as Least Concern in 2008 and was considered at this time to have a wide distribution, presumed large population and tolerance to a degree of habitat modification. The species occurs in Costa Rica, Honduras, Nicaragua and Panama. It is regularly encountered in Costa Rica and considered to be common in Panama although there are few records from Honduras and Nicaragua (Solís* et al.*, 2010a). A total of 12 live wild-sourced specimens were imported into the USA between 2004-2017 for scientific and zoological purposes. Other imports included specimens, eggs and bodies for educational and scientific purposes (US LEMIS Database).*

Hyalinobatrachium aureoguttatum: *Assessed as Near Threatened in 2008 and at this time the extent of occurrence for this species was believed to be no greater than 20,000 km^2. This species was considered to be very common at the time of assessment but faced localised threats of habitat loss and fragmentation due to expanding agriculture, cultivation of illegal crops and water pollution. The species occurs in Colombia, Ecuador and Panama (Solís* et al., *2010b). Between 2004-2017 a total of 10 live specimens were imported into the USA, all of which were reported as wild-sourced (US LEMIS Database).*

Hyalinobatrachium fleischmanni: *Assessed as Least Concern in 2009 and was considered at this time to have a wide distribution, presumed large population and tolerance to a degree of habitat modification. The species occurs in Belize, Colombia, Costa Rica, El Salvador, Guatemala, Guyana, Honduras, Mexico, Nicaragua, Panama and Suriname. It was considered at the time of assessment to be reasonably common species in South America although it had declined substantially in Monteverde, Costa Rica and in montane areas in southern Mexico. At Monteverde, densities had declined from one male per metre of stream transect to a few males over several hundred metres (Coloma* et al., *2010b).* This species has been observed for sale at the Terraria far in the Netherlands and were offered for USD 50 each, and in Spain online advertisements offered a breeding pair for USD 120. *According to the US LEMIS Database, a total of 842 live specimens were imported into the USA between 2004-2017 of which 639 were reported to be captive-bred and 203 wild-sourced, all for commercial purposes. Other imports included specimens and bodies for scientific and commercial purposes.*

Hyalinobatrachium valerioi: *Assessed as Least Concern in 2008 and was considered at this time to have a wide distribution, presumed large population and a tolerance to a degree of habitat modification. The species range occurs in Colombia, Costa Rica, Ecuador and Panama. It was reported at the time of assessment to be moderately rare and patchily distributed in Ecuador and Colombia although in Costa Rica the population was considered stable. Identified threats included deforestation for agricultural development and human settlement, illegal crops and water pollution (Solís* et al., *2008a). Between 2004-2017 a total of 50 live specimens were imported into the USA, all reported to be captive-bred (US LEMIS Database).* This species has been observed for sale online; live specimens have been offered on Terraristik (www.terraristik.com) and in Spain individuals were advertised online for USD 100 each.

Additional Information

Threats

The main threat to glass frogs is the loss of habitat and fragmentation due to smallholder and industrial agriculture, including livestock and illegal crops. Habitat loss has also increased due to activities such as logging, mining, construction of human settlements and hydroelectric plants. Moreover, water contamination with herbicides, pesticides and oil spills is also a major threat to glass frogs, along with the disease Chytridiomycosis. In addition, climate change poses a threat to the stability of glass frog populations.

Conservation, management and legislation

There are currently no management measures or population monitoring schemes for these species. These species are not protected by any international legislation. National legislation within the range States affords varying levels of protection; for most species their extraction from the wild for commercial purposes is either prohibited completely or requires a permit. For example, in Colombia, Costa Rica and Panama, trade is allowed if adequate permits are required. *There is legislation that provides varying levels of protection for species within these genera in almost all their range States, however, there appears to be a gap in international efforts to curb illegal trade (V. Henríquez & F. S. Álvarez Calderón,* in litt.*, 2019).*

Artificial propagation/captive breeding

Of the live individuals reportedly imported into the USA between 2004 and 2016, about 7% were recorded as captive-bred, *and many of the individuals advertised on online forums are reported to be from captive-bred sources.* The largest exporter of captive-bred specimens to the USA was Canada (74 individuals exported for the purpose of commercial trade).

In some range States, captive breeding is taking place to supply commercial trade and benefit in situ conservation efforts. For example, the Costa Rican Amphibian Research Center has government approval which allows for the commercial export of limited numbers of animals to Canada (CRAC, 2019). Half of the proceeds from commercial sales are utilised to support in situ conservation projects (Understorey, 2019). Farms in Ecuador are producing *Hyalinobatrachium aureoguttatum* in captivity for the purpose of export - *Wikiri is a company based in Ecuador which captive breeds some species of glass frogs for commercial purposes and is said to be the only Ecuadorian company that has been authorised to commercialize amphibians for national and international markets. In 2014 Wikiri began the commercialization of* H. aureoguttatum *in June 2014 and* Sachatamia albomaculata *in 2017 (L. Guarderas,* in litt., *2019).*

Implementation challenges (including similar species)

The taxonomic classification of glass frogs is the result of a very complex combination of 18 morphological characteristics and 7 ecological characteristics. The same attributes of the most obvious characteristics, such as size, dorsal coloration, partial or complete transparency of the peritoneum and the presence of the humeral spine, can be shared by several species of different genera. Therefore, the selection and distinction of the species of *Hyalinobatrachium, Centrolene, Cochranella* and *Sachatamia* is very difficult for non-experts. *Species identification within* Hyalinobatrachium *is complex as species tend to have conserved morphology, in addition preserved specimens often lose colour and more distinctive features which complicates species identification (Guayasamin* et al., *2017).*

However, as research on these species has developed in recent years more knowledge has been gained on Centrolenid morphology and ID guides for species within the family Centrolenidae *are available online (BioWeb, 2019) and in print. One expert reported that many of the species within the family* Centrolenidae *appear to be morphologically distinct and considering the relatively small numbers that appear in trade, identification may not be an issue in terms of enforcement (L. Coloma.,* in litt., *2019).*

Searches for online advertisements of "glass frogs" find adverts for species from several other genera not included within this proposal, indicating there may be problems identifying the four proposed genera from other "glass frogs". Guayasamin et al., *(2009) note that "the morphological similarity between* Centrolene *and* Espadarana *is remarkable. At present, we are unaware of any single morphological character that would unambiguously differentiate species from the two clades". The genus* Sachatamia *cannot be morphologically differentiated from* Rulyrana *- however, the mitochondrial phylogeny indicates that* Rulyrana *and* Sachatamia *do not form a monophyletic group (Guayasamin* et al.*, 2009).*

The continual changes to the taxonomy of Centrolenidae *may also pose challenges in terms of implementation, as species may be moved in/out of the genera subject to this proposal depending on taxonomic revisions.*

Potential risk(s) of a listing

This proposal includes only four of the 11/12 genera within the family Centrolenidae*. There is a risk that demand may shift to non-listed genera should the proposal be adopted (or that specimens of species from the listed genera are mis-named as non-listed species to facilitate trade).*

The listing of species within the family Centrolenidae *on Appendix II could halt or hinder conservation efforts, research and sustainable management of species (L. Coloma,* in litt.*, 2019) and has the potential to discourage research and field investigations which could inform and prioritise conservation efforts (Mrosovsky, 1988).*

Potential benefit(s) of listing for trade regulation

There are legal tools for the conservation of these species in most of the range States, however international efforts in regulating trade and regional consistency in the protection of glass frogs could provide potential conservation benefits (V. Henríquez & F. S. Álvarez Calderón, in litt.*, 2019). An Appendix II listing could benefit sustainable harvest/captive breeding operations which could deter unsustainable collection and smuggling (S. Chng,* in litt.*, 2019).*

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Inclusion of Spiny Newts *Echinotriton chinhaiensis* and *Echinotriton maxiquadratus* in Appendix II

Proponent: China

Summary: The genus *Echinotriton* contains three species of spiny newts: *E. chinhaiensis* and *E. maxiquadratus* which are endemic to China, and *E. andersoni* which is now only found in Japan and is not included in this proposal.

Echinotriton chinhaiensis
Adult *E. chinhaiensis* are terrestrial and inhabit forests in low hills. The adult population was estimated to be around 300 mature individuals in 2004, and one of the three known subpopulations (the type locality) may have been extirpated. The habitat of *E. chinhaiensis* was estimated to total around 50 km^2 in 1999 and 30 km^2 in 2004 and is greatly fragmented. While part of the species' range is within protected areas, some of its range is being degraded by human activities (agriculture, pollution, tourism). *Echinotriton chinhaiensis* was assessed as Critically Endangered on the IUCN Red List (2004) due to its limited distribution, all individuals being in a single location, the low number of subpopulations, and continuing decline in the extent and quality of habitat. The number of breeding females reduced from 107 in 1999 to 82 in 2000 and 47 in 2008. The species is vulnerable to weather events such as typhoons and floods, which have caused a decline in the population. A number of attempts at *ex-situ* conservation have been made; individuals were collected from the wild and more than 800 larvae were released in 1998. The species is protected in the wild, meaning the hunting, catching or killing, as well as the sale and purchase or utilisation of the species and their products, is strictly prohibited within China. Records of the species in trade appear to be limited to low numbers observed for sale (two in a pet store in Japan, and a trader in Hong Kong SAR posting a picture on social media of at least five) and some discussion on online forums which could indicate demand.

Echinotriton maxiquadratus
This species was only described in 2014 and has not yet been assessed by IUCN. Like *E. chinhaiensis*, the species is considered to have a very small wild population with a restricted distribution; one expert stated it was known from two restricted areas of approximately 10 to 20 km^2 with populations each estimated to be less than 100 - 150 individuals. It faces many of the same threats as *E. chinhaiensis*, but due to its recent discovery little is known on population trends. The species is not protected under State law, although at least part of its range is within protected areas where harvest is prohibited. Little is known about the trade of this species, but due to the species' rarity it is highly likely to be in demand. One expert is aware of several specimens apparently being kept outside of China. *Echinotrition maxiquadratus* is morphologically very similar to *E. chinhaiensis*.

Analysis: The endemic spiny newt *E. chinhaiensis* has a very restricted, fragmented range in China (around 30 km^2) and is estimated to have a very small wild population (less than 400 adults) which is decreasing. It is known to be in demand for the hobbyist trade, although the species is protected in the wild. The species may already meet the biological criteria for inclusion in Appendix I and therefore would appear likely to meet the criteria for inclusion in Appendix II under criterion 2aA of *Res. Conf. 9.24 (Rev. CoP17)*. Less is known about the recently described *E. maxiquadratus,* but it appears to face a similar challenge of a very small wild population and very restricted range and is not yet protected (though part of its range is within protected areas). The possibility of trade seems high due to its recent discovery and rarity, and therefore also meets criterion 2aA for inclusion in Appendix II.

Other Considerations: *Echinotriton chinhaiensis* is protected in its only range State and any trade is already illegal, so if this proposal is accepted China could publish a zero-export wild quota on the CITES website to reflect national legislation. An Appendix II listing may help close the apparent loophole of specimens being illegally exported via Hong Kong SAR where they are not protected.

Japan may wish to list the third species in the genus, *E. andersoni,* in Appendix III to monitor any potentially increased trade in this species as a result of the other two species being listed.

It is reportedly difficult to differentiate *Echinotriton* and *Tylototriton* (the latter genus is also subject to a listing proposal (Prop. 41)). If one proposal is accepted then the other could be accepted for inclusion in Appendix II under criterion 2bA for look-alike reasons.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

There are three known species in the genus *Echinotriton*. The proposal includes the two species that are endemic to China, *E. chinhaiensis* and *E. maxiquadratus.* Echinotriton andersoni (native to Japan (*an old record exists for Taiwan Province of China where the species is now presumed extinct (Kaneko & Matsui, 2004*)) is not included in this proposal. The genus of *Echinotriton* has a disjunct geographic distribution.

Echinotriton chinhaiensis was originally named as *Tylototriton chinhaiensis* when it was discovered and described in 1932. Synonyms include: *Echinotriton chinhaiensis: T. chinhaiensis; Pleurodeles chinhaiensis. Echinotriton maxiquadratus* was described in 2014, and is morphologically very similar to *E. chinhaiensis.*

Range

China.

IUCN Global Category

Echinotriton chinhaiensis: Critically Endangered b1ab(iii,iv)+2ab(iii,iv) (assessed 2004 *needs updating, ver.3.1*).
Echinotriton maxiquadratus: Not yet assessed.

Biological and trade criteria for inclusion in Appendix II (*Res. Conf. 9.24* (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

Echinotriton chinhaiensis

Adult *E. chinhaiensis* are terrestrial and live in low hills in Beilun and Yinzhou District in Ningbo City, Zhejiang Province. The habitat of *E. chinhaiensis* (including Ruiyansi timberland, Xinlu timberland and surrounding farming areas) had an area of 51 km^2 in 1999 and 30 km^2 in 2004. Overall, the habitat of *E. chinhaiensis* has been greatly fragmented, and the area of habitat has been reduced. Apart from those core habitats inside the Ruiyansi timberland (4.2 km^2) and local small protected areas, all the other habitats are in the process of degradation. Subpopulations in the surrounding areas west of Ruiyansi exist, however, these areas are inside cultivated farms. Pollution is relatively more severe in the surrounding areas of the farms. *Even habitat within protected areas is at risk: three ponds in Ruiyansi Forest Park that were the sites of a study in 2008 on the species' reproductive behaviour were visited again in 2015 and were found to be surrounded by destroyed habitat, and very few larvae were surviving a growing invasion of predators such as dragonfly larvae that were proliferating probably due to climate change (Hernandez, 2016). The species was assessed in 2004 as being Critically Endangered as at the time its Extent of Occurrence was less than 100 km^2 and its Area of Occupancy less than 10 km^2, all individuals were in a single location, and there was continuing decline in the extent and quality of its habitat, and in the number of subpopulations (Feng & Gu, 2004). More recently, the population was assessed nationally as critically endangered (Jiang* et al., *2016). A new population was discovered in the Shashan Mountains (Ningbo County) in 2014, but this area became a construction site and 80% of it was destroyed in 2015 (Hernandez, 2016).*

In 2004 the global population was estimated at less than 300 individuals (Feng & Huiqing, 2004). The Ruiyansi population was estimated at 318 to 369 in 1998 and 1999 *(in Ruiyansi Forest Park (Hernandez, 2016))*: the number of adults was estimated to be 296 with a density of 9.87 individuals/km^2 based on capture-recapture of females surveyed from 1997 to 1999. The SS states that the subpopulation in the type locality has gone extinct, *although one expert says it has not extirpated and has been protected (J. Jiang,* in litt.*, 2019*). *More recent surveys place the total adult population at 200 (Hernandez, 2016).*

The population of *E. chinhaiensis* has declined due to weather disasters like typhoons in 2007 and freezing in 2008, tourism development and crop planting. The fecundity in 2008 is less than 50% of that from 1998 to 2000: the number of breeding females greatly reduced, with 107 in 1999, 82 in 2000 and 47 in 2008. Recent surveys have shown that the number of nests of eggs found in Ruiyansi Timberland each year varies over 2015-2018 (82 nests in 2015, 33 in 2016, 32 in 2017 and 88 in 2018).

Observations of captive animals suggest that the species is late-maturing, and long-lived with lifespans at least 20 years.

Echinotriton chinhaiensis has been found in Hong Kong SAR and Japan in the illegal trade. Two live *E. chinhaiensis* were found to be sold in a pet store in Japan with prices as high as USD 1,400 per individual. *One expert has observed photos of the species on the Facebook page of a Hong Kong SAR exporter, which totalled at least five individuals, apparently all wild-caught (K. Wang,* in litt.*, 2019). The two species are not protected in Hong Kong SAR as they are only found in mainland China, therefore it appears the newts are being smuggled into Hong Kong SAR where they can then be exported legally (K. Wang,* in litt.*, 2019).* There is demand for this species as a pet in the USA and EU based on internet searches (detailed in the SS) *and also Japan (K. Wang,* in litt.*, 2019)*. Illegal collection of *E. chinhaiensis* was reported in 2011.

Some scientific institutions and universities both in China and abroad have collected specimens of *E. chinhaiensis* with the approval of the authorities. Very few individuals have been collected and raised by animal collectors in China (but see the SS which details a case).

Echinotriton maxiquadratus
Lives in low depressions near the tops of mountains in north-east Guangdong Province*. The species was described based on a single holotype, an adult female collected in southern China in November 2013.* Echinotriton maxiquadratus *is clustered at mtDNA level with* E. chinhaiensis*, and the two species are geographically separated by 600 - 1,000 km (Hernandez, 2016).*

Some of the habitats of *E. maxiquadratus* are located inside a provincial nature reserve and protected areas. However, the habitat quality is reducing, and the habitat area is gradually decreasing due to the effect of commercial plantations and tourism in the surrounding mountains. Increasing climate warming may induce further loss of habitat of the species. Echinotriton maxiquadratus *is native to two mountains in south-eastern China: the second locality was found later in July 2015 during a field study, and is considered very sparsely populated (Hernandez, 2016). The exact localities have not been disclosed in an effort to protect the species from poaching, but these sites are under pressure from tourism, human presence and habitat destruction (Hernandez, 2016).*

Echinotriton maxiquadratus *has not yet been assessed on the IUCN Red List.* Although few surveys have been done on the distribution and population status of *E. maxiquadratus, based on available information it is considered likely to be critically endangered based on criteria A3 (>80% projected decline within next three generations) and B1(a, b) (extent of occurrence <100 km^2, severely fragmented and declining) (Hou* et al.*, 2014).* Only six breeding ponds, about ten individuals and a few larvae have been recorded in two distribution sites in field surveys from 2011 to 2016. Ten individuals were found in 2013 when this species was first discovered. However, no individual was found in the type locality during several surveys after 2013. *This species was unknown to science and even local people for a surprisingly long time despite dense human population and heavy human activity in this region, suggesting a very small population size. Experts believe it is highly likely that the new species will be pursued by hobbyists and pet dealers due to its rarity (Hou* et al.*, 2014). Available information suggests it occurs only in two very restricted areas of approximately 10 – 20 km^2 each with populations of less than 100 - 150 individuals (Hernandez, 2016).*

The population trend is not clear for *E. maxiquadratus* since it is newly discovered. The species might face similar threats as *E. chinhaiensis*, and the population appears to be reducing based on the results of field surveys. The quality and area of habitat are gradually decreasing due to the effect of agricultural pollution, agricultural and forest industry and tourism revealed by surveys in 2011. Human activities are increasing in the surrounding forests of the known distribution sites of *E. maxiquadratus.*

Echinotriton chinhaiensis and E. maxiquadratus are similar in biological characteristics, especially in their reproductive biology such as low fecundity and reproductive success and their specific spawning microhabitat requirements.

Although no trade has been recorded for the newly discovered *E. maxiquadratus,* the possibility of illegal hunting and trade is high. For this reason, authors of *E. maxiquadratus* concealed the locality information, urged all hobbyists to refrain from collecting and boycott any trade. There have been unverified records of illegal hunting and collection with prices as high as RMB5,000 *[USD 736]* per individual. *One expert noted that a few individuals are apparently being kept by a French amphibian keeper who displayed photos on their Facebook page (K. Wang,* in litt., *2019).* The only known specimen of *E. maxiquadratus* (female) which was used to describe the morphology of the species is kept in Shenyang Normal University.

Additional Information

Threats

Along with threats from illegal collection, the decline in habitat quality, fragmentation and the loss of reproductive ponds are the main threats. Natural disasters like typhoons also threaten the species' habitats.

Conservation, management and legislation

The hunting, catching or killing, as well as the sale and purchase or utilisation of the species and their products, are strictly prohibited according to the 1989 listing in the second class of the List of Wildlife under Special State Protection of China. There are special exemptions e.g. for scientific research which need permission from the provincial wildlife conservation authorities. A small nature reserve with an area of 8.7km^2 around Runyansi Timberland was established in 1996. As *E. maxiquadratus* is newly discovered it is not included in any conservation list. However, some of its range is within a nature reserve where the illegal harvest of any individuals is prohibited.

Artificial Propagation/captive breeding

A number of attempts at ex-situ conservation have been made; *E. chinhaiensis* individuals were collected from the wild and more than 800 larvae were released into the wild in original nature reserves after hatching in 1998. Zhejiang Chaiqiao Middle School have conducted research on artificially assisted hatching and metamorphosis since 2008 and released more than 100 larvae of *E. chinhaiensis* into the Ruiyansi Timberland.

Implementation challenges (including similar species)

Echinotriton chinhaiensis was originally named as *Tylototriton chinhaiensis*. While *Echinotriton* can be differentiated from members of *Tylototriton* by having sharp-tipped ribs that penetrate the enlarged dorsolateral warts (they do not penetrate in *Tylototriton*), *it is difficult for non-specialists to differentiate between Tylototriton and Echinotriton (M. van Schingen.,* in litt.*, 2019).*

Echinotriton andersoni, which is not included in this proposal and is not included in the Appendices, is distributed in the Ryukyu Islands (Japan) (*and formally Taiwan Province of China (Kaneko & Matsui, 2004*)) and has the key feature of two longitudinal rows of warts on each side of the back, inner row small and sparse. *Echinotriton* and *Tylototriton* might enter the pet market together in the USA.

Potential risk(s) of a listing

If the proposal is accepted, E. andersoni *would be the only species in the genus not listed in Appendix II which may cause implementation problems, or put more pressure on this species, which was assessed as Endangered on the IUCN Red List in 2004.*

Other comments

Echinotriton andersoni *was listed in Annex D of the EU Wildlife Trade Regulations in 2009. No legal imports have been reported into the EU since its listing. Imports into the USA of nine skeletons and nine specimens of this species (all wild, for scientific purposes) were reported between 2007-2014 (USFWS LEMIS Database). Illegal trade has been reported: 10 live* E. andersoni *were seized in 2015 in Belgium having been imported from Japan (Musing* et al.*, 2018) and the species was observed for sale online in the UK (UNEP-WCMC,2016).*

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Inclusion of Asian Warty Newts *Paramesotriton* spp. endemic to China and Viet Nam in Appendix II

Proponents: China and European Union

Summary: This proposal is for the inclusion of all species of the genus *Paramesotriton* endemic to Viet Nam and China in Appendix II, with the exception of *P. hongkongensis,* which is already included in Appendix II. All currently described species in the genus are endemic to southern China and northern Viet Nam. In the past 15 years the number of described species in the genus has doubled and the updated version of the CITES Standard Nomenclature Reference for newts recognises 14 species.

Each species is thought to have a restricted distribution and to only occur in a few, small known populations. Sexual maturity is normally reached at between three and seven years (sometimes up to 10 years). They are found in or close to forest streams; the adults of some species can be found in water all year round while others become particularly aquatic during the breeding season.

Information on the population size and status of many of the species is lacking. Six of the 14 currently described species have been assessed on the IUCN Red List (one as Endangered, one Vulnerable, two Near Threatened and two Least Concern). Most of these assessments were conducted in 2004 and require updating. Based on China's Red Data List (2016), of the ten species that were assessed, four were endangered and three vulnerable. Wild populations are threatened by habitat loss (deforestation and infrastructure development) and some species are exploited for the Traditional Asian Medicine, food and pet trades. *Paramesotriton hongkongensis* was listed in Appendix II at CoP17, although no trade has been reported for this species in the CITES Trade Database to date. All species in the genus have been listed in Annex D of the EU Wildlife Trade Regulations since 2009.

Some of the species are protected within range States, and commercial imports of all *Paramesotriton* species into the EU and the USA (two of the major markets for the pet trade) have been prohibited since 2018 and 2016 respectively due to concerns over the spread of disease. Illegal collection and trade have been reported. Captive breeding is possible for some species, but the extent to which this is happening appears to be limited to date.

Trade is reported to be mainly in live or whole dried or preserved animals. However, available trade data is restricted predominantly to live animals exported to Europe and the USA for pets and does not capture the volume of harvest and trade for traditional medicine in Asia, or the domestic food or pet market. Imports of live *Paramesotriton* into the EU between 2009-2017 totaled over 1,600 live animals. Imports into the USA involved more than 38,000 individuals between 2000-2016 (species-specific data were only available for the period 2007-2013). The main species reported in trade (aside from *P. hongkongensis)* were:

- *Paramesotriton labiatus:* Endemic to China, this species was previously recognised as *Pachytriton labiatus* including in the latest IUCN assessment (Least Concern, 2004). In 2016 the species was nationally assessed as vulnerable. *Paramesotriton labiatus* (reported as *Pachytriton labiatus*) was the species of this genus imported into the USA in the highest number between 2007 and 2013 (8,400 live, all wild). At the time of the Red List assessment, *Pachytriton labiatus* was considered to have a wide distribution and presumed large, though declining, population. Over-exploitation for use in traditional Chinese medicine and for the international pet trade was identified a major threat, as was habitat destruction and degradation.

- *Paramesotriton chinensis:* Endemic to China, recent research has concluded this species actually encompasses a number of different lineages (e.g. *P. longliensis, P. yunwuensis, P. fuzhongensis, P. labiatus, P. qixilingensis*), and many *Paramesotriton* species recognised in the CITES Standard Nomenclature Reference still enter the international trade as

P. chinensis. It was nationally assessed as near threatened in 2016. Imports of 1,100 live *P. chinensis* were reported into the EU from 2009-2017 (source unspecified), and 1,400 live into the USA from 2007-2013 (1,100 wild). *Paramesotriton chinensis* was assessed as Least Concern in 2004 and was considered common but the population was declining. At the time of assessment, the international pet trade was not reported to be a threat. It is also kept as a pet in China.

Other more threatened species were reported in lesser quantities in trade (although they may have been reported under incorrect names either because they were mis-identified or because the species had not been described at the time):

- *Paramesotriton fuzhongensis*: Endemic to China. The species was assessed by IUCN as Vulnerable in 2004 as at the time the extent of occurrence was less than 20,000 km^2, with all individuals in fewer than ten locations, and there was a continuing decline in the extent and quality of habitat and in the number of mature individuals. At the time of the Red List assessment it was considered rare and the population to be declining. Habitat loss and over-harvesting for the pet trade were identified as major threats). However, the current status is unknown. Nationally it was considered vulnerable in 2016. Many specimens reported as *P. chinensis* in trade were believed to be mis-identified *P. fuzhongensis*. Although no trade data are available, the species has been observed for sale in Europe.

- *Paramesotriton guangxiensis:* Endemic to China and Viet Nam, *P. guangxiensis* was assessed in 2004 as Endangered since the area of occupancy was less than 500 km^2, none of which was within protected areas. The Chinese population was assessed as endangered in 2016. Wild specimens have been observed for sale in pet shops in Viet Nam (including animals originating from China). *Paramesotriton guangxiensis* was formerly treated as a synonym of the morphologically similar *P. deloustali.*

- *Paramesotriton zhijinensis:* Endemic to China, *P. zhijinensis* is considered to have a restricted range. While it has not yet been assessed by IUCN (it was described in 2008), it was nationally assessed as endangered in 2016. No information on trade could be found, although the species resembles *P. chinensis* so may be traded under that name.

Species identification is difficult - especially if animals are traded in a dried state for traditional medicine. Imports into the EU and USA are reported to be frequently labelled incorrectly (as *P. hongkongensis* or *P. chinensis*). Whilst it does appear possible to distinguish *P. hongkongensis* from other similar species based on morphological characteristics, identification by non-experts may be difficult. The genus is relatively understudied and future taxonomic work is likely to lead to more species being described. Species of the genus *Pachytriton* (not currently listed in the Appendices) are also reportedly difficult for non-experts to distinguish from *Paramesotriton.*

Analysis: There is little information available on the wild populations of most *Paramesotriton* species, although many are believed to have small ranges and probably low population sizes. Habitat loss and degradation is a significant threat. Species are used for traditional medicine in Asia, for some species it is thought this could be in significant volumes, although no quantitative information is available. Some species are also traded domestically and internationally as pets. The only data available on legal international trade relate to imports reported by the EU and USA, which have both recently prohibited commercial imports due to concerns over disease. Trade has also been reported between China and Viet Nam.

For most species there is not enough information to determine whether current levels of international trade are having an impact on wild populations, particularly with restrictions on trade to EU and USA markets. A number of species are globally Endangered (*P. guangxiensis)* and/or nationally endangered (e.g. *P. guangxiensis, P. longliensis, P. yunwuensis, P. zhijinensis*), and it seems possible that certain species meet the Appendix I biological criteria (e.g. *P. maolanensis* which is known only from one 60 m^2 pool, although no information was found on international trade).

Morphological identification of species in this genus is considered to be difficult or even impossible by a non-specialist. Although it is probably possible to differentiate some species when traded live, this is likely much more difficult for dried specimens. It is reported to be difficult for non-experts to identify some *Paramesotriton* species from *P. hongkongensis,* which is already listed in Appendix II, so these species may meet the look-alike criterion in 2bA.

The proponents state that they wish to include all species in the genus that are endemic to China and Viet Nam. However, based on past convention it would seem more logical to list *Paramesotriton* spp. (populations of China and Viet Nam), which would not be expanding the scope of the proposal and would mean that if a species' range was found to extend outside of China/Viet Nam, the national populations in China/Viet Nam would still be covered.

Other Considerations: As some species are nationally protected and thus trade is illegal, the range State concerned could put in place a voluntary zero quota listed on the CITES website for those species and empower re-exporting and importing Parties to assist with enforcing the law.
Due to the evolving taxonomy of *Paramesotriton*, there is potential for the current CITES Standard Nomenclature Reference to become out of date. The CITES Standard Nomenclature Reference used for the newt species currently listed in CITES is Frost (2015). CoP18 Doc. 99 recommends a change to a 2017 version of Frost which recognises all 14 species of *Paramesotriton* currently described.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

The CITES Standard Nomenclature Reference for the newts currently listed in the CITES Appendices is the Taxonomic Checklist of Amphibian Species extracted from Frost (2015). As this checklist is an online resource constantly being updated, it is not possible to immediately confirm how many species of Paramesotriton *were recognised in 2015. However, one species in the genus* Paramesotriton *has been described since the checklist was extracted in 2015 (*P. aurantius *(2016)). The current version recognises the 14 species listed in the table below (Frost, 2019) as does the 2017 version recommended as the new Standard Reference (see AC29. Doc. 35 Annex and CoP18 Doc. 99). Further taxonomic revisions may lead to additional species being described.*

IUCN Global Category and Range

In addition to the global IUCN Red List assessments, the threat status on the China Red List is also provided where available (Jiang et al.*,2016).*

Species and year of description	***IUCN Global Category***	***Threat status in China*** *(from Jiang et al., 2016)*	***Range***
P. guangxiensis (1983)	*Endangered (2004 – needs updating) ver.3.1*	*endangered*	*China, Viet Nam*
P. fuzhongensis (1989)	***V****ulnerable (2004 – needs updating) ver.3.1*	*vulnerable*	*China*
P. caudopunctatus (1973)	*Near Threatened (2004 – needs updating) ver.3.1*	*vulnerable*	*China*
P. hongkongensis	*Near Threatened (2004 – needs updating) ver.3.1*	*near threatened*	*China, Hong Kong SAR*
P. chinensis (1859)	*Least Concern (2004 – needs updating) ver.3.1*	*near threatened*	*China*
P. deloustali (1934)	*Least Concern (2017) ver.3.1*	-	*Viet Nam*
P. labiatus (1930) *(as Pachytriton labiatus)*	*Least Concern (2004 – needs updating) ver.3.1*	*vulnerable*	*China*
P. aurantius (2016)	*Not assessed*	-	*China*
P. longliensis (2008)	*Not assessed*	*endangered*	*China*
P. maolanensis (2012)	*Not assessed*	*data deficient*	*China*
P. qixilingensis (2014)	*Not assessed*	-	*China*
P. wulingensis (2013)	*Not assessed*	-	*China*

P. yunwuensis (2010)	*Not assessed*	*endangered*	*China*
P. zhijinensis (2008)	*Not assessed*	*endangered*	*China*

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

Species in the genus *Paramesotriton* are endemic to northern Viet Nam and southern China and are found in or close to forest streams. The different species generally have small distribution ranges and in some cases are only known from the type locality, and there is a general lack of knowledge regarding their status and distribution. *As current populations appear to be extremely small, they can no longer sustain the high levels of harvest that occurred in the past (M. van Schingen,* in litt.*, 2019).* They are relatively slow breeders, reaching sexual maturity on average at three to 10 years. As *Paramesotriton* spp. generally form large breeding groups in breeding pools, high numbers can be collected from known sites during breeding season with minimal effort.

The trade of this genus includes mainly live or dried animals, as well as animals soaked in alcohol. In China they are consumed as food and traditional medicine, and for the domestic pet trade. In Viet Nam they are collected for use as traditional medicine and the domestic pet trade. The international pet trade has been identified as an increasing pressure, *although the volume of harvest for pets compared with traditional medicine is unknown but likely to be lower. Species of* Paramesotriton *have been very regularly available as wild-caught animals in the European pet trade over the past 20 years (although currently far fewer given legislation to restrict the spread of disease (BSal) (F. Pasmans,* in litt.*, 2019).* Usually the price for adults is higher than for juveniles; females may reach higher prices than males.

Evidence suggests that the level of harvest of *Paramesotriton* is far higher than the limited numbers in trade statistics might suggest. There is evidence for illegal trade occurring within each range country, between range States and into the international market.

Imports to the USA involved just over 38,000 individuals of *Paramesotriton* between 2000 and 2016, of which 50% were wild caught. The majority of these animals were traded live (90%), for commercial purposes (97%), but also for educational and scientific uses (3%).

A more detailed investigation of imports of live Paramesotriton *into the USA over a shorter time period (between 2007-2013) showed that of the live animals imported (almost 17,700), the most highly traded species were* Paramesotriton labiatus *(48%) (reported as* Pachytriton labiatus*),* P. hongkongensis *(43%) and* P. chinensis (8*%), and the remaining 1% comprised* P. caudopunctatus, P. deloustali*,* Laotriton laoensis *(reported as* Paramesotriton laoensis*) and* Paramesotriton spp*. Around half of imports (54%) were reported as being from the wild, of which the majority were* P. labiatus *(all of which were reported as wild) (U.S. Fish and Wildlife Service's LEMIS Database).*

According to the CITES Trade Database, imports of live Paramesotriton *into the EU between 2009 and 2017 totalled 1,621, all of which were reported as* P. chinensis *(68%),* P. labiatus *(32%) and* P. hongkongensis *(<1%).*

Several species known to be threatened and/or in international trade are discussed further below:

Paramesotriton labiatus: Endemic to China. *Paramesotriton labiatus* was still recognised as *Pachytriton labiatus* in the most recent IUCN assessment (2004) as *Paramesotriton labiatus* was only recognised in 2011. *The Taxonomic Checklist of Amphibian Species recognises* Paramesotriton labiatus *as a species, but not* Pachytriton labiatus *(Frost, 2015)*. Pachytriton labiatus *was assessed as Least Concern as at the time it was considered to have a wide distribution, presumed large population, and because it was unlikely to be declining fast enough to qualify for listing in a more threatened category. It was said to be a relatively common species, although the population was decreasing. Over-exploitation for use in traditional Chinese medicine and for the international pet trade were considered major threats to this species, as was habitat destruction and degradation (Huiqing & Zhigang, 2004). However, current status is unknown*. Several protected areas are present within its range. It is consumed in China as food and used as pets locally as well as nationally, and also used in traditional medicine locally. *Local inhabitants may sell them as juvenile Chinese Giant Salamander* Andrias davidianus *(AmphibiaWeb, 2015).* The SS states it to be one of the most commonly advertised species, and that it is known to have been bred in captivity successfully. The SS details offers for sale in the UK (GBP 18/ ~USD 23). *Between 2011 and 2013 the USA imported 8,397 live* Pachytriton labiatus *(presumably* Paramesotriton labiatus*) specimens (all wild), mostly exported from China directly (3,200) or from China via Hong Kong SAR (2,877) (U.S. Fish and Wildlife Service's LEMIS database).*

Paramesotriton chinensis: Endemic to China. *Paramesotriton chinensis* was formerly known as a single widely distributed species but in recent years has been found to actually encompass several different lineages (e.g. *P. longliensis, P. yunwuensis, P. fuzhongensis, P. labiatus, P. qixilingensis*). *The species was assessed as Least Concern in 2004 and was said at the time to be a common species although the population was considered to be declining (Huiqing* et al.*, 2004).* Currently, most *Paramesotriton* species still enter the international trade named as *P. chinensis or Paramesotriton spp. Paramesotriton chinensis* is frequently found for sale in Chinese pet markets and is the most common species in the international trade. *Ye* et al. *(1993) noted the collection of several hundred thousand* P. chinensis *and related species for export during a field survey in eastern China, although it is not known what the "related" species were. The 2004 IUCN Red List assessment notes that small numbers were exported for the international pet trade, though probably not at a level to constitute a threat to the species at that time (Huiqing et al., 2004). According to the CITES Trade Database, imports of 1,095 live* P. chinensis *have been reported into the EU (predominantly from China and Hong Kong SAR) since the genus was listed in the EU Wildlife Trade Regulations in 2009 (source of specimens not specified). Imports into the USA between 2007 and 2013 included 1,401 live* P. chinensis *(US Fish and Wildlife Service's LEMIS database).* In the USA prices for an adult were around USD 10 and rose to USD 40 in 2016, whereas in Europe an adult costs about USD 12.

Paramesotriton caudopunctatus: Endemic to central China. *Assessed as Near Threatened in 2004 because its Extent of Occurrence was probably not much greater than 20,000 km^2, and the extent and quality of its habitat were probably declining, thus making the species close to qualifying for Vulnerable. At the time it was said to be a very common species, but the population was decreasing, although it was reported that it may occur more widely than thought previously (Zhigang & Lau, 2004)*. Several protected areas are present within the range. In China *P. caudopunctatus* is collected from the wild to be used both locally and nationally in traditional medicine and as pets. The SS states it to be one of the most commonly advertised species, and wild specimens were offered for sale in the UK. The species is reportedly difficult to breed although it is known to have been successfully bred. *This species is affected by habitat destruction and degradation for dam construction and subsistence wood collecting, and also by harvesting for use in traditional medicine and international pet trade (Sparreboom, 2014). In 2004 it was noted that small numbers were exported for the international pet trade, though probably not at a level to constitute a threat to the species at the time (Zhigang & Lau, 2004). Imports into the USA between 2007 and 2013 were limited to two scientific specimens from Viet Nam (U.S. Fish and Wildlife Service's LEMIS database).*

Paramesotriton deloustali: *Endemic to Viet Nam but may extend into southern China. The species was assessed as Least Concern in 2017 as it was considered relatively widespread, although the population size was not known; however the population was thought to be declining due to habitat loss, logging, collection for trade (traditional medicine and pet) and pollution (IUCN SSC Amphibian Specialist Group, 2017). The species is considered endangered in the Vietnam Red Data Book in 1992 (Sparreboom, 2014). Imports into the USA between 2007 and 2013 totalled eight live* P. deloustali *reported as originating from Viet Nam (U.S. Fish and Wildlife Service's LEMIS database).* The SS states it to be one of the most commonly advertised species, and in 2018 the species was observed for sale in several pet shops in a number of Vietnamese cities, with new stock available each year around breeding season, and recent online prices were USD 7. The species has also been offered for sale in Germany and has been bred in captivity. *Paramesotriton guangxiensis* was formerly treated as a synonym of *P. deloustali.*

Paramesotriton fuzhongensis: Endemic to China. *Listed as Vulnerable (2004) as at the time its extent of occurrence was less than 20,000 km^2, with all individuals in fewer than ten locations, and there was a continuing decline in the extent and quality of its habitat and in the number of mature individuals. It was considered a rare species, and the major threats included habitat loss due to subsistence wood collection, and over-harvesting for the pet trade (Ermi & Zhigang, 2004).* The current status of the species is not known. Only a few protected areas overlap its range. *P. fuzhongensis* was being commercialized one to two decades prior to description as a new species. Known to be used locally. Many specimens reported as *P. chinensis* were actually *P. fuzhongensis*. In Europe the *P. fuzhongensis* was observed for sale for about USD 15. The SS states it to be one of the most commonly advertised species. It was observed for sale in the UK for GBP 25 (*USD 37*). A difficult species to keep, *but some European herpetologists have successfully bred the species in captivity (Ermi & Zhigang, 2004).*

Paramesotriton guangxiensis: *Apart from its type locality in China and one other location in Guangxi, China, it has been reported from northern Viet Nam (Sparreboom, 2014).* The species was assessed in 2004 as Endangered and the current population is said to be declining. Estimates in 2004 calculated an extent of occurrence of less than 5,000 km^2, and area of occurrence of less than 500 km^2, but the assessment only included the Chinese populations. The major threat to this species is habitat loss and degradation due to agriculture. The range in China is not included within any protected area, but part of its distribution in Viet Nam is under protection. A recent survey in 2018 found *P. guangxiensis* for sale at several pet shops in Ha Noi, Ho Chi

Minh City and Bien Hoa, Dong Nai Province with new stock being available every year around the species' breeding season, when animals can most easily be collected from the wild (the SS details offers for sale in Viet Nam at USD 15 each). In Viet Nam, *P. guangxiensis* appears less in the trade than the morphological similar *P. deloustali* and is sold for higher prices. One pet shop in the south of Viet Nam informed that the origin of *P. guangxiensis* previously in stock had Chinese and not Vietnamese origin, showing a complex trade network (the trade between range countries occurs usually in the direction of China). The SS also states it to be one of the most commonly advertised species.

Paramesotriton maolanensis: Endemic to China where it is known only from a single 60 m² pool.

Paramesotriton yunwuensi: Endemic to China, this species is known only from a single mountain chain. The species has not been assessed on the IUCN Red List; *it is considered to meet the criteria of Near Threatened (Wu* et al.*, 2010) although the authors did not state under which criteria.* At the type locality (a scenic park) local people collect and sell this species to tourists as juveniles of Chinese giant salamander *Andrias davidianus*, *which is probably not sustainable (Sparreboom, 2014).* They are also probably consumed as food.

Paramesotriton zhijinensis: This species is endemic to China, and is known from only one artificial pond fed by a subterranean spring of warm water, and also from slow-flowing, shallow streams. It has not yet been assessed by the IUCN Red List, *however due to its apparent restricted range it is "probably very vulnerable" (Sparreboom, 2014).*

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

Non-experts may find it difficult to distinguish P. hongkongensis *(Appendix II) from other similar species in the genus (IUCN and TRAFFIC, 2016).*

*At least two species (*P. labiatus *and* P. yunwuensi*) are known to be sold locally as juvenile Chinese Giant Salamanders* Andrias davidianus*, which has been listed in Appendix I since 1975. However, there is no information to suggest that this trade is international.*

Additional Information

Threats

The economic development and human population growth in the region has led to an increased consumption of natural resources (e.g. logging), associated with habitat loss and degradation, and increased construction of infrastructures (e.g. water dams, roads). Suitable habitats for *Paramesotriton* will likely decrease further in the future. Slash and burn practices for substitution with more profitable plants (e.g. paper industry or agriculture), dam constructions and pollution, are also common throughout the distribution range and thus affect habitat suitability for *Paramesotriton.* Fishing, tourism and climate change are also threats. Given that salamanders in general show both low vagility and reproduction potential, by staying in the near vicinity of their breeding habitat and by being long-lived and slow-breeding species, they have a rather low ability to colonize new habitats. Overharvest is considered a threat for a number of the species, and *as specimens of some species accumulate in fresh water habitats during the breeding season, they can be captured with nets and as such entire cohorts may be easily collected (M. van Schingen.,* in litt., *2019).*

Conservation, management and legislation

The genus has been listed in Annex D of the EU Wildlife Trade Regulations *since 2009.*

Detailed information on legal protection is provided in the SS. In summary some species are protected within range States, some appear not to be, and for others the status of protection is unclear. Harvest in protected areas is restricted, though harvest has been reported inside of protected areas. Some species do not occur in any protected area (e.g. *P. guangxiensis*).

Species in the genera native to mainland China but not Hong Kong SAR are not protected in Hong Kong SAR, therefore it appears the newts are being smuggled from mainland China into Hong Kong SAR from where they can then be exported legally to destination countries (K. Wang, in litt., 2019).

EU Decision 2018/320 (came into effect in 2018) requires that all members of the order Caudata *that are imported into the EU must be held in quarantine post-importation in an appropriate establishment, and that when being moved between EU countries they must be held in an appropriate establishment under the supervision of the competent authority, immediately prior to their exportation. The new rules seek to stop the spread of the*

fungal pathogen, Batrachochytrium salamandrivorans *(Bsal) which can be spread to native species and cause morbidity and mortality (CEFAS, 2018).*

In 2016 the USA listed 201 salamander species as "injurious wildlife" under the Lacey Act due to concerns over importing Bsal, including all species in the genus Paramesotriton*. This means importation into the USA is prohibited (there are some exceptions e.g. scientific research, zoos) (USFWS, 2016).*

Artificial Propagation/captive breeding
Although present in the pet trade, accurate information on captive breeding of *Paramesotriton* remains limited. *Paramesotriton caudopunctatus*, *P. chinensis, P. deloustali, P. fuzhongensis* and *P. labiatus* have successfully been bred in captivity – however, no published evidence has been found on captive breeding of the remaining species. Most species are reportedly difficult to keep (e.g. *P. fuzhongensis*) and breed (e.g. *P. chinensis and P. caudopunctatus*). Animals show aggressive behaviour and sensitivity to stress and need specific environmental conditions. Paramesotriton *are problematic in captivity due to their sensitivity to stress (Sparreboom, 2014). One group of experts noted that even when kept in captivity by knowledgeable and conscientious terrarium keepers, many of these animals will die in their first year of captivity, and to their knowledge very few keepers in Europe had managed to establish self-sustaining captive breeding colonies of Paramesotriton – at least partly because of the availability of cheap wild imports and late age of maturity which means captive breeding is less attractive (Sparreboom,* in litt., *to Shenyang Normal University, 2015).*

Implementation challenges (including similar species)
Species identification within the genus is difficult due to morphological similarities between species including *P. hongkongensis* - especially if animals are traded in dried state for traditional medicine.

Recent imports continue to misclassify these species, sometimes deliberately since rare and new species can be sold for a high price*. In the past, several shipments were imported into the EU and USA with misnamed specimens (e.g. as* P. chinensis*), where the shipments actually contained a large variety of species, including rare and threatened ones (F. Pasmans,* in litt., *2019).* Paramesotriton *species imported into the EU are mainly reported as* P. hongkongensis *or* P. chinensis *(M. van Schingen.,* in litt., *2019).*

The genera Pachytriton *and* Paramesotriton *have had a confused taxonomic history, complicated by the fact that a number of these salamanders have long been in the pet trade, identified as Paddle-tailed and Warty Newts, respectively (AmphibiaWeb, 2015).*

Paramesotriton spp*. are difficult (not to say impossible in most cases) to identify from each other without application of genetic tools. Genetic identification is necessary in most if not in all cases, as morphological variation is still poorly understood and not all cryptic taxa discovered. Thus, confusion of very rare, endemic species with more widely occurring taxa is very common (T. Ziegler,* in litt*., 2019).* Paramesotriton *can be differentiated from* Tylototriton *(F. Pasmans,* in litt., *2019); the latter genus is also proposed for inclusion in Appendix II (CoP18 Prop. 41).*

Laotriton laoensis *(endemic to Lao PDR) was previously considered a synonym of* Paramesotriton laoensis*, but it has since been included in a different genus (AmphibiaWeb, 2014). However, it is possible that* Paramesotriton *specimens may be being mis-labelled as* Laotriton laoensis *in trade (or vice versa) which could cause implementation issues as non-experts may not be able to differentiate these species.* Laotriton laoensis *was assessed as Endangered (2013) and is in high demand for the international pet trade (IUCN SSC Amphibian Specialist Group, 2014).*

Potential risk(s) of a listing
Given the difficulties in identification, a genus-level listing may encourage trade to be reported at the genus level as Paramesotriton spp.*, which could mask trade in the most vulnerable species.*

Other comments
Many species have proved to be exceptionally sensitive to being caught and confined to narrow enclosures, let alone being shipped long distances. This causes so much stress to the animals that it leads to diseases that mostly prove fatal. Packaging salamanders in large numbers often leads to diseases to which most animals in a shipment will succumb. Paramesotriton *are liable to be particularly sensitive due to their specific territorial behaviour (Sparreboom, in litt to Shenyang Normal University, 2015).*

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Inclusion of Crocodile Newts *Tylototriton* spp. in Appendix II

Proponents: China and European Union

Summary: The species in the genus *Tylototriton*, known as Crocodile Newts, are distributed within montane forests from the eastern Himalaya, through Indochina, to southern and central China. Their taxonomy is rapidly evolving, and the number of described species has increased threefold to 24 since 2010. Twenty species are thought to be endemic to one country and are mainly considered to be of restricted range and consist of few, small known populations, although it is likely that due to their cryptic nature the range of certain species is larger than currently known. Several currently recognised species are considered morphological complexes that may contain multiple species with smaller ranges.

Sexual maturation is reached at between three and five years, and clutches usually consist of less than 100 eggs. Seasonally they accumulate in breeding pools which leaves them susceptible to capture for the domestic and international pet and traditional medicine trades. Outside of the breeding season, adults and juveniles are mostly terrestrial and fossorial.

Thirteen of the species have been assessed on the IUCN Red List (two as Endangered, five as Vulnerable, three as Near Threatened, and three as Least Concern). Of the 14 species assessed on China's Red List (2016), six are threatened and seven near threatened. There is little population information for many species, although most are considered to be declining as a result of habitat loss and degradation (especially around breeding sites), unsustainable harvest and other factors.

It appears that at least some of the species are being impacted by international trade. The genus has been listed in Annex D of the EU Wildlife Trade Regulations since 2009. Available international trade data are restricted predominantly to live animals exported to Europe and the USA for pets, and do not capture the volume of harvest and trade for traditional medicine in Asia, or the domestic food or pet market, which appears to be significant for some species. Some of the species are protected within range States, and commercial imports into the EU and the USA (two of the major markets for the pet trade) have been prohibited since 2018 and 2016 respectively due to concerns over the spread of disease. Illegal harvest and trade have been reported. Captive breeding is possible for some species, but the extent to which this is happening appears to be limited to date.

Key species reported in trade (mainly as live) to the major markets of Europe and the USA include *T. kweichowensis* and *T. verrucosus* (considered a morphological complex), which are discussed below:

- *Tylototriton kweichowensis* is endemic to China and was assessed as Vulnerable in 2004 due to a restricted area of occupancy (<2,000 km^2) which was fragmented and declining. The number of mature individuals was also said to be declining. *T. kweichowensis* were imported in large numbers into Europe in the 1990s, and although captive breeding has occurred it is not clear on what scale. *T. kweichowensis* was the species of this genus reportedly imported into the EU in the greatest quantity (850 individuals) since the genus was listed in the EU Wildlife Trade Regulations in 2009. A further 1,130 wild and 350 captive-bred *T. kweichowensis* were imported into the USA between 2007 and 2013. Habitat destruction and degradation are major threats to this species, and it is also collected for use in traditional Chinese medicine.

- *Tylototriton verrucosus* was previously considered to be widely distributed but is now believed to comprise other related species with significantly smaller ranges and fewer populations. *T. verrucosus* was assessed as Least Concern in 2004, but this could change if future taxonomic revisions split the species. It was assessed as near threatened in China (2016). *T. verrucosus* was the species of this genus reportedly imported into the USA in the highest numbers (5,031 wild and 40 captive-bred live individuals between 2007 and 2013), is

commonly advertised for sale (in China, Europe and the USA), and was imported into Europe in the 1960s in large quantities for medical research (although these imports likely comprised other species not yet described). Some populations are highly threatened by habitat loss and degradation.

A number of other (often newly described) species with limited ranges and/or smaller populations have also been observed in trade. While there is little or no data on imports of these species into the EU or USA (suggesting the scale is less than *T. kweichowensis* and *T. verrucosus,* trade but potentially could have a greater impact if they are range-restricted/smaller populations) some specimens may have been traded under incorrect names. Examples of these include:

- *Tylototriton lizhenchangi* is an endemic with a restricted distribution described in 2012, and although not yet assessed on the IUCN Red List, it was nationally assessed as vulnerable (2016). One expert stated that intensive collection following its formal description had reduced the wild populations close to extinction (large adult individuals were already difficult to find in 2014 and 2015). No legal imports were reported into the EU (although it was offered for sale in Germany) or the USA (although it may have been imported under "*Tylototriton* spp.").

- *Tylototriton vietnamensis*, assessed as Endangered in 2016, has an estimated extent of occurrence of 1,345 km^2 and appears to be an uncommon species with small and fragmented populations. Tay Yen Tu Nature Reserve and Yen Tu Landscape Protection Area were reported to harbour the largest population of this species, and a survey of all known breeding sites in the Reserve in 2010 found 216 individuals. Unsustainable harvest is reported to be a threat to this species, in addition to intensive deforestation, climate change, and erratic rainfall. Local people are reported to collect newts for private medicinal use, or to sell to local tourists or at Chinese markets for the international pet trade. Limited legal imports into the EU and USA are reported but given historic confusion with *T. asperrimus* and others in the genus it is likely that some specimens have been traded under incorrect names.

- *Tylototriton wenxianensis,* endemic to China, was assessed as Vulnerable in 2004 due to its limited area of occupancy (<2,000 km^2) that was declining in extent and quality, as well as limited localities. It was nationally assessed as vulnerable in 2016. The global population was estimated at 30,000 in 2008 but was reported to have more than halved by 2015; threats included habitat loss and degradation. There are reportedly undescribed taxa within *T. wenxianensis,* meaning the population may be smaller should the species be split. No legal imports were reported into the EU (although it was offered for sale in Portugal and Spain) or the USA (although it may have been imported under "*Tylototriton* spp.").

- *Tylototriton yangi* is endemic to China and reported to be highly threatened by overharvest for the terrarium trade; a year after its discovery in 2012, specimens were reported to be exported to Europe and the USA in significant numbers, which was greatly reducing wild populations. One expert has observed considerable population declines since 2014, and thousands are reported to be exported illegally. The species has not yet been assessed by IUCN, but is nationally assessed as near threatened (2016).

Many *Tylototriton* species are considered morphological complexes (e.g. *T. verrucosus*, *T. shanjing* and *T. asperrimus*) and morphological identification is considered difficult or even impossible by a non-specialist. In addition, there is great morphological variation between individuals of the same species. Species are frequently traded using an incorrect name, either erroneously or deliberately. It is also said to be difficult for non-specialists to differentiate between *Tylototriton* and *Echinotriton* (two species in the latter genus are also proposed for inclusion in Appendix II; see CoP18 Prop. 39). The genus is relatively understudied and future taxonomic work is likely to lead to more species being described.

Analysis: There is little information available on the wild populations of many species of *Tylototriton*, although they are generally believed to have small and probably declining ranges, and small population sizes. Habitat loss and degradation is a significant threat. Species are used for traditional medicine in Asia, for some species it is thought this could be in significant volumes, although no quantitative information is available. Regarding the pet trade, it was thought that the USA and European markets were the largest, although this may no longer be the case as imports into both are now restricted due to concerns over disease.

Some species have shown declines likely caused by overharvest (including but not limited to *T. lizhenchangi, T. vietnamensis* and *T. yangi*), and although it is not known what proportion were used domestically for traditional medicine (or pets) versus the international pet trade, significant numbers have been recorded in the latter. There appears to be a pattern of new species being described and then impacted by international trade, although some have likely been in trade previously but under a different name. Certain species already appear to meet the biological criteria for Appendix I (including but not limited to *T. lizhenchangi, T. vietnamensis, T. wenxianensis and T. yangi)* based on apparent marked declines, restricted ranges and/or small wild populations that are declining, and therefore meet the Appendix II 2a criterion.

For other species, there is insufficient information to determine whether the criteria are met. However, since morphological identification in this group is considered to be difficult or even impossible by a non-specialist, and the taxonomy is evolving, for ease of implementation a genus listing seems appropriate.

Other Considerations: As some species are nationally protected and thus trade is illegal, the range States concerned could put in place a voluntary zero-export quota for wild specimens which would be listed on the CITES website and empower re-exporting and importing Parties to assist with enforcing the law.

Due to the evolving taxonomy of *Tylototriton*, there is potential for the current CITES Standard Nomenclature Reference to become out of date. The CITES Standard Nomenclature Reference used for the newt species currently listed in CITES is Frost (2015). CoP18 Doc. 99 recommends a change to a 2017 version of Frost, but this will already be out of date (*T. ngarsuensis* was described in 2018). There is some debate regarding whether *Tylototriton* should be split into multiple genera (*Tylototriton, Liangshantriton* and *Yaotriton*), so further taxonomic revisions are potentially significant.

It is said to be difficult to differentiate *Tylototriton* and *Echinotriton.* Two species in the latter genus are also subject to a listing proposal (Prop. 39). If one proposal is accepted, then it seems appropriate for the other to also be accepted on the basis of 2bA (look-alike).

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

The high rate of species discoveries within the genus has increased the number of known species from 8 to 24/25 since 2010, *although the taxonomic status of some species is still under debate (Bernardes* et al.*, 2017a). Phylogenetic analyses divided the genus into the subgenera Yaotriton and Tylototriton (Bernardes* et al.*, 2017b). Taxonomic inconsistencies indicate that deep phylogenetic investigations in the Tylotrotriton group are required to resolve species relationships and uncover any cryptic diversity within this group; some consider Tylototriton to be three genera (based on morphology): Tylototriton sensu stricto*, *Yaotriton* and *Liangshantriton (Wang* et al.*, 2018).*

The CITES Standard Nomenclature Reference for the newts currently listed in the CITES Appendices is the Taxonomic Checklist of Amphibian Species extracted from Frost (2015). Some species in the genus Tylototriton have been described since the checklist was extracted (see table below) and further taxonomic revisions may lead to additional species being described.

Range and IUCN Global Category

In addition to the global IUCN Red List assessments, the threat status on the China Red List is also provided where available (Jiang et al.*,2016).*

Species and year of description	***IUCN Global Category***	***Threat status in China*** *(from Jiang* et al.*, 2016)*	***Range***
T. hainanensis (1984)	*Endangered (2008) B1ab(iii)+2ab(iii) ver 3.1*	*endangered*	*China*
T. vietnamensis (2005)	*Endangered (2016) B1ab(iii) ver 3.1*	*-*	*Viet Nam*
T. kweichowensis (1932)	*Vulnerable (2004 needs updating) B2ab(iii,v) ver 3.1*	*vulnerable*	*China*
T. notialis (2010)	*Vulnerable (2014) B1ab(iii) ver 3.1*	*-*	*Lao PDR, Viet Nam*
T. shanorum (2014)	*Vulnerable (2016) B1ab(iii) ver 3.1*	*-*	*Myanmar*
T. wenxianensis (1984)	*Vulnerable (2004 needs updating) B2ab(iii) ver 3.1*	*vulnerable*	*China*
T. ziegleri (2013)	*Vulnerable (2015) B1ab(iii) ver 3.1*	*-*	*Viet Nam*
T. asperrimus (1930)	*Near Threatened (2008) ver 3.1*	*near threatened*	*China, Viet Nam*
T. taliangensis (as Liangshantriton taliangensis)(1950)	*Near Threatened (2004 needs updating) ver 3.1*	*vulnerable*	*China*
T. shanjing (1995)	*Near Threatened (2004 needs updating) ver 3.1*	*near threatened*	*China*
T. anguliceps (2015)	*Least Concern (2016) ver 3.1*	*-*	*Lao PDR, Thailand, Viet Nam*
T. podichthys (2015)	*Least Concern (2016) ver 3.1*	*-*	*Lao PDR*
T. verrucosus (2012)	*Least Concern (2004 needs updating) ver 3.1*	*near threatened*	*China, India, Myanmar, Nepal, Thailand*
T. anhuiensis (2017)	*Not assessed*	*-*	*China*
T. broadoridgus (2012)	*Not assessed*	*near threatened*	*China*
T. dabienicus (2010)	*Not assessed*	*endangered*	*China*
T. himalayanus (2015)	*Not assessed*	*-*	*Nepal, India*
T. liuyangensis (2014)	*Not assessed*	*data deficient*	*China*
T. lizhenchangi (2012)	*Not assessed*	*vulnerable*	*China*
T. ngarsuensis (2018)	*Not assessed*	*-*	*Myanmar*
T. panhai (2013)	*Not assessed*	*-*	*Lao PDR, Thailand*
T. pulcherrimus (2012)	*Not assessed*	*near threatened*	*China*
T. pseudoverrucosus (2012)	*Not assessed*	*near threatened*	*China*
T. uyenoi (2013)	*Not assessed*	*-*	*Thailand*
T. yangi (2012)	*Not assessed*	*near threatened*	*China*

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

The genus *Tylototriton* mainly comprises small-ranged species that consist of few and small populations. Twenty species are endemic to one country (see Range). Declines have been observed in the extent and quality of habitats, as well as in the number of individuals. Species are harvested from the wild as a food source, for use in traditional medicine and to supply the international pet trade. *As current populations appear to be extremely small, they can no longer sustain the high levels of harvest that occurred in the past (M. van Schingen, in litt., 2019). The trade in newts within Southeast Asia appears negligible in comparison to the international trade (Rowley* et al.*, 2016). Wild-caught animals of species of Tylototriton have been very regularly available in the European pet trade over the past 20 years (although currently far less given legislation to restrict the spread of disease (BSal) (F. Pasmans, in litt., 2019).*

The usually predictable high concentration of individuals in small breeding sites during the reproductive season makes most *Tylototriton* species especially vulnerable to over-harvesting at known localities. At least 12 species are recorded at species level in the international trade and are mainly exported to the European, USA, and Japanese markets, even though many species are protected in their native countries.

Reported imports of live Tylototriton into the USA between 2007-2013 totalled 9,701 individuals: 52% of which were reported as T. verrucosus*, 21%* T. kweichowensis *and 18%* Tylototriton spp*. according to US Fish and Wildlife Service's LEMIS database.*

Reported imports of live Tylototriton into the EU between 2009 – 2017 totalled 1,555 (55% T. kweichowensis*, 36%* T. asperrimus*) according to the CITES Trade Database.*

Several species known to be threatened and/or in international trade are discussed further below:

Tylototriton asperrimus: *Native to southern China and northern Viet Nam (although populations in Viet Nam may actually be* T. ziegleri *or* T. vietnamensis *(Sparreboom, 2014)), this species was assessed as Near Threatened in 2008 because it is in significant decline (formally a common species) due to over-harvest and is suffering from habitat loss and degradation, making the species close to qualifying for Vulnerable. The major threat to this species in China is harvest for its use in traditional medicine (it is a substitute for* Gekko gecko, *which is widely used in medicine) (van Dijk* et al.*, 2008).* It is also said to be collected in large numbers during the breeding season and sold for the pet trade via Hong Kong SAR. *Habitat loss and degradation, resulting from smallholder agriculture and subsistence wood collecting, were also reported as a threat to this species (van Dijk* et al.*, 2008).* Threat levels are expected to be underestimated, and *T. asperrimus* is now believed to encompass other related species with significantly smaller ranges and fewer populations. Population assessments revealed generally small population sizes in investigated species: *T. asperrimus* has been found in 18 of 103 plots with a total of 68 individuals in Houhe National Nature Reserve, China. The SS details offers for sale in Europe; it is said to be one of the most commonly advertised species. *According to the CITES Trade Database, imports of 367 live animals were reported as directly imported into the EU (mainly Germany), predominantly from China and Hong Kong SAR, since the genus was listed in the EU Wildlife Trade Regulations in 2009 (120 were wild and the source for the remainder was not reported); an additional 200 were re-exported from Hong Kong SAR to Spain from China.* There are reports of successful breeding of *T. asperrimus*.

Tylototriton hainanensis: Endemic to Hainan Island in South China. *Assessed as Endangered in 2008 because its Extent of Occurrence was less than 5,000 km^2 and its Area of Occupancy was probably less than 500 km^2, with all individuals in fewer than five locations, and the extent of its habitat was probably declining. The species was reported to be uncommon and the population decreasing (Haitao & Chan, 2008). In the Jianfengling National Nature Reserve, its population density was reported as 31 per km^2 (Wang* et al.*, 2008), which one expert considered was probably the largest population in existence, facilitated by an environment with a high canopy density and without major anthropogenic disturbances (only eco-tourism) (Hernandez, 2016). Imports into the USA between 2007 and 2014 were limited to five scientific specimens from China (US Fish and Wildlife Service's LEMIS database). No imports have been reported to the EU since the genus was listed in the EU Wildlife Trade Regulations in 2009 (CITES Trade Database). Not observed for sale online or in physical markets in Japan in 2017 (Wakao* et al.*, 2018).* The SS notes international demand for the species (post on a forum in Canada).

Tylototriton kweichowensis: Endemic to China. *Assessed as Vulnerable in 2004 because its Area of Occupancy was less than 2,000 km^2, its distribution was severely fragmented, and there was a continuing decline in the extent and quality of its habitat, and in the number of mature individuals. There is little information on the population of this species; in 2004 it was reported to be probably common in its known localities but believed to be in decline (Yang* et al.*, 2004). Although its range overlaps with a few small protected areas, habitat destruction and degradation caused by industry (brick factories) and mining are major threats to this species; it is also collected for use in Traditional Chinese Medicine (TCM). A small number of individuals are also traded in the international pet markets (Yang* et al.*, 2004). Tylototriton kweichowensis* was imported in large quantities into Europe between 1990 and 1995, but only a few keepers have managed to keep the species*. It is bred in captivity in both China and Europe (Datong,* et al.*, 2004). Imports of 858 live* T. kweichowensis *were reported into the EU (Germany) from China and Hong Kong SAR since the genus was listed in the EU Wildlife Trade Regulations in 2009 (source of specimens not specified) according to the CITES Trade Database. Imports into the USA between 2007 and 2013 included 1,134 wild* T. kweichowensis *from Hong Kong SAR and 895 captive-bred/born (source codes C & F, from Hong Kong SAR, Germany, Canada) (US Fish and Wildlife Service's LEMIS database).* The SS includes online adverts for the species in a number of countries (e.g. Germany (EUR 70 (*ca. USD 79*)) and demand was expressed on online fora in Europe and the USA.

Tylototriton lizhenchangi (or T. lizhengchangi): Endemic to Hunan Province and adjacent areas. *The species is used in Traditional Chinese Medicine and thousands are said to be exported illegally for the pet trade under the name* T. asperrimus *(Sparreboom, 2014). Since its recent description (2012) the number of specimens illegally imported into Europe and the USA (via Hong Kong SAR) has risen. Its range is very small, and intensive collecting following its formal description as a separate species has reduced the wild populations to a point where it could become extinct in the very near future; large adult individuals were already hard to find in 2014 and 2015 (Hernandez, 2016). The species has not yet been assessed on the IUCN Red List, although experts indicate it should be considered Endangered (Hernandez, 2016).* The SS notes this species to be one of the most commonly advertised in the genus; it was found for sale in Germany and demand was reported in France (post on Facebook). *No imports into the EU were reported in the CITES Trade Database or into the USA in US Fish and Wildlife Service's LEMIS database.*

Tylototriton shanjing: Endemic to Yunnan Province, *this species was assessed as Near Threatened in 2004 because it was in significant decline (Ohler, 2004), but some populations were more recently reported to face an immediate risk of becoming extinct (Hernandez, 2016). While reported to be common in the past, populations of* T. shanjing *have shrunk considerably in recent years due to deforestation and habitat destruction, carp farming, mass exports of specimens for the terrarium trade (1990 - 2000), and overexploitation for TCM and food (Hernandez, 2016). Commonly seen in medicinal markets, for instance in Kunming and Chengdu, not many TCM dealers sell* T. shanjing*, but each dealer usually has a few kilograms stockpiled (mainly wild) (H. Chou, in litt., 2019). One expert reported the confiscation of hundreds of dried* T. shanjing *destined for the traditional medicinal trade within China (K. Wang, in litt., 2019). Three specimens were observed for sale in a pet shop in Japan in 2017; one reportedly captive-bred specimen was priced at 7,800 JPY (ca. 70 USD) (observations from Wakao* et al.*, 2017). Imports into the USA between 2007 and 2013 included 264 wild live individuals imported from Hong Kong SAR, in addition to 216 captive-bred individuals (US Fish and Wildlife Service's LEMIS database). Imports of 20 live individuals (source not specified) have been reported into the EU (Germany) from China since the genus was listed in the EU Wildlife Trade Regulations in 2009, according to the CITES Trade Database.*

Tylototriton shanorum: Endemic to Myanmar. Currently known only from two populations, and formally described in 2014, although s*pecimens of* T. verrucosus *that were imported in large numbers into Europe for medical research from the 1960's -1980's correspond to the taxon* T. shanorum *(and others), thus, these newts were commonly found in the terrarium trade under the name* T. verrucosus *(Hernandez, 2016). The species was assessed in 2016 as Vulnerable due to a low number of threat-defined locations (which take into account that the species probably exists beyond its known limit) its estimated extent of occurrence (EOO) is only 11,058 km^2, and there is a decline in the extent and quality of parts its habitat (IUCN SSC Amphibian Specialist Group, 2017). Forest loss as well as potentially high rates of collection for the pet trade, are very likely causing population declines in this species (IUCN SSC Amphibian Specialist Group, 2017). International trade does occur (e.g. two captive-bred juveniles sold in a pet-shop in Japan (Nishikawa* et al.*, 2014), the SS details offers for sale in Europe (EUR 30-100 (USD 34 – 115) per adult) but species-specific data is limited. Given the historic confusion of this species with* T. verrucosus*, it is likely that additional individuals are also traded under the misnomer* T. verrucosus*. (IUCN SSC Amphibian Specialist Group, 2017).*

Tylototriton verrucosus: Considered a morphological complex which was previously considered to be widely distributed but now known to enclose multiple related species with significantly smaller ranges and fewer populations. *The species was assessed as Least Concern in 2004 due to its wide distribution, tolerance of a broad range of habitats, presumed large population (although it was reported to be an uncommon species through most of its range), and because it was considered unlikely to be declining sufficiently to qualify for a more threatened category (van Dijk* et al.*, 2004), but this could change drastically if this taxon were determined to actually comprise several distinct taxa (Hernandez, 2016). The status of individual populations differs geographically, with some being highly threatened by anthropogenic activities such as agriculture, as is currently the situation in China (Hernandez, 2016).* Threats include habitat loss and degradation, human-induced forest fires and water pollution, and in parts of its range it is used as bait for fishing and in traditional medicine, as well for as the international and domestic pet trade *(van Dijk* et al.*, 2004).* One of the species exported in the largest quantities to Europe in the 1960s for medical research (although only four species were known in the genus at that time). One of the most commonly advertised species (examples of offers for sale in China, Europe and the USA are provided in the SS). *Imports into the USA between 2007 and 2013 included 5,031 wild live individuals mostly from China and Hong Kong SAR (US Fish and Wildlife Service's LEMIS database). Imports of 400 live individuals (source unspecified) have been reported into the EU (Germany) from China since the genus was listed in the EU Wildlife Trade Regulations in 2009 according to the CITES Trade Database. In 2016 a gang in China selling multiple species online were intercepted by police, and 32 T. verrucosus seized (CWCA, 2016).*

Tylototriton vietnamensis: *Assessed as Endangered in 2016 as this species had only two threat-defined locations in northern Viet Nam, its estimated extent of occurrence (EOO) was only 1,345 km^2, and there was a*

decline in the extent and quality of parts its habitat. The species was reported to be known only from Bac Giang and Quang Ninh Provinces, northeastern Viet Nam. Although Tylototriton populations elsewhere have been assigned to this species, subsequent phylogenetic investigation has shown that of all reported localities, only those around its type-locality are in fact assignable to this species. The population was reported to be decreasing, although little appears to be known about the population size and trends except that it was only recorded from three surveys (IUCN SSC Amphibian Specialist Group, 2016). It appeared to be an uncommon species (Hernandez, 2016). Tay Yen Tu Nature Reserve and Yen Tu Landscape Protection Area were reported to harbour the largest population of this species, and a survey of all known breeding sites in the Reserve in 2010 found 216 individuals (Bernardes et al.*, 2013). Unsustainable harvest to supply demand is very likely a threat to this species (IUCN SSC Amphibian Specialist Group, 2016). The small fragmented populations are affected by intensive deforestation, climate change, erratic rainfalls and improved access by humans to ponds (Hernandez, 2016).* In Tay Yen Tu Nature Reserve, northern Viet Nam, researchers found no evidence of human disturbance within the vicinity of *T. vietnamensis in 2012, but two years later cattle were using the ponds and trails that had been created, and local people were collecting newts for private medicinal use, selling them at Chinese markets for the international pet trade, or selling to local tourists (Bernardes* et al.*, 2017a).* One trader from Hanoi referred to Tay Yen Tu Nature Reserve as the origin of his newts. Animals are only available in stock from May to August, during breeding season. A shop in southern Viet Nam offered *T. vietnamensis* incorrectly labelled as different species (and were possibly being exported to Thailand). One of the most commonly advertised species; SS provides examples of offers for sale in Viet Nam (wild, USD 15-25) and demand in France (social media offers). *Imports of 10 live* T. vietnamensis *(source unspecified) have been reported into the EU (Germany) from Viet Nam since the genus was listed in the EU Wildlife Trade Regulations in 2009 according to the CITES Trade Database. Imports into the USA between 2007 and 2014 were limited to two scientific specimens from Viet Nam (US Fish and Wildlife Service's LEMIS Database). Given the historic confusion of this species with* T. asperrimus *and others in the genus it is likely that additional individuals are also traded under incorrect names (IUCN SSC Amphibian Specialist Group, 2016).* There have been successful reports of captive breeding, *although the species appears to be sensitive to disease and not easy to keep (Sparreboom, 2014).*

Tylototriton wenxianensis: This species is endemic to China. *Assessed in 2004 as Vulnerable because it's area of occupancy was less than 2,000 km^2, it was known from only four locations, and there was continuing decline in the extent and quality of its habitat although this assessment needs updating (Liang & Changyuan, 2004). Density was estimated at 16 newts/km^2 giving a global population of 30,000 in 2008, but the population was reported to have more than halved by 2015 with threats including habitat loss and degradation due to tea plantations and associated firewood collection (Hernandez, 2016).* There are reportedly undescribed taxa within *T. wenxianensis, meaning the population may be smaller should the species be split.* Said to be one of the most commonly advertised species, the SS details offers for sale in Europe (EUR 45 (ca. USD 59)). Successful captive breeding has been reported.

Tylototriton yangi: Endemic to South Yunnan Province in China with an estimated extent of occurrence of <20,000 km^2*, this species has not been assessed on the IUCN Red List but is said to be highly threatened by excessive collecting for the terrarium trade. Researchers have observed a rapid decline in numbers and consider the species endangered (Sparreboom, 2014). A year after its discovery in 2012, specimens were exported to Europe and the USA in significant numbers, which was reported to be reducing the wild populations to a major extent (Hernandez, 2016). One expert has noticed considerable population declines since 2014 (K. Wang, in litt., 2019). Thousands were reported to be exported illegally (Sparreboom, 2014). Tylototriton yangi* is said to be collected in large numbers during the breeding season and sold for the pet trade via Hong Kong SAR: o*ne expert witnessed a single poaching incident of 500 individual* T. yangi*, whereby the newts were smuggled to Hong Kong SAR and ended up in the USA pet market one month later (K. Wang, in litt., 2019).* Said to be one of the most commonly advertised species. Wild specimens for sale in Europe and the USA have been reported; the SS details offers for sale in Europe (up to EUR 100 (ca. USD 112)). A shop in southern Viet Nam offered *T. verrucosus* incorrectly labelled as *T. yangi* as well as several other Chinese species (and were possibly being exported to Thailand). *Other threats include habitat destruction due to mining and pollution with agrochemicals, and the species has reportedly been collected for traditional medicine purposes for some time (Hernandez, 2016).*

Tylototriton ziegleri: *Described in 2013 and known from one location in northern Viet Nam (although these are unlikely to represent the actual limits of the species' range). Assessed as Vulnerable in 2015 due to an extent of occurrence of only 16,218 km^2, it is known from a single location and there is ongoing decline in the quality of its habitat. The species' population was reported to be likely decreasing due to disturbance at breeding habitats and potential collection for traditional medicine and the pet trade (IUCN SSC Amphibian Specialist Group, 2017).* Several field surveys of *T. ziegleri* found only small numbers of adults per breeding site during the breeding season. *One expert noted that at sites near the Chinese border they had already found depleted breeding sites, and local people had observed visits from foreign poachers* (the SS notes from China)*, who easily persuade*

locals to help them collect newts in return for a very low payment (M. Bernardes, in litt., 2019). Thirty wild specimens were exported from the USA to Viet Nam via Canada for scientific purposes between 2007 and 2014 (US Fish and Wildlife Service's LEMIS database). The SS details offers for sale in Viet Nam (USD 15-25) and demand in France.

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

Paramesotriton hongkongensis *was listed in Appendix II in 2017. The listing proposal for* P. hongkongensis *mentions all genera in the family Salamandridae* *except* *Tylototriton (Pachytriton, Paramesotriton, Cynops and Hypselotriton) as being similar to* P. hongkongensis *(CoP17 Prop. 41, 2016).*

Additional Information

Threats

Main threats to *Tylototriton* are habitat loss and degradation especially around the breeding habitats, as these species show philopatry and limited mobility. Another leading threat to *Tylototriton* is over-harvesting for the international pet trade, food, traditional medicine, and use as baits for fishing. *Newts are used for traditional medicine and pets within Viet Nam, Myanmar and Lao PDR. Trade in newts within Southeast Asia appears negligible in comparison to the international trade (Rowley* et al.*, 2016). As specimens accumulate in fresh water habitats during the breeding season, they can be captured with nets and as such entire cohorts may be easily collected (M. van Schingen., in litt., 2019).*

Sexual maturation is reached between 3 and 5 years. *Generation time is long (between 4-8 years to mature) (F. Pasmans, in lit., 2019).*

Conservation, management and legislation

The genus has been listed in Annex D of the EU Wildlife Trade Regulations *since 2009.* Detailed information on legal protection is provided in the SS. In summary some species are protected within range States, some appear not to be, and for others the status of protection is unclear.

Species in the genera native to mainland China but not Hong Kong SAR are not protected in Hong Kong SAR, therefore it appears the newts are being smuggled from mainland China into Hong Kong SAR where they can then be exported legally to destination countries (K. Wang, in litt., 2019).

EU Decision 2018/320 (came into effect in 2018) requires that all members of the order Caudata that are imported into the EU must be held in quarantine post-importation in an appropriate establishment, and that when being moved between EU countries they must be held in an appropriate establishment under the supervision of the competent authority, immediately prior to their exportation. The new rules seek to stop the spread of the fungal pathogen, Batrachochytrium salamandrivorans (Bsal) which can be spread to native species and cause morbidity and mortality (CEFAS, 2018).

In 2016 the USA listed 201 salamander species as "injurious wildlife" under the Lacey Act due to concerns over importing Bsal, including all species in the genus Tylototriton. This means importation into the USA is prohibited (there are some exceptions e.g. scientific research, zoos) (USFWS, 2016).

Artificial Propagation/captive breeding

There are mixed reports on the success and extent of captive breeding. Captive breeding has occurred in these species for many years (e.g. captive breeding in T. shanjing *reported from 1986) and it is likely that some of the trade is in captive specimens (M. van Schingen, in litt., 2019). The SS also notes incidences of captive breeding. However, one expert noted that as Chinese salamanders are easily available as wild-caught specimens, often at a relatively low price, this renders captive breeding to many people unattractive, and while certainly feasible, captive breeding of many of these species requires dedication, expert care and patience (most species require at least 3-4 years before reaching maturity in captivity) (Sparreboom, in litt to Shenyang Normal University, 2015). These are species that in general can be kept and bred in captivity without too many difficulties, but due to inappropriate husbandry, the majority of those traded will probably die quickly (F. Pasmans, in litt., 2019). New rules on importation into the EU and USA may increase the need for captive breeding within these markets to meet demand.*

Implementation challenges (including similar species)

Most *Tylototriton* species look alike and are considered morphological complexes, like *T. verrucosus, T. shanjing* and *T. asperrimus* masking together at least nine known other species. *Morphological identification in this group*

is difficult or even impossible by a non-specialist, and several of the newly described Tylototriton species were previously confused with T. verrucosus *and are likely to have been (and continue to be) traded under this name, erroneously or deliberately (Rowley* et al.*, 2016). Tylototriton spp. are difficult (not to say impossible in most cases) to identify without application of genetic tools. Thus, confusion of very rare, endemic species with more widely occurring taxa is very common. Genetic identification is necessary in most if not in all cases, as morphological variation is still poorly understood and not all cryptic taxa discovered (T. Ziegler, in litt., 2019).* To further complicate the identification of *Tylototriton* species, there is also great morphological variation within individuals of the same species. *Morphological identification of many species of the genera Tylototriton (and distinguishing species in this genus from Paramesotriton and Pachytriton) is very difficult and needs specialist intervention (Sparreboom, in litt to Shenyang Normal University, 2015). According to one expert it is possible to differentiate species of Tylototriton from Paramesotriton (F. Pasmans, in lit., 2019).* Phylogenetically, *Tylototriton* spp. forms a clade with the genera *Pleurodeles* and *Echinotriton* sharing also morphological similarities with the latter genus *(of which two species are also subject to a listing proposal at COP18),* from which it was separated in 1982. *It is difficult for non-specialists to differentiate between Tylototriton and Echinotriton (M. van Schingen., in litt., 2019).*

*In the past, several shipments were imported into the EU and the USA under wrong names (e.g. "*Tylototriton verrucosus*,* T asperrimus*,* Paramesotriton chinensis*"), where the shipments actually contained a large variety of species, including rare and threatened ones (F. Pasmans, in litt., 2019). Tylototriton specimens were mainly traded under the names* T. verrucosus*,* T. asperrimus *or* T. shanjing*, or just as "Chinese Newts" / "Birma-Krokodilmolch" as most species morphologically resemble each other.*

Potential risk(s) of a listing

If this proposal (and proposals to include other newts (Echinotriton and Paramesotriton) in Appendix II) is accepted, it may increase trade pressures on unlisted species such as Laotriton laoensis *(which used to be placed within the Paramesotriton genus), however this is not reason enough not to list Tylototriton (J. Rowley, in litt., 2019).*

Given the difficulties in identification, a genus-level listing may encourage trade to be reported at the genus level as Tylototrition spp., which could mask trade in the most vulnerable species.

Other comments

Many species have proved to be exceptionally sensitive to being caught and confined to narrow enclosures, let alone being shipped long distances. This causes so much stress to the animals that it leads to diseases that mostly prove fatal. Packaging salamanders in large numbers often leads to diseases to which most animals in a shipment will succumb (Sparreboom, in litt. to Shenyang Normal University, 2015).

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Inclusion of Mako Sharks *Isurus oxyrinchus* and *Isurus paucus* in Appendix II

Proponents: Bangladesh, Benin, Bhutan, Brazil, Burkina Faso, Cabo Verde, Chad, Côte d'Ivoire, Dominican Republic, Egypt, European Union, Gabon, Gambia, Jordan, Lebanon, Liberia, Maldives, Mali, Mexico, Nepal, Niger, Nigeria, Palau, Samoa, Senegal, Sri Lanka, Sudan and Togo

Summary: Shortfin Mako Shark *Isurus oxyrinchus* is a fast, large (4 m), widely distributed, migratory shark with low biological productivity. It can be found in all temperate and tropical ocean waters from 50°N (60°N in the North Atlantic) to 50°S. It is distributed across the following oceans: North Atlantic (14.5% of distribution), South Atlantic (12%), North Pacific (32.5%), South Pacific (22%), Indian Ocean (17.9%) and Mediterranean (1.1%).

Longfin Mako Shark *I. paucus* appears in similar waters, although its complete distribution remains unclear. Very little is known about its biology.

The primary threats to *Isurus oxyrinchus* and presumably *I. paucus* are directed and incidental catch in multi-specific fisheries found throughout its range. *Isurus oxyrinchus* is generally retained for its high-valued meat for both national and international markets, whilst its fins are mostly destined for the international market. Its meat is consumed all over the world and is considered a premium product. Fins from *I. oxyrinchus* have been observed in markets in Hong Kong SAR's main commercial centre, where this was reported as the fourth and fifth most abundant species in 1999-2000 and 2014-2015 respectively. *Isurus paucus* fins have also been observed in this market. Other products from this trade include liver oil, skin and teeth. The form in which species are traded (primarily meat) makes it hard to differentiate between species. Although it is possible to visually differentiate the fins of the two species using macro-morphology based on differences in the dermal denticles, it is reported that *I. paucus* fins are often combined in *I. oxyrinchus* and thresher (*Alopias* spp.*)* fin categories, due to a similarity in appearance and market value.

Isurus oxyrinchus is also the target of sport fishing and at risk of being caught in shark protection nets. Climate change may also be a threat to *I. oxyrinchus;* warming ocean waters may affect its spatial and temporal distribution.

Both species are considered to have low productivity. Global population sizes are unknown but may number in the millions. Various studies and sources have used a range of indicators to examine the trends in each of the ocean areas including spawning stock fecundity, spawning abundance, biomass and mortality. However, due to different datasets and methods used for analysis, these studies are often not directly comparable and therefore a percentage decline is not always possible to calculate. Available information has been examined for evidence of historical and recent declines in relation to the quantitative guidelines contained in the footnote to Annex 5 of *Res. Conf. 9.24 (Rev. CoP17)* for commercially exploited aquatic species. We take these guidelines to refer to the criterion 2aA. Information has also been examined for evidence of decreasing populations considered in relation to criterion 2aB.

The FAO Expert Advisory Panel examined available datasets for robust information on the extent of marked declines for *I. oxyrinchus* (for which more data are available) to determine if there have been historical and recent declines near to the guideline figures in the footnote to Annex 5 of *Res. Conf. 9.24 (Rev. CoP17)*. The Panel concluded that in none of the species' areas of distribution were there historical declines near to the guideline figures, however they did note that there was reliable evidence of historical population decreases in the North Atlantic, Mediterranean and North Pacific (combined distribution of 48.1%). They considered there was not enough reliable evidence for the South Atlantic and Indian Ocean to calculate the extent of decline. The Panel considered that the population of the South Pacific has historically been stable and possibly increasing in recent years.

The Panel determined that recent decreases in the North Atlantic were of between 23-32%. Although they acknowledged decreases in the Mediterranean, they found that the extent of decline was not well determined. They found data to determine recent declines for the South Atlantic and Indian Ocean were not robust enough to calculate the extent of decline, but noted there were marginal increases (by 0.16% per year) in the North Pacific.

While there appear to be no historic or recent marked declines near to the guideline figures in the footnote to Annex 5, taking into consideration available datasets there is evidence that populations of *I. oxyrinchus* in the North Atlantic, South Atlantic, Mediterranean, Indian Ocean and North Pacific (making up 78% of the distribution) have all undergone historical decreases in population. In recent years the North Atlantic, Mediterranean and Indian Ocean (33.5% of distribution) populations have been decreasing. In the South Pacific there is general agreement that the population is likely increasing marginally, however in the North Pacific there is a lack of consensus over the trend, with some considering a possible continuing decline whereas others consider there to have been a marginal increase. There is a lack of data for the South Atlantic, but in the most recent IUCN Red List assessment it is accepted that the situation in the North Atlantic (decreasing population) is representative of the South Atlantic.

Less information is known on the population size of *Isurus paucus,* although it is considered the rarer of the two species; expert judgement suggests global declines would be similar to *I. oxyrinchus* as it is caught as target and incidental catch alongside *I. oxyrinchus* in offshore and high seas waters.

Recent global IUCN Red List assessments due to be published in March 2019 have categorised both species as Endangered.

Some range States have adopted a variety of legislative measures including quotas, finning bans, fishing gear restrictions, and area and season bans. Within the distribution of *I. oxyrinchus*, at least some areas are known to have stricter legislation in place, often in the form of recommendations or resolutions established by Regional Fisheries Management Organisations (RFMOs), including the banning of shark finning or the requirement for live *I. oxyrinchus* to be released.

Analysis: *Isurus oxyrinchus* and *I. paucus* are both widely distributed, occurring in temperate and tropical ocean waters. *Isurus oxyrinchus* meat is utilized both locally and internationally and considered to be high value. Its fins have been observed in some of the largest fin markets. The species' low productivity place them at risk of overexploitation if stocks are overfished and unable to recover. There is no robust evidence of historic or recent marked declines for *I. oxyrinchus* that would meet the guidelines for listing under Annex 2aA. However, historical population decreases have been reported for *I. oxyrinchus* across large parts of its range (78%). Recent data suggest populations are continuing to decrease in 33.5% of its distribution (North Atlantic, Indian Ocean and Mediterranean), and if the condition of the North Atlantic is representative of the South Atlantic, this would add a further 12% of the species' distribution. The populations in the South Pacific appear to be stable or marginally increasing and there are differing opinions on the trend in the North Pacific. When considering the historic and recent trends in populations in conjunction with one another, overall it would appear that regulation of trade in *I. oxyrinchus* is required to ensure that the harvest of specimens from the wild is not reducing the wild population to a level at which its survival might be threatened by continued harvesting. Therefore, *I. oxyrinchus* meets the criteria in Annex 2aB of *Res. Conf. 9.24 (Rev. CoP17)*.

Population trend data for *I. paucus* are limited but it is likely to be undergoing similar decreases to *I. oxyrinchus*, thus potentially also meeting the criteria in Annex 2aB of *Res. Conf. 9.24 (Rev. CoP17)*.

Some legislation and regulations are in place in some of the regions where population declines are occurring. The extent to which these are being implemented is unclear. Any CITES listing would reinforce the implementation of any existing legislation and management measures.

Fins of both species are sometimes mixed in the same market category and although it is possible to differentiate the fins due to differences in dermal denticles, *I. paucus* is commonly misidentified as *I. oxyrinchus*. Meat would be less readily identifiable to the species level and therefore enforcement officers who encounter specimens of CITES-listed species are unlikely to be able to distinguish between species. Thus, if either species is considered to meet the criteria in Annex 2a, then the other species should be included in the Appendices in line with Annex 2bA.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS

IUCN Global Category and Range

There are two species in the genus Isurus:

Shortfin Mako Isurus oxyrinchus

Global - Endangered A2bd (2018) ver 3.1. (Rigby et al.*, 2019a, accepted for publication in the March 2019 Red List update)*

Regional - Mediterranean: Critically Endangered A2bd (2016) ver 3.1.
Regional - European: Data Deficient (2015) ver 3.1.

It has a worldwide distribution and is highly migratory with 64% (of 2,459 marked sharks) recaptured in the North Atlantic captured 500 km away from the tagging site and some up to 4,542 km away. It can be found in all temperate and tropical ocean waters from 50°N (60°N in the North Atlantic) to 50°S. *It is occasionally found close inshore where the continental shelf is narrow (Cailliet* et al., *2009). It is not normally found in waters below 16°C (Compagno, 2001). Isurus oxyrinchus* is found in the following FAO Fishing Areas: 21, 27, 31, 34, 37, 41, 47, 51, 57, 61, 67, 71, 77, 81 and 87. Its country occurrence is 143 countries.

Longfin Mako Isurus paucus

Global - Endangered A2d (2018) ver 3.1. (Rigby et al.*, 2019b, accepted for publication in the March 2019 Red List update)*

Regional - Mediterranean: Data Deficient (2016) ver 3.1.
Regional - European: Data Deficient (2014) ver 3.1.

It appears to be a widely distributed species in tropical and warm temperate waters. However, at present records are sporadic and the complete distribution remains unclear. This is in part due to the confusion with the more common Isurus oxyrinchus. It is found in the following FAO Fishing Areas: 21, 27, 31, 34, 41, 47, 51, 57, 61,71, 77 and 81. Range countries/territories are Australia (New South Wales, Queensland), Brazil, Cuba, Ghana, Guinea-Bissau, Japan, Liberia, Madagascar, Mauritania, Micronesia, Federated States of, Morocco, Nauru, Portugal, Solomon Islands, Spain (Canary Is.), Taiwan Province of China, USA (Hawaiian Is., California, Florida) and Western Sahara (Reardon et al.*, 2006).*

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

Biology

Isurus oxyrinchus is a species of large shark (~ 4 m), that is fast-swimming, reaching 70 km/hour. It is highly migratory and distributed in temperate and tropical ocean waters (50˚ N to 50˚ S). Their seasonal movements depend on food availability, water temperature and stages of growth of the species, it can sometimes be found on the coast.

Isurus oxyrinchus has low biological productivity according to the criteria of *Res. Conf. 9.24 (Rev. CoP17)*, with a natural mortality lower than 0.2 (0.072 to 0.223), an intrinsic growth rate lower than 0.14 (0.031 to 0.123); a Von Bertalanffy growth constant less than 0.15 (0.05 to 0.266), an average age of maturity greater than 8 years (up to 21 years), a maximum age greater than 25 years (up to 45 years) and a longer generation time than 10 years (25 years). Additionally, this species produces between 4 and 25 pups per litter with a gestation period of 12 to 25 months and reproduces every two or three years.

Isurus spp. are ovoviviparous and oophagous, that is, live offspring are born and feed on infertile eggs during gestation. Like other lamnid sharks, *Isurus oxyrinchus* uses a circulatory heat exchange system to maintain the

temperature of its muscles and viscera above that of the surrounding seawater, which allows for a higher level of activity.

Isurus paucus has a litter size of 2-8 and a maximum size of 427cm (Castro et al., 1999; Compagno, 2001).

Population size
The global population size for *Isurus oxyrinchus* is unknown. Different assessments have considered independent stocks. *There are oceanic stock assessments available (North Atlantic, South Atlantic, North Pacific and Indian) that can provide estimates of biomass and/or spawning stock biomass (Barreto,* in litt.*, 2019). Information on* Isurus paucus *appears to be even more limited.*

Population trends
Globally
Isurus oxyrinchus
The latest IUCN Red List assessment (Rigby et al.*, 2019a, accepted for publication in the March 2019 Red List update) reports that steep population declines have occurred in the north and south Atlantic, with declines also evident, though not as steep in the north Pacific and Indian Oceans. The south Pacific population appears to be increasing but with fluctuating catch rates. The weighted global population trend estimated a median decline of 46.6%, with the highest probability of a 50-79% population reduction over three generation lengths (72-75 years), and therefore* Isurus oxyrinchus *was assessed as Endangered in the new Red List Assessment.* Previously (2004) it was assessed as being Vulnerable.

Isurus paucus
The latest IUCN Red List assessment (Rigby et al.*, 2019b, accepted for publication in the March 2019 Red List update) reports that limited population trend data indicates strong declines and it is suspected to have undergone a population reduction of 50-79% globally over the last three generations (75 years), similar to its congener, Isurus oxyrinchus. Isurus paucus is therefore assessed as Endangered in the new Red List Assessment. Previously (2006) it was assessed as being vulnerable (Reardon* et al.*, 2006).*

Various studies and sources have used a range of indicators to examine the trends in each of the ocean areas including spawning stock fecundity, spawning abundance, biomass and mortality. The results of these are summarised in Table 1, and discussed in more detail below.

Table 1. *Isurus oxyrinchus population trends for oceans, sourced from FAO Expert Advisory Panel report* (FAO, 2019), SS**, original source*** and accepted IUCN Red List assessments**** reporting the indicator used, time period and original source.*

Ocean (% of distribution)	Indicator	Extent of decline	Time period	Source
North Atlantic (14.5%)	Spawning stock fecundity	Historical: 50%* Recent (2006-2015): 32%*	1950-2015	ICCAT, 2017
	Spawning stock fecundity	Historical: 39%** Recent (2006-2015): 32%** Annual: 4%** Projected 10 year: 60% **	1950-2015	ICCAT, 2017
	Biomass	Declining****	1950-2017	ICCAT, 2017
	CPUE	43%***	1986-2005	Cortes et al., 2007
	CPUE	34%***	1992-2005	Baum and Blanchard, 2010
South Atlantic (12%)	Spawning stock fecundity	Uncertain***	1950-2015	ICCAT, 2017
	Biomass	IUCN RLA considers decrease in N. Atlantic population trends representative of S. Atlantic****		ICCAT, 2017
	CPUE	99% declines between 1979-1997 and 1998-2008	1979-2011	Barreto et al., 2016

		*17% increase between 1998-2009 and 2008-2011****		
North Pacific (32.5%)	*Spawning abundance*	*Historical: depletion to 58% (CI:30~86%) of unfished or 42% decline** *Recent (2007-2016): Increasing by 0.16% per year**	*1975-2016*	*ISC, 2018*
	Spawning abundance	*Historical: 16.4%*** *Recent (2006-2016): Increase of 1.8%*** *Annual: Increase of 0.18%***	*1975-2016*	*ISC, 2018*
	Spawning abundance	*Declining*****	*1975-2016*	*ISC, 2018*
	Spawning potential ratio (SPR)	*SPR in 2003=20% which is less than Biological Reference Point SPR=35%****	*1990-2003*	*Chang and Liu, 2009*
	Fishing mortality	*Fishing mortality in 2003=0.066/year>Biological Reference Point=0.045/year****	*1990-2003*	*Chang and Liu, 2009*
	Abundance	*Infer decreasing trend****	*1995-2005*	*Tsai et al., 2011*
	Abundance	*Infer decreasing trend****	*1995-2010*	*Tsai et al., 2014*
	CPUE	*69% over time period or 7% per year****	*1996-2009*	*Clarke et al., 2013*
	CPUE	*Stable population****	*2000-2010*	*Rice et al., 2015*
	CPUE	*Increasing trend****	*2006-2014*	*Kai et al., 2017*
South Pacific (22%)	*CPUE*	*Trends not significant****	*1996-2009*	*Clarke et al., 2013*
	CPUE	*Possibly declining between 2009-2013 but trend unreliable****	*2009-2013*	*Rice et al., 2015*
	CPUE	*Three datasets having a "nil" trend and one having an increasing trend****	*1993-2013*	*Francis et al., 2014*
	CPUE	*Increasing trend*****	*1995-2013*	*Francis et al., 2014*
Indian Ocean (17.9%)	*Biomass*	*Historical: 26%*** *Recent (2005-2015): 18.8%*** *Annual: 2.1%*** *Projected 10 year: 41.6%***	*1970-2015*	*Brunel et al., 2018*
	Biomass	*Declining*****	*1971-2015*	*Brunel et al., 2018*
	CPUE and mean weight	*Declining abundance****	*1964-1988*	*Romanov et al., 2008*
	CPUE	*Decline from 1994-2005 and subsequent increase until 2010****	*1994-2010*	*Kimoto et al., 2011*

	CPUE	*High variability until 2008, followed by increasing trend until 2016****	*2000-2016*	*Coelho et al., 2017*
Mediterranean (1.1%)	*Different indices of shark abundance*	*Historical: declines to over 90%****	*Multiple time periods*	*Ferretti et al., 2008*

North Atlantic (14.5% of the total distribution of the species)
The ICCAT (International Commission for the Conservation of Atlantic Tunas) carried out the most recent evaluation of the North Atlantic stock in 2017, with data from 1950 to 2015 and used four models (Bayesian Surplus Production Model, Just Another Bayesian Biomass Assessment model, Catch only Monte-Carlo and Stock Synthesis 3). The combined probability of all models being in a state of overexploitation while still experiencing overfishing is 90%.

Based on the spawning stock fecundity (SSF) generated by the Stock Synthesis 3 model (table 7 of ICCAT, 2017) the SS estimated that there is a historical decrease of 39% (1,126,000 average SSF of 1950-1960 against 686,600 average SSF of 2006-2015), a recent decrease of 32.1% (822,000 SSF in 2006 and 558,000 SSF in 2015) with an annual decrease rate of 4.2%. Using the 4.2% decrease rate and starting at 558,000 SSF, an estimated projected 10-year decrease of 60% of the historical baseline was reported (1,126,000 average SSF of 1950-1960 versus 443,758 average SSF of 2016-2025).

In the latest IUCN Red List assessment (accepted for publication in the March 2019 Red List update) the trend analysis of the north Atlantic modelled biomass for 1950-2017 (68 years) from ICCAT, 2017 revealed annual rates of decline of 1.2%, consistent with an estimated mean decline of 60% over three generation lengths.

Satellite telemetry was used as a tool to document fishing interactions and quantify the fishing mortality of *Isurus oxyrinchus* in the northern Atlantic Ocean. During 2013 to 2016 in the Yucatan peninsula, Mexico and in Maryland, the USA, a total of 40 sharks were tracked. The MARK model was used to estimate the survival probability of sharks annually, which is formulated as a generalized linear model (GLM), which allows modeling survival based on variables at the individual level (e.g. size, age, sex). The outcome of the modelling estimated fishing mortality between 0.19-0.56, 5-18 times higher than the estimated mortality in the maximum sustainable yield ranging from 0.031-0.038. *This implies the North Atlantic stock of* Isurus oxyrinchus *is currently experiencing overfishing (i.e. F > Fmsy). Thus, if the level of fishing mortality observed is representative of the western North Atlantic population, it is likely to be unsustainable (Byrne et al., 2017).*

A 43% decline in relative abundance has been observed for Isurus *spp. between 1986 and 2005 from a logbook data analysis (Cortes* et al.*, 2007). This decline was largely driven by a 21% decline in the first 3 years of the series (1986-1988), followed by an increase in 1989, and a progressive decline from 1989 to 1999, after which the series progressively started increasing until 2005 (Cortes* et al.*, 2007).*

Isurus spp. standardized catch rates were estimated to have declined, although the trend (instantaneous rate = −0.032) equating to a 34% decline (95%CI: 1–56%) between 1992 and 2005, was only marginally significant and imprecisely estimated. The estimated decline for Isurus oxyrinchus, which accounted for 79% of all recorded Isurus spp., was slightly greater (instantaneous rate = −0.040, 95% CI: −0.005 to −0.074, p = 0.026) (Baum & Blanchard, 2010).

South Atlantic (12% of the total distribution of the species)
The ICCAT (2017) carried out the most recent assessment of the South Atlantic stock, with data from 1950 to 2015 and using three models (Bayesian Surplus Production Model, Just Another Bayesian Biomass Assessment model and Catch only Monte-Carlo). The combined results of the model indicate a 19% probability that the stock is overfished and that it experiences overfishing. The evaluation group of ICCAT SCRS considered that the results of the state of the stock for the South Atlantic are highly uncertain. They concluded that, despite this uncertainty, it is likely that in recent years the number of females in the stock may be below the expected level of maximum sustainable yield and that the fishing mortality already exceeds the expected mortality in the maximum sustainable yield.

Due to unreliable stocks assessments, the latest IUCN Red List assessment (accepted for publication in the March 2019 Red List update) report that the results from the north Atlantic would be representative of the south Atlantic for the trend analysis.

Barreto et al.*, 2016 identified three temporal phases of exploitation of* Isurus oxyrinchus *on longlines in the South Atlantic. Using generalised linear mixed models to standardise catch rates and identify trends in each of the three*

phases (1979-1997, 1998-2008 and 2008-2011). Steep declines of 99% in the average CPUE were observed between 1979-1997 and 1998-2008, with a 16.97% increase between 1998-2008 and 2008-2011. They observed extremely low catch rates in 1998-2008 and 2008-2011, suggesting that the population of Isurus oxyrinchus *is depleted in the South Atlantic.*

North Pacific (32.5% of the total distribution of the species)
The ISC SWG (International Scientific Committee Shark Working Group) evaluated the stock of *Isurus oxyrinchus* with the best scientific information available to date, with data from the North Pacific, provided by the USA, Japan, Taiwan Province of China and Mexico of catches (1975 to 2016) that were standardized. They used the Stock Synthesis 3 from which a base model was generated, and a sensitivity analysis was carried out to determine the possible weaknesses of this modelling. Based on this, six scenarios were also modeled that address the identified weaknesses (e.g. increasing catch data for the period with greater uncertainties by 1975-1993). As a result of the base model (which is consistent with the different modelled scenarios), it was determined that there is a more than 50% probability that the stock is not in overfishing conditions (the current spawning abundance, 910,000 sharks is 36% higher than the expected spawning abundance in the maximum sustainable yield 633,700 sharks) and overexploitation does not exist (the impact of current fishing, 0.16, is less than the expected impact on maximum sustainable yield, 0.26). The predictive power of the model for the future is limited and uncertain. However, three scenarios were run projecting the behaviour of the stock to 10 years with which it is estimated that if the average catches of 2013-2015 are maintained or reduced by 20%, the abundance of females may increase. Based on the spawning abundance generated by the model (Table 7 of the publication ISC, 2018) the SS estimated that there is a historical decrease of 16.4% (1,024,000 average spawning abundance from 1975-1985 against 855,700 spawning abundance average of 2006-2016), a recent increase of 1.8% with an annual increase rate of 0.18% (844,800 spawning abundance in 2006 and 860,200 spawning abundance in 2016).

In the latest IUCN Red List assessment (accepted for publication in the March 2019 Red List update) the trend analysis of the modelled spawning abundance for 1975-2016 (42 years) from ISC, 2018 revealed annual rates of the decline of 0.6%, consistent with a median decline of 36.5% over three generation lengths.

Until 2018, the trends and state of the North Pacific stock had been evaluated mainly in a regional manner, with short time series and with different approximations. Based on information from the Taiwan Province of China longline fleet for the years 1990 to 2004, an assessment of the Pacific Northwest stock was conducted through a virtual population analysis. They observed a decrease trend from 2000, finding that the Potential Spawning Rate (SPR) had reached a level of 20% in 2003, being lower than the biological reference point (BRP SPR = 35%) and fishing mortality (F) in 2003 exceeded the BPR of current mortality (F2003 = 0.066/year, BRP F35% = 0.045/year). In this evaluation, it was concluded that the population could have been overexploited and recommended a 32% reduction in fishing efforts. Later, for the same fishery, a different group with the same information, but updated (1995-2005), included an analysis of the uncertainty in their estimates of the BRP and the same group in 2014 used a matrix demographic analysis by stages. They also concluded that the abundance of the stock of the Northeast Pacific was decreasing under fishing conditions during the study periods. A study using generalized linear models to standardize the catch rates of the longline fisheries of the Central and Northeast Pacific and making use of biological indicators, identified a recent rate of significant decrease in the catch rate for *Isurus spp.* of 7% per year during 1996-2009 (equivalent to approximately 69% for the 15-year period analyzed). However, the study indicates that the performance of the standardization model for *Isurus spp.* (North and South Pacific), was the worst compared to the models for the other shark species studied; therefore, the trends are less reliable. A study published in 2015 concluded that, based on CPUE data, catch rates were relatively stable in the North Pacific during 2000 and 2010, although they acknowledge the lack of data in some years, without being able to infer during the last 4 years. Conversely, a study published in 2017 developed a linear delta-generalized disaggregated by size and space-temporarily mixed model to analyze the catch rates of *Isurus oxyrinchus* in the Japanese Pacific Northwest and Central Pacific fishery during 2006-2014. They found that, as of 2008, the capture rates showed an increasing trend.

South Pacific (22% of the total distribution of the species)
In the South Pacific, changes in abundance for *Isurus oxyrinchus* were not significant (1996-2009) and the performance of the standardization model for the species (north and south) was worse than for the other shark species studied; therefore, their tendencies are less reliable. A study published in 2015 indicated that *Isurus oxyrinchus* in the South Pacific may have been decreasing for the past five years (2009-2013), however, the same authors argue that they are based on relatively few data and therefore, the trend estimated can be unreliable.

In Francis et al., *2014 population trends of* Isurus oxyrinchus *in New Zealand waters are based on logbooks from the domestic fishery (north and south), logbooks from the Japanese fishery in the south and observer data.*

Although it was difficult to determine trends due to wide and overlapping confidence intervals for the models covering the southern fishing grounds (i.e. TLCER Japan South, TLCER Domestic South and approximately two-thirds of the observer data), all datasets indicated peak catches during the period 2011–2013. One of the datasets (i.e. the TLCER North) suggests, on the basis of non-overlapping confidence intervals, that mako catch rates have increased between 2005 and 2012, but then dropped in 2013; however, the 2013 values are higher than values observed in the mid 2000s.

In the latest IUCN Red List assessment (accepted for publication in the March 2019 Red List update) a trend analysis using data from Francis et al., 2014 indicated annual rates of increase of 0.5% consistent with a median increase of 35.2% over three generation lengths (72 years).

Mediterranean (1.1% of the total distribution of the species)
A study in 2008 conducted an evaluation with bibliographic information of the Adriatic Sea fishery (76 records of *Isurus oxyrinchus*, *Lamna nasus*, *Sphyrna tudes* and *S. zygaena* from 1827 to 2000), the Spanish fleet catching swordfish of the Mediterranean (1991-1992) and the longline fleet of the Ligurian Sea (1990-1997). With this information they applied generalized linear models, with which they estimated a decrease greater than 96% of the baseline. *The meta-analytical estimate of the rate of decline was >99.99% for biomass (IRD: –0.15; CI 95%: –0.21, –0.10; time range: 106 years) and abundance (IRD: –0.12; CI 95%: –0.22, –0.03; time range: 135 years) (Ferretti et al., 2008). It should be noted that catch rates were very low even at the beginning of the data series, with an average of 0.2 sharks per 1,000 hooks (Walls & Soldo, 2016).* The species *Isurus oxyrinchus* was classified as Critically Endangered in a 2016 regional assessment, based on (i) inadequate management resulting in continuous (or increasing) fishing pressure, (ii) the high value of its meat and fins, (iii) the sensitive life history of the species, (iv) absence of records of some areas located in the Mediterranean Sea, (v) evidence of large declines in other regions and (vi) captures of juveniles in a probable area of nurseries. *Isurus oxyrinchus* was considered common throughout the Mediterranean Sea, currently the species is rarely found. The last known sighting of the species in the Mediterranean was in Malta in 2005 and it was reported that the species has not registered in the Adriatic Sea since 1972. A decrease of the regional population of at least 80% in the last three generations (75 years) was calculated according to the available data and it was expected that this trend would continue given the lack of management and current fishing levels.

Indian Ocean (17.9% of the total distribution of the species)
A preliminary stock assessment was conducted using reduced information, mainly catch rates of the longline fleet of European Union countries. Two models were applied, a Bayesian Schaefer-type production model and another one analyzing only the trends of the catches, reporting that the current exploitation rate exceeds the exploitation levels where the Maximum Sustainable Yield (MSY) obtained since 1990 estimated that the rate of fishing mortality F has a value much higher than the expected fishing mortality rate in the maximum sustainable yield Fmsy (F2015 / Fmsy = 2.57). It was concluded that *Isurus oxyrinchus* in the Indian Ocean has overfishing (its fishing mortality is 2.57 times greater than the Fmsy value), but it is not overfished.

Based on the proportions of Biomass / Bmsy generated by the Schaefer model (Figure 6B of the publication Brunel et al., 2018) the SS estimated that there is a historical decrease of 26% (1.6 B / Bmsy average from 1970-1980 against 1.1 B / Bmsy average from 2005-2015), a recent decrease of 18.8% with an annual decrease rate of 2.1% (1.31 B / Bmsy in 2005 against 1.06 B / Bmsy in 2015). Using the 2.1% decrease rate and starting from a value of 1.06 B / Bms, a projected 10-year decrease of 41.6% of the historical baseline was estimated (1.6 B / Bmsy average of 1970-1980 against 0.93 B / Bmsy average of the 2015-2025).

In the latest IUCN Red List assessment (accepted for publication in the March 2019 Red List update) a trend analysis of the biomass for 1971-2015 (45 years) from Brunel et al.*, 2018 revealed annual rates of decline of 0.9% consistent with a median decline of 47.9% over three generation lengths (72 years).*

A study examined the incidental catch of longlines during a research program on longline fisheries of tuna in the western equatorial Indian Ocean (1964-1988). They identified *Isurus oxyrinchus* as the second most frequent shark species caught, and a major decline in catches and mean weight for all principal shark species.

A study from the Japanese longline fleet showed trends in standardised longline Isurus oxyrinchus *CPUE with a gradual decreasing trend, with some unnatural up and down from 1994 to 2005, turning into a steady increase to 2010 (Kimoto* et al., *2011).*

A study of the Portuguese pelagic longline fishery in the Indian Ocean between 2000 and 2016 found CPUE to have high variability in the early years until 2008, followed by a general increasing trend in the more recent years until 2016 (Coelho et al., *2017).*

A review of the species in a regional assessment of elasmobranchs in the Arabian Sea and adjacent waters published in 2017 assigned *Isurus oxyrinchus* a near threatened status for the area. The authors found that the available standardized CPUE data suggested a variable abundance, but that there was little evidence of a significant population reduction. However, they noted that there was evidence of decreases in the average size of individuals in countries such as Oman, and estimated that, given the intensive pelagic fishing in the region and the high susceptibility of the species to the longline fishing gear, purse seines and drift nets, it was suspected that *Isurus oxyrinchus* had presented population decrements of 20-30% in the last three generations (75 years).

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

The main threat to *Isurus oxyrinchus* is fishing, since it is caught as a directed and incidental catch in multispecific fisheries throughout its range, particularly in pelagic longlines, targeting tuna, billfish and swordfish in national and international waters. *Isurus oxyrinchus* is valued for its high-quality meat, and fins. Because of this, it is discarded less frequently than other pelagic sharks. Its liver oil is considered of average quality. *Isurus oxyrinchus* is a target of sport fishing in the USA, New Zealand and some European countries.

Fins

Estimates of the average declared value of global imports of shark fins have been reported. These were ~ USD 22.5 /kg from 2000 to 2011, reaching USD 25.6 /kg in 2011. *Isurus oxyrinchus* is the fourth (in 1999-2000) or fifth (in 2014-2015) most abundant species observed in the shark fin trade through Hong Kong SAR's main commercial center. *Isurus paucus has also been observed in this market, although only making up 0.08% of the sample (Fields et al., 2018).* Isurus oxyrinchus *fins are found in the "Qing lian" fin category which* Isurus paucus *was sometimes mixed with (Clarke* et al.*, 2006).* Isurus paucus *was also found in the "Wu gu" fin category which was supposed to be for thresher shark species* Alopias spp*. (Clarke* et al.*, 2006).*

Using commercial data on weight, fin sizes marketed and trade name of *Isurus oxyrinchus*, together with statistical and Bayesian analysis of DNA to obtain missing records, it was estimated that *Isurus oxyrinchus* fins represent at least 2.7% of the world trade in fins from 1999-2001, may be higher given its presence in other commercial categories, reaching up to 1 million mako sharks (*Isurus spp.*= 40,000 t of the two species combined) captured annually. Between 2014 and 2015, a genetic analysis of processed fin clips and by products of fin preparation in Hong Kong SAR found that 0.2 to 1.2% of reported samples were *Isurus oxyrinchus*. Regarding the global volume of shark fin trade reported in Hong Kong SAR, in 2012 it decreased by 22% from the average recorded between 2008-2010, but the reported total average of fin volume traded in Hong Kong SAR remains at least 6,000 metric tons between 2012-2015.

Meat

Isurus spp. meat is of high quality (it is known as "veau de mer" in Europe) and it is used fresh, dried, salted, frozen and smoked for human consumption all over the world. Its price is USD 22-44 /kg in American supermarkets and it is a premium product in Japan. In Spain, *Isurus spp.* meat in wholesale markets is double the cost of blue shark meat (~ USD 14.17 /kg fresh, versus USD 7.63 /kg for blue shark, and USD 5.21 /kg versus USD 4.42 /kg frozen). This is considered high-end in Venezuela. In some areas, *Isurus spp.* meat is used as animal feed and fish meal. In Mexico, the highest commercial value of *Isurus spp.* products is reflected in the meat, which is more valued compared to that of other sharks in the market (~ USD 1/kg), followed by the caudal peduncle (for export) and subsequently the rest of the fins of the species, it is also registered that all the derivatives of the species are used, including the jaw and the heads for decoration and ornament.

Isurus spp. has the highest wholesale price of shark meat for Namibian shark exports USD 2-3 /kg. *Isurus spp.* meat and fins has been found in the Singapore market, probably imported. There are reports of Japanese companies producing 240 t/year of frozen *Isurus spp.* fillets for export to Italy and Spain.

Other products

Isurus oxyrinchus oil is extracted to obtain vitamins; the skin is processed as leather, the jaws and teeth are used for ornaments. The jaws are sold to tourists in countries such as Sudan. Most of these by-products are of low value, are commercialized in small quantities and are not recorded in trade statistics.

Extent of trade

According to FAO's global catch production statistics (1981-2016), total landings of *Isurus oxyrinchus* increased by 69% from 2004-2009 (54,155 t total in the period) to 2010-2016 (*91,989 t total in the period, reported in the SS as 45,956*). In the period from 2010 to 2016, the Atlantic contributed 50% of the total catches (45,956 t total for the period), the Pacific with 34% (31,838 t total) the Indian Ocean with 15% (14,043 t total) and the Mediterranean with less than 1% (152 t total). In these periods, the average catches per year were 9,025 t in the period from 2004 to 2009 and 12,141 t/year between 2010 and 2016 (*calculated at 13,141 t/year for this*

analysis). Spain, Taiwan Province of China and Portugal represent 62% of the annual catches reported to FAO between 2006 to 2016 (35%, 15% and 12% respectively).

Vulnerability
An ecological risk assessment (ERA) concluded that *Isurus oxyrinchus* was the second shark species most vulnerable to the Atlantic longline fisheries, and the most vulnerable in the Indian Ocean. In 2015, the ERA was reviewed, finding that *Isurus oxyrinchus* were the most vulnerable to pelagic longline fisheries in the Atlantic Ocean and are among the most vulnerable species (vulnerable to catch and mortality) from the biological point of view.

The breeding areas identified to date have been the product of fishery-dependent data, which is why there is likely to be direct exploitation pressure on them.

In terms of fishing gear, *Isurus oxyrinchus* have a post-release survival of up to 70% (depending on management and release time), higher than other shark species, making it feasible to implement measures of selective fishing management. In southeastern Australia, it was estimated that *Isurus oxyrinchus* caught by recreational fishermen on the southeastern coast of Australia (n = 30) had a survival rate of 90%.

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

Isurus paucus is very similar in appearance to *Isurus oxyrinchus*, although it has longer pectoral fins. *Isurus spp.* are grouped for the catch, landings and trade data, although a study differentiated the fin clips of the two species in 2017. *Isurus paucus* is the rarest species, its habitat and behavior are little known, but it is thought to have a more tropical distribution in oceanic waters. Of the two species, *Isurus paucus* is less abundant, but according to a study in 2006, most traders reported that they placed these fins on the same market category as *Isurus oxyrinchus* and the thresher sharks (*Alopias spp.* listed in Appendix II of CITES), due to its similarity in appearance and market value.

The dorsal fins of *Isurus oxyrinchus* and *Isurus paucus* are similar; both are dark grayish brown, have a short free tail tip, and are very erect due to the steep angle of the anterior border. The second dorsal and anal fins are extremely small. *The pectoral fins differ considerably in size between the two species (Barreto, in litt., 2019)*, and are shorter than the length of the head, and their ventral surface is uniform white or light in color, with no obvious dark markings. There are strong lateral keels on the caudal peduncle. The caudal fin is crescent shaped, with symmetrical upper and lower lobes. Despite the similarity it presents with *Isurus paucus*, the fins of *Isurus oxyrinchus* are easy to visually identify whether fresh or dry, since there are differences in the dermal denticles of the two species.

Additionally, these two species differ in the lower jaw with 11-13 rows of teeth in *Isurus paucus* and over 13 rows in *Isurus oxyrinchus*.

However, since both species are traded for the value of their meat (more than 90% of the total volume of their body), most of the volume traded will be difficult to identify.

Additional Information

Threats
As noted above, the main threat to *Isurus oxyrinchus* (*and presumably Isurus paucus*) is fishing, since it is caught as a directed and incidental catch in multispecific fisheries throughout its range, particularly in pelagic longlines, targeting tuna, billfish and swordfish in national and international waters. *Isurus oxyrinchus* is also a target of sport fishing in the USA, New Zealand, South Africa and some European countries.

Other threats include incidental catch in bather protection in the South East Indian Ocean, with reports of a small number of individuals caught annually in shark nets off the beaches of KwaZulu-Natal.

Finally, because temperature is an important environmental factor for the spatial and temporal distribution of *Isurus oxyrinchus*, the use and distribution of its habitat would likely be affected by the warming of ocean waters as a result of climate change.

Conservation, management and legislation
National
Range States have adopted a variety of national instruments, some applied through laws and regulations on fisheries and trade, others through wildlife legislation or other environmental legislation.

International
General measures across all sharks have been adopted by Regional Fisheries Management Organizations (RFMOs) in the form of recommendations or resolutions which contracting parties must then implement and report. Almost all have adopted a ban on the practice of finning (cutting shark fins and discarding the body at seas), and members require that their vessels do not have fins that add more than 5% of the weight of sharks on board, until the first landing point.

Isurus oxyrinchus is considered a highly migratory species in Annex I (adopt measures for the conservation of species) of the United Nations Convention on the Law of the Sea (UNCLOS) and Appendix II (species conserved through agreements) of the Convention on Migratory Species. In turn, several RFMOs recommend their Parties to a number of measures such as improving data collection and having population and risk assessments.

Isurus oxyrinchus is listed in Annex 3 of the Berne Convention on the Conservation of European Wildlife and Natural Habitats (species that need protection but can be exploited in exceptional cases) and is one of the 20 sharks and rays listed in the Annex II of the Protocol on Specially Protected Areas and Areas of Biological Diversity in the Mediterranean (under the Barcelona Convention). The General Fisheries Commission for the Mediterranean (GFCM) adopted the list in Recommendation GFCM / 36/2012 / 3. Likewise, the Contracting Parties and Cooperators (CPC) were requested to protect these species from fishing activities and release them. They cannot be retained on board, transshipped, disembarked, stored, exhibited or sold. The recommendation, I / OAT 17/08, requires (with some exceptions and conditions) that vessels promptly release live *Isurus spp.* retained in the North Atlantic; dead sharks can be preserved, and in some cases also live sharks over a minimum size. Records should be retained and sent to ICCAT, and landings should not exceed the previous average of vessels for *Isurus oxyrinchus*. The current measure will be evaluated and will expire on December 31, 2019. The scientific advice for this overfished stock is that harvest should not exceed an annual catch of 500 t, to stop overfishing and begin to rebuild the stock. The measures adopted will probably result in catches well above this minimum. The recommendation is oriented towards the conservation of *Isurus oxyrinchus* stock of the North Atlantic, considering that it is caught in association with ICCAT fisheries and presents a state of overexploitation and overfishing.

In 2008, the CMS proposal to include *Isurus oxyrinchus* in Appendix II identified a number of national protection measures, including: incidental catch and recreational bag limits - South Africa; management under a quota system - New Zealand; regulation of fishing gear for artisanal fisheries - Chile; commercial quotas, limited entry and time closings, and recreational bag limits - United States Atlantic; closure of the directed longline fishery, recreational fishing limits in California and catch guidelines for California, Oregon and Washington - United States Pacific; COSEWIC assessment as an "at-risk" species, catch and incidental catch limits, license limits, fishing gear restrictions, area and season bans, single hook and recreational dump - Canadian Atlantic; limited closures of entry and time - Pacific of Canada.

Potential risk(s) of a listing
There is currently legislation in place in its range where population trends are declining. It is unclear whether this legislation is having a conservation benefit yet *as it is difficult to assess the extent to which they are being implemented and the extent to which they are reducing declines*. However, any CITES listing should be careful not to have a detrimental affect on the current legislation or be seen as replacing current legislation.

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Inclusion of Guitarfish *Glaucostegus* spp. in Appendix II

Proponents: Bangladesh, Benin, Bhutan, Brazil, Burkina Faso, Cabo Verde, Chad, Côte d'Ivoire, Egypt, European Union, Gabon, Gambia, Maldives, Mali, Mauritania, Monaco, Nepal, Niger, Nigeria, Palau, Senegal, Sierra Leone, Sri Lanka, Syrian Arab Republic, Togo and Ukraine

Summary: Giant Guitarfish species in the family Glaucostegidae are shark-like batoid species occurring in the coastal waters of the Mediterranean and Black Sea, Atlantic, Pacific and Indian Oceans. There are six species in the family, all in the genus *Glaucostegus*. The majority of the species are strongly associated with soft bottom habitats in shallow (<50 m) warm-temperate to tropical coastal waters. Species in the Glaucostegidae family grow slowly, mature late, have a generation length of 10–15 years, and exhibit low to medium productivity with many growing larger than 2 m in total length.

The global population sizes are unknown for all of the species of Glaucostegidae. There are no known stock assessments for any of the species and all information on population trends is based upon fisheries landings and inferred from fishing effort. All six species in the family have recently been assessed by IUCN as Critically Endangered with estimated declines of greater than 80% over the last three generations having occurred (accepted for publication in the July 2019 Red List update). These estimates have been based on new datasets documenting global population reductions.

The primary threats to Glaucostegidae are considered to be unmanaged and unregulated fisheries. Some of the catch is targeted and incidental catch is often retained. Fishing techniques used in these coastal regions mean that species of Glaucostegidae are exposed to intensive fishing. Coastal development is also a threat to these species with human populations increasing, leading to increased fishing pressure, but also habitat degradation, threatening key habitats in the species' life cycle.

Scant species-specific catch or trade data are available; information is often reported using generic terms such as "guitarfish", "giant guitarfish" or "rhinobatid", which likely also include species from other families. Glaucostegids are also known to be caught and reported alongside Rhinidae (wedgefish) species using terms such as "guitarfish etc, nei". Localised landing declines have been reported, for example, in India, an 86% decline in landings reported as guitarfish and wedgefish has been observed from one landing site over a five-year period (2002–2006). In Iran, 66% declines in "giant guitarfish" landings occurred over 20 years (1997–2016), while in Pakistan landings data for "rhinobatids" at two sites showed landings decline by 72% between 1999 to 2011, and 81% between 1994 and 2011. While most of the declines have been reported for species in the Indo-West Pacific, it is highly likely that similar declines are occurring in other regions where fishing pressure is likely to be similar.

Detailed information is given below for two species known to be affected by trade:

- *Glaucostegus granulatus:* A recent IUCN Red List assessment categorised this species as Critically Endangered due to declines of >80% over the last three generations. In India, *G. granulatus* landings declined by 94% over five years from 2002.

- *Glaucostegus cemiculus:* A recent IUCN Red List assessment also categorised this species as Critically Endangered due to declines of >80% over the last three generations. There is evidence that 95% of *G. cemiculus* caught are below their size at maturity. The species' status in the Mediterranean is unclear; while local extirpation is reported in the northern Mediterranean it is still present in the south, but it is expected that there will be declines. The species is reported to be taken in Gambia, Guinea, Guinea-Bissau, Mauritania and Sierra Leone.

Glaucostegus granulatus and *G. cemiculus* occur in some of the most heavily fished coastal regions in the world.

While it appears the meat from glaucostegid species is primarily used locally, the fins of these species are reported to be exported. Fins from *Glaucostegus cemiculus* have been identified in shops in Hong Kong Special Administrative Region (SAR), included in the highest valued categories of fins, "Qun chi". They have been observed for auction/sale in Bangladesh, Oman and United Arab Emirates. The form that species are traded in (fins, meat) makes it hard to differentiate between species within the Glaucostegidae family, often requiring genetic analyses. There is conflicting information on whether Glaucostegidae fins are morphologically similar to those from the family Rhinidae (subject to a separate listing proposal, Prop. 44) and Pristidae (listed in Appendix I in 2007except *Pristis microdon* which was transferred from Appendix II to I in 2013), once removed from the whole animals, particularly in the processed form.

Legislation and regulation for Glaucostegidae vary by location and country; where they exist, they primarily cover a range of measures involving either the banning of commercial fishing of sharks in certain zones, size limits or banning the practice of finning. Fishing restrictions in Mauritania have apparently resulted in increases in relative abundance of *Glaucostegus cemiculus*.

Analysis: Species of Glaucostegidae are found in coastal waters in the Mediterranean and Black Sea, Atlantic, Pacific and Indian Oceans. While their meat appears to be utilised locally, fins from these species have been observed in international trade, which is presumed to drive the retention of the species as incidental catch.

In the most recent IUCN Red List assessments (to be published July 2019), all six of the species were said to have undergone declines of greater than 80% over the past three generations and therefore already meet the biological criteria for inclusion in Appendix I owing to a marked recent rate of decline. Over-harvest is identified as the main factor driving these declines. Therefore, it is likely that for all of the species in this family, regulation of trade is required to ensure that harvest from the wild is not reducing populations to levels where survival might be threatened by continued harvest or other influences. Management measures have shown to be successful in restoring populations where they are implemented.

Due to the difficulties in differentiating the species in the form that they are traded in, if any of the species are considered to meet the criteria then all should be listed.

Other Considerations: Species of Rhinidae (subject to a separate listing proposal, Prop, 44) and Glaucostegidae are often landed and traded together. Therefore, if one of the proposals is accepted then the species in the other family should be included in Appendix II to ease implementation.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Glaucostegidae (Giant guitarfishes) are one of five families in the Order Rhinopristiformes which is comprised of four families of "guitarfishes" (Rhinobatidae (guitarfishes), Rhinidae (wedgefishes), Trygonorrhinidae (banjo rays), and Glaucostegidae (giant guitarfishes)) and family Pristidae (sawfishes). There are six species in the Glaucostegidae family. *Some recent changes in the systematics of the genus Rhinobatos have elevated the subgenus* Glaucostegus *to full generic status and placed this genus into a family of its own: Glaucostegidae (Compagno* et al., *2005, Last* et al., *2016). Therefore, some of the information below may be originally reported as Rhinobatos but where possible notes are provided on what was originally reported and which species it should be interpreted as.*

IUCN Global Category and Range

The species in the Glaucostegidae family have recently been reassessed for the IUCN Red List, with new assessments being accepted for publication in the July 2019 Red List update. All species have had their Red List category changed due to new datasets documenting global population reductions in Glaucostegidae. All the

species in the family have been assessed as Critically Endangered, with declines of >80% over the last three generations having been inferred to occur.

Blackchin Guitarfish *Glaucostegus cemiculus*
Critically Endangered A2d (ver 3.1, assessed 2018) (Kyne and Jabado, 2019a, accepted for publication in the July 2019 Red List update)

Glaucostegus cemiculus is an eastern Atlantic species of Giant Guitarfish, the range of which extends from Portugal, throughout the Mediterranean and as far south as Angola. *It is found in the following FAO Fishing Areas: southeast Atlantic, northeast Atlantic, east central Atlantic, Mediterranean and Black Sea. Range countries/territories are Albania, Algeria, Angola, Bosnia and Herzegovina, Cameroon, Cape Verde, Congo, Croatia, Cyprus, Côte d'Ivoire, Egypt, France, Gabon, Gambia, Ghana, Greece, Guinea, Guinea-Bissau, Israel, Italy, Lebanon, Liberia, Libya, Malta, Mauritania, Monaco, Montenegro, Morocco, Namibia, Nigeria, Senegal, Sierra Leone, Slovenia, South Africa, Spain, Syrian Arab Republic, Togo, Tunisia, Turkey and Western Sahara (Notarbartolo di Sciara* et al., *2016).*

Sharpnose Guitarfish *Glaucostegus granulatus*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne and Jabado, 2019b, accepted for publication in the July 2019 Red List update)

Glaucostegus granulatus is found in the north-western Indian Ocean, *as north as Kuwait and Iraq* to Myanmar *(R. Jabado* in litt., *2019). It is found in the following FAO Fishing Areas: the western central Pacific Ocean, the western and eastern Indian Ocean (Marshall & Last, 2016). Range countries are Australia; India (Andaman Is.); Indonesia; Kuwait; Myanmar; Pakistan; Papua New Guinea; Philippines; Sri Lanka; Thailand; Viet Nam (Marshall and Last, 2016).*

Halavi Guitarfish *Glaucostegus halavi*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne and Jabado, 2019c, accepted for publication in the July 2019 Red List update)

Is endemic to the Arabian Seas region, occurring in coastal waters of the Red Sea, Gulf of Aden, the Arabian Sea from Yemen to northern India (Gujarat) and the Gulf (Simpfendorfer et al., *2017). It is found in the western Indian Ocean FAO fishing area (Simpfendorfer* et al., *2017). Range countries are Bahrain, Djibouti, Egypt, Eritrea, India (Gujarat), Iran, Islamic Republic of, Oman, Pakistan, Qatar, Saudi Arabia, Sudan, United Arab Emirates and Yemen (Simpfendorfer* et al., *2017).*

Widenose Guitarfish *Glaucostegus obtusus*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne and Jabado, 2019d, accepted for publication in the July 2019 Red List update)

It is primarily an eastern Indian species with range countries including Bangladesh, India, Indonesia, Malaysia, Myanmar, Pakistan, Sri Lanka and Thailand (Compagno and Marshall, 2016). It is found in the following FAO Fishing Areas: western and eastern Indian Ocean and the western central Pacific (Compagno and Marshall, 2016).

Clubnose Guitarfish Glaucostegus thouin
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne and Jabado, 2019e, accepted for publication in the July 2019 Red List update)

It is widespread with an Indo-West Pacific distribution. Possibly Suriname and the Mediterranean (Compagno and Last 1999). It is native to the following FAO Fishing Areas: western central and northwest Pacific and the eastern and western Indian Ocean. It is also found in the western central Atlantic and the Mediterranean and Black Sea although the origin is uncertain (White and Marshall, 2016). Range countries are Bangladesh, Djibouti, Egypt, Eritrea, Ethiopia, India, Indonesia (Sumatra, Kalimantan, Java), Iran, Islamic Republic of, Iraq, Japan, Kuwait, Malaysia, Myanmar, Oman, Pakistan, Papua New Guinea, Qatar, Saudi Arabia, Singapore, Somalia, Sri Lanka, Sudan, Thailand, United Arab Emirates, Viet Nam and Yemen.

Giant Guitarfish *Glaucostegus typus*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne et al., 2019a, accepted for publication in the July 2019 Red List update)

It is widely distributed in the Indo-Pacific (Compagno and Last 1999; Last and Stevens, 2009). It is native to the following FAO Fishing Areas: western central and southwest Pacific and the eastern and western Indian Ocean (White and McAuley, 2016). Range countries are Australia (Western Australia, Queensland, Northern Territory, New South Wales), Bangladesh, India, Indonesia, Malaysia, Singapore, Sri Lanka and Viet Nam (White and McAuley, 2016).

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

Biology

The vast majority of guitarfish species (referring to the four families: Rhinidae, Rhinobatidae, Glaucostegidae, and Trygonorrhinidae) are strongly associated with soft-bottom habitats in shallow (<50 m, but usually considerably less) warm-temperate to tropical coastal waters (Moore, 2017). Most available information relates to Glaucostegus cemiculus and G. granulatus*, and these are discussed further below.*

Glaucostegus cemiculus are subtropical coastal species that live in marine and brackish waters. They are usually found on sandy bottoms close to the coastlines, from the intertidal up to maximum depths of 80–100 m. Pregnant *G. cemiculus* move inshore to give birth and mate, making them more susceptible to targeted fisheries during this period. Males of this species tend to move inshore several months after females have given birth, to mate close to shore.

Glaucostegus granulatus also reside on sandy bottoms, from the coast to mid-continental shelf, with a maximum depth of at least 120 m. Much less is known of this species' habitats and migratory patterns, although they are also thought to migrate to mate and give birth.

Glaucostegidae are genetically more closely related to sawfishes than to the other wedgefish or guitarfish families and have very similar biological characteristics. Glaucostegidae grow slowly, mature late and *are considered to be species of low to medium productivity (FAO, 2019)*, making populations highly susceptible to overfishing.

Glaucostegus cemiculus
Litter sizes: 6–7 *(max. 24) (Seck* et al., *2004)*
1–2 litters per year
Gestation period: 4–6 months
Generation period: 5–10 years
Maximum total length: 265 cm

Glaucostegus granulatus
Litter sizes: 6–10
1 litter per year
Gestation period: 4–8 months
Generation period: 13 years

Population size

The global population size is unknown for any *Glaucostegus* species. There are no stock assessments for any Glaucostegidae. All information on population trends centres upon fisheries landings.

Population trends

Both Glaucostegus cemiculus *and* G. granulatus *populations have significantly declined, new datasets suggest that both species are likely to have decreased by >80% over the last three generations (IUCN Red List assessments, accepted for publication in the July 2019 Red List update)*. Historically, giant guitarfish *(presumed to mean Glaucostegidae)* catches have only been reported at the genus level, making it difficult to determine levels of population declines at the species-level.

Glaucostegus cemiculus
Atlantic Ocean
In recent years, overall catches of sharks in the east Atlantic have decreased. Some species, such as sawfish, have almost completely disappeared, while formerly common species such as giant guitarfish *(presumed to mean Glaucostegidae)* have become very rare, indicating sharp population declines. Prior to the 1970s elasmobranchs were caught primarily for local consumption. Starting in the 1970s, in response to the high demand for shark fins in eastern and southeastern Asia (primarily China), an unsustainable shark fishing industry

developed and grew rapidly. Because of this new demand, in addition to being targeted for their meat, guitarfishes were targeted for their fins, which are regarded as highly valuable. This combined with an increase in coastal populations resulted in the over-exploitation of marine resources with significantly increasing fishing effort and decreasing yields beginning in the 1990s, including the overfishing of demersal species. Since 2003, there has been a significant decline in elasmobranch landings. In the sub-region, elasmobranch fishing was initially concentrated in Gambia and Senegal, but it has spread to the other member countries as fishers migrate in response to areas becoming over-exploited.

Glaucostegus cemiculus used to be taken in great numbers in Senegal, Guinea, Guinea-Bissau and Sierra Leone, until increased fishing pressure on shark stocks since the 1980s depleted populations of the more susceptible species within these waters. Giant guitarfishes *(presumed to mean Glaucostegidae)* have almost completely disappeared from the region. In Senegal, landings have dropped by 80% in 7 years—from 4,050 t in 1998 to 821 t in 2005—indicating a severe drop in the population of this species*. In the United Nations Food and Agriculture Organization (FAO) statistics, Senegal reported 171 t in 1998 and 920 t in 2005 for "Guitarfishes, etc. nei" from the eastern central Atlantic (FAO, 2018). It is unclear why this difference exists with the SS.* Data collection in the region is poor, and much of the species' decline must be inferred from reduction in catches, particularly since fishing pressure in the region has increased as additional people have moved into coastal regions. *This needs to be interpreted with the knowledge that fishing effort in the region has not decreased during this period (C. Simpfendorfer,* in litt., *2019). As one of the most heavily fished coastal regions in the world, these declines in catch are a useful indicator of declining abundance (C. Simpfendorfer,* in litt., *2019)*.

There is evidence that 95% of *Glaucostegus cemiculus* that are caught are below their size at maturity, affecting the ability of the population to reproduce and recover. In addition, populations of *G. cemiculus* are suspected to decline *on the basis of the severe declines in other guitarfishes and wedgefishes, the continuation of fishing pressure in shallow coastal habitats, the potential for fishing effort to shift towards the further targeting of guitarfish in light of their highly valued fins, particularly in the absence of other sharks (Notarbartolo di Sciara* et al., *2016).*

<u>Mediterranean Sea</u>
Recent attempts to assess population status have concluded that there is an apparent local extinction in the Mediterranean (Balearic Islands and Sicily), *based on fisherman reporting their relative frequency during the first half of the 20th century and a personal observation reporting they seem to be extinct in the area*. *Glaucostegus cemiculus* is no longer recorded in bottom trawl surveys, as well as from landings in several places across the Mediterranean. Around Italy, *G. cemiculus* appear to have been extirpated. *Glaucostegus cemiculus* was once thought to be common in the Gulf of Gabes, and on the eastern coast of Tunisia in the southern Mediterranean. *A study analysing gillnet fishery elasmobranch catches in the Gulf of Gabes conducted in 2007 and 2008 found that G. cemiculus (reported as Rhinobatos cemiculus) was the most abundant species, comprising 52% of catches (4.6 ind km^{-2} net per day) with a total weight of 2.85 t (Echwikhi* et al., *2013). In the Mediterranean, a decrease in localised landings and reduction in the size of specimens landed has been observed since the 1990s (Soldo and Bradai, 2016). In a Regional Red List assessment for the Mediterranean in 2016, a projected decline of ≥50% within three generations (15–30 years) was suspected based on i) the disappearance of* G. cemiculus *from localised regions of the Mediterranean Sea; ii) severe declines in other guitarfishes (family Rhinobatidae) and wedgefishes (family Rhyncobatidae); iii) the continuation of fishing pressure in shallow coastal habitats; and iv) the potential for fishing effort further to target guitarfishes for their highly valued fins, particularly in the absence of other overfished shark species (Soldo and Bradai, 2016).*

<u>*Glaucostegus granulatus*</u>
<u>Arabian Sea and adjacent waters</u>
Glaucostegus granulatus is endangered in the Arabian Sea and adjacent waters region. Although it was the most commonly reported shark-like batoid in research trawl surveys in the United Arab Emirates (UAE) during 2002–2003, a 2016 trawl survey did not record any specimens. Recent estimates have put *G. granulatus* population declines at 50–80% over the past three generation periods *(39 years) in the Arabian Sea.* With consistent ongoing high levels of fishing pressure, especially in coastal areas, declines will most likely continue into the future *(2017–2056)* unless effective management measures are introduced.

<u>Indian Ocean</u>
Fishing efforts for shark-like batoids are particularly high in South-east Asia, where steadily decreasing population sizes have been inferred from declining catch rates.

In India, a decline of 86% was noted in the landings of wedgefishes and guitarfishes, including Glaucostegidae species *(*Glaucostegus granulatus *and* G. obtusus*, and likely to include* G. typus *and* G. thouin *(Kyne et al., 2019b)*, over the five-year period 2002–2007. *At the species level,* G. granulatus *(reported as*

Rhinobatos granulatus*) landings had declined by 94% (Mohanraj* et al., *2009). It should however be noted that this data is from one landing site only in Chennai and care needs to be taken extrapolating when additional data are not available. However, R. Jabado (*in litt., *2019) notes that fishing pressure is intense and increasing along the whole coast of India.*

"Giant guitarfish" (a grouping which likely includes Glaucostegus granulatus *and* G. halavi*) landings data are available from Iran for 1997–2016 (FAO, 2018). Landings declined by 66% over this period, which is the equivalent of 81–99% population reductions over the last three generations of wedgefishes and guitarfishes (30–45 years) (Kyne et al., 2019b).*

Landing data for the "rhinobatid" category are available from Pakistan for 1993–2011 (19 years) covering the country's two coastal provinces (M. Gore, unpubl. data). This grouping likely includes G. granulatus, G. halavi *and* G. obtusus*. Data from the Sindh province showed a 72% decrease, from peak landings in 1999 to a low in 2011, and data from Balochistan province showed an 81% decrease from landings in 1994 to a low in 2011. These decreases are the equivalent of 94–99% population reductions over the last three generations of wedgefishes and giant guitarfishes (Kyne et al., 2019b).*

Landings data for "whitespotted wedgefishes" are available from Indonesia for 2005–2014 (10 years) (DGCF, 2015). This grouping likely includes Rhina ancylostoma, Rhynchobatus australiae, Rhynchobatus cooki, Rhynchobatus palpebratus *and* Rhynchobatus springeri *(this grouping may also include species in the Glaucostegidae family, but trends are likely to be representative). Landings declined by 74% over this period, which is the equivalent of 98–99% population reductions over the last three generations of wedgefishes and giant guitarfishes (Kyne et al., 2019b).*

Catch data for myliobatoid rays (this includes a variety of demersal rays but does not include rhinopristoids) which can be used to infer declines in wedgefishes and guitarfishes given overlapping distributions, habitat and susceptibility to capture in the same fishing gear are available from Maharashtra, western India for 1990–2004 (15 years) (Raje and Zacharia, 2009). The catch rate declined by 63% over this period (while fishing effort doubled), which is the equivalent of 86–95% population reductions over the last three generations of wedgefishes and giant guitarfishes (Kyne et al., 2019b).

Global Trade – FAO statistics

The SS states that in 2016, the FAO (2016) reported 5000 t of "guitarfish" landed globally in 2014 *(total weight was 5155 t with 56 t estimated by the FAO for Libya)*, but this is most likely an underestimation of total global landings, because underreporting and misidentification is common for elasmobranch species. When elasmobranch catches are reported, the data were generally not reported at the species level.

In the United Nations Food and Agriculture Organization (FAO) statistics:

- Glaucostegus cemiculus *is reported in capture production data from Mauritania from the eastern central Atlantic dating back to 2010, averaging 141 t per year. The quantities appear relatively stable across the years.*
- *"Guitarfishes, etc. nei" are reported in capture production data for Benin, Cote d'Ivoire, Liberia and Senegal averaging 49, 8, 89 and 1,117 t per year.*
- *"Giant guitarfish" are reported in capture production data for Iran dating back to 1997, averaging 187 t per year from the Indian Ocean. Initially large volumes are reported in 1997 and 1998, 880 and 593 t respectively, in the following years although volumes are lower, they currently appear relatively stable.*
- *Three countries report "Guitarfishes, etc. nei" capture from the Indian Ocean: Eritrea in very small volumes (3, 1, 1 and 7 t in 1994, 1995, 2001 and 2002 respectively), Indonesia averaging 171 t between 2009 and 2016, and Pakistan averaging 1,192 t between 1987 and 2016 (FAO, 2018).*

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

From 2004 to 2010, the number of trawlers in *the Gujarat region of India* increased from around 6,600 to 11,500 boats, but no regulations were introduced to manage this increased threat to the guitarfishes and other vulnerable species.

In UAE, bodies of glaucostegids landed were either sold fresh at market stalls after the fins were removed, or whole bodies were sold dried at the market with fins attached (Jabado, 2018). Bodies of small rhinobatids and glaucostegids (with fins removed) sold for less than AED 30 (USD 8) (Jabado, 2018). Separated fins were not sold at markets or landing sites and therefore prices were not available (Jabado, 2018).

Guitarfish *(reported as Glaucostegus granulatus, Glaucostegus obtusus, Rhina ancylostoma and Rhinobatos annandalei)* meat is considered to be good quality and is often consumed locally *in Bangladesh*.

In West Africa, the price paid to fishermen for shark fins *(not clear what falls under "shark fins")* may be as high as USD100 per kg or EUR100 per kg for *Glaucostegus cemiculus (based on a personal communication, as reported in Notarbartolo di Sciara* et al., *2016)*. *Glaucostegus cemiculus* meat is consumed domestically as fresh or dried fish, known as "sali". *Glaucostegus cemiculus* landed in The Gambia have their dorsal and caudal fins removed, likely for export because of their high value.

Glaucostegus cemiculus fins have been seen in shops in Hong Kong SAR *after genetic analysis of Qun chi category fins (Bloom, 2018)*, for auction and sale in Oman and United Arab Emirates *(Jabado, 2018)*, and in some instances in Bangladesh, where large numbers of *G. granulatus* are caught (*Haque* et al., *2018*). Because no part of the animal goes to waste, but no shark fin soup is consumed within Bangladesh, these high value fins are likely sold for export. 15,000 kg of guitarfish products (including *G. granulatus* and *G. obtusus*) were recorded moving through a single processing centre in Cox's Bazar from 2012–2015.

Giant guitarfish (*presumed to mean Glaucostegidae)* products enter trade legally, unless taken in contravention of national legislation or regional fisheries management measures (*although R. Jabado (*in litt., *2019) reports that there are no regional fisheries management measures in place for these species*). Because management is so scarce, almost all trade of Glaucostegidae products is legal. *Due to this, illegal trade does not occur, however, trade can occur in illegally harvested specimens when specimens are* caught within marine protected areas and shark sanctuaries, and those that are finned in range States or fisheries where shark finning is prohibited.

The regions that these two species occur in are some of the most heavily fished coastal regions in the world (C. Simpfendorfer, in litt., *2019). Given their coastal distribution, and the massive fleets of vessels fishing these areas, I would conclude that declines of these species will continue unless significant action is taken (C. Simpfendorfer,* in litt., *2019).*

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

The most common fins in trade from Glaucostegidae are the first and second dorsal fins and entire caudal fins, which are typically traded as a set from each individual animal. Due to distinctive fin shapes and denticle size, it is likely that these fins can be identified to the family level (i.e., all *Glaugostegus* species) in their most commonly traded dried and unprocessed form.

Because of the morphological similarities among the six Glaucostegidae species, these are often mislabelled by both the scientific and fisheries communities. For example, a study has misidentified *Glaucostegus halavi* as *G. granulatus*. This issue is not unique to giant guitarfish *(presumed to mean Glaucostegidae)*, but rather is common across all Rhinopristiformes. Identification even of whole bodies of giant guitarfish *(presumed to mean Glaucostegidae)* species from this family can be difficult, which is why these are proposed for listing as lookalikes for *G. cemiculus* and *G. granulatus.*

Fins from the family Glaucostegidae are morphologically similar to those from the family Rhinidae (wedgefishes) and Pristidae (sawfishes), once removed from the whole animals. Although, visual identification to the family level for unprocessed fins is possible, it makes identification of fins between these three groups that have been heavily processed (skin removed) challenging. *Jabado* in litt., *2019 reported that the fins from the family Rhinidae are not morphologically similar to those from the family Glaucostegidae, they are similar to each other in the family—but Rhina and Rhynchorhina are distinct (dark, spotted)—and the fins of the remaining Rhynchobatus are different from the giant guitarfishes (Glaucostegidae) because of the enlarged denticles present on those (vs lacking on Rhinidae) for the dorsal fins—and for the caudals (which are usually sold whole) Rhynchobatus have a distinct upper and lower caudal lobe whereas Glaucostegidae species don't.*

*C. Simpfendorfer (*in litt., *2019) reported that there is morphological similarity between the species in Glaucostegidae. It is especially true in the form that they are traded in (fins, skins and meat) that can only be delineated with confidence using genetic techniques (C. Simpfendorfer,* in litt., *2019).*

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

The SS states that with similar demand, high value, morphological, genetic, and life history characteristics to sawfishes, whose declining populations were ignored by fisheries and trade management for almost three decades and whose populations are now unlikely to recover, the trade of giant guitarfish (*presumed to mean Glaucostegidae*) species must be regulated in order to prevent a similar fate of near extinction.

Additional Information

Threats

The primary threats to *Glaucostegus cemiculus* and *G. granulatus* are unmanaged and unregulated fisheries and trade. *Glaucostegus cemiculus* and *G. granulatus* are targeted species in West Africa, north-western Indian Ocean, and South Asia, largely due to their highly valuable fins.

Furthermore, females are more susceptible to fisheries because they are often larger than males, with more valuable fins, and move in closer to shore to give birth and mate. With an increase in coastal development, and the increasingly high demand for their large fins, population declines will likely be further accelerated.

Coastal development is a major threat to giant guitarfish species, impacting the habitat where they mainly occur and particularly threatening their inshore breeding, nursery and mating grounds.

Conservation, management and legislation

Management is likely to vary by species and geographic areas, examples of species-specific legislation are provided below.

Glaucostegus cemiculus

Mediterranean

As reported by Newell (2016), in 2009, Glaucostegus cemiculus *was listed on Specially Protected Areas and Biological Diversity Annex III: List of Species Whose Exploitation is Regulated, which was adopted under the Barcelona Convention in 1995 (Bradai* et al., *2012). In 2012 it was uplisted to Annex II: List of Endangered of Threatened Species. This protocol charges all parties with identifying and compiling lists of all endangered or threatened species in their jurisdiction, controlling or prohibiting (where appropriate) the taking or disturbance of wild protected species, and co-ordinating their protection and recovery efforts for migratory species, among other measures that are likely less relevant to* G. cemiculus *(RAC/SPA 1996). Currently all coastal Mediterranean countries where this species occur are contracting parties (European Commission, 2016). Further, since 2012, it has been protected by General Fisheries Commission for the Mediterranean (GFCM) recommendation GFCM/36/2012/3. To protect elasmobranchs this recommendation prohibits the sale of sharks that cannot be identified before the first point of sale, such as those that have been finned, skinned, or beheaded, and it prohibits trawling in the first three nautical miles off the coast or up to the 50 m isobaths (whichever comes first). Additionally, Annex II and III species cannot be retained on board, transhipped, landed, transferred, stored, sold or displayed or offered for sale, and must be released unharmed and alive to the extent possible (GFCM, 2012). Any capture of these species in the GFCM area of competence, which includes all national and high seas waters of the Mediterranean and Black Seas, is considered illegal, unreported and unregulated fishing (FAO, 2016).*

*The Mediterranean Action Plan for the conservation of chondirchthyan fishes recognises the urgent need to assess the status of Rhinobatos species (*Glaucostegus cemiculus *was formerly* Rhinobatos cemiculus*), as a species that may be at high risk of threat in the region (Notarbartolo di Sciara* et al., *2016).*

Mauritania

*This species is protected as part of a ban on directly targeted elasmobranch fishing in the Banc d'Arguin, Mauritania, which was implemented in December 2003 (R. Jabado (*in litt., *2019), reports that some traditional fishing using nets is still allowed and there is a high level of bycatch of sharks and rays). In this national park, management measures specific to this species were also introduced (Notarbartolo di Sciara* et al., *2016). The fishery was closed from February to September, to avoid the parturition period and therefore the capture of pregnant females. After negotiations with local fishermen, gear restrictions were introduced (to stop fishing with bottom gillnets of 1–2 m height and 11–16 cm (one side) square mesh). Increased abundance has been observed in and around the bank, suggesting the protection was effective (Notarbartolo di Sciara* et al., *2016).*

Sierra Leone

There are no species-specific regulations for the management of shark and shark fisheries in Sierra Leone. However, a licensing system for artisanal fishing canoes, both foreign and Sierra Leone owned is payable to the

Ministry of Fisheries and Marine Resources and the Local Government Administration (Seisay, 2005). A National Action Plan for the conservation and management of sharks is being proposed. Some of the recommended management measures include: area and seasonal closure to shark fisheries, effort limitation of the shark fishery and discarding of immature and/or juvenile shark and ray species.

Guinea-Bissau
There are marine protected areas inside the Bijagos archipelago (the Formosa Islands UROK marine reserve), the PNO marine reserve (Orango Islands) and the PNMJVO marine reserve (Joao Vieira and Poilao Islands). Within these areas, trawling and the use of nets is forbidden, the only gear type allowed is fishing with longlines. Furthermore, fishing is only allowed for subsistence purposes, commercial fishing is not permitted (Bucal, 2006).

Glaucostegus halavi
Kuwait and Pakistan are the only countries across the range of this species with regulations specifically banning catches of rays. Kuwait bans the catches of all rays, while Pakistan protects all guitarfishes, wedgefishes and the Bowmouth Guitarfish Rhina ancylostoma *and therefore has specific regulations protecting this species. The United Arab Emirates, Qatar and Oman have banned trawling in their waters (since 1980, 1993 and 2011, respectively) while the Islamic Republic of Iran, Kuwait and Saudi Arabia have seasonal trawl bans that might benefit the species. However, incidental catches occur in other fisheries (e.g., gillnetting). Finning has also been banned in the United Arab Emirates, Oman and Iran, yet trade statistics indicate that some trade in the fins and meat of this species still occurs.*

Implementation challenges (including similar species)
As they are often being caught in multi species net fisheries, they are likely to continue being caught. The species survival probability after capture will be important when determining whether implementation will have conservation benefits.

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Inclusion of all species of Wedgefish in the family Rhinidae in Appendix II

Proponents: Bangladesh, Benin, Bhutan, Brazil, Burkina Faso, Cabo Verde, Chad, Côte d'Ivoire, Egypt, Ethiopia, European Union, Fiji, Gabon, Gambia, India, Jordan, Kenya, Lebanon, Maldives, Mali, Mexico, Monaco, Nepal, Niger, Nigeria, Palau, Philippines, Saudi Arabia, Senegal, Seychelles, Sri Lanka, Sudan, Syrian Arab Republic, Togo and Ukraine

Summary: Rhinidae (known as Wedgefish) are shark-like batoid species comprising up to ten species, in three genera (*Rhynchobatus*, *Rhynchorhina* and *Rhina)*. They inhabit shallow, inshore continental waters of the east Atlantic, Indian and western Pacific Oceans, often occurring in muddy enclosed bays, in estuaries and on coral reefs. They are not known to penetrate fresh water. Little is known about their biology, but some species are known to grow to 3 m in total length, and they are considered low productivity species with a generation length of 10–15 years.

Whilst the global population sizes are unknown for all Rhinidae species, populations are reportedly declining based on inferences from fisheries landings, fishing effort, or declines of similar species. Eight species of Rhinidae have recently been assessed by IUCN as Critically Endangered (declines of greater than 80% over the last three generations), with one additional species assessed as Critically Endangered (Possibly Extinct) and one as Near Threatened (with declines of 20–30% over the last three generations).

The primary threats to these species are considered to be unsustainable and unregulated fishing throughout their range. Their retention in catches appears to be driven by the value of their fins on the international market, with fishing pressure being intense across much of their range. Their dependence on inshore habitats makes them susceptible to habitat damage and loss due to anthropogenic impacts. Where shark protective nets occur near beaches, some species get tangled. In South Africa species are sought by sports anglers because of their fighting ability, although they appear to be released live afterwards.

Very little species-specific catch or trade data are available. Information is often reported using generic terms such as "wedgefish" or "rhinobatids". Rhinidae are also known to be caught and reported alongside Glaucostegidae (Guitarfish) species using terms such as "guitarfish etc, nei". Localised landing declines have been reported, for example in India (87% for *Rhynchobatus djiddensis* and 86% for *Rhina ancylostoma* over a five-year period), Pakistan for "rhinobatids" and Indonesia for Wedgefish.

Detailed information is given below for two species known to be affected by trade:

- *Rhynchobatus australiae:* a recent IUCN Red List assessment categorised this species as Critically Endangered due to declines of >80% over the last three generations. Declines in this species have been inferred from landing data in Indonesia, India and Pakistan. The species' distribution may not be fully defined due to confusion with other members of the *Rhynchobatus djiddensis* species-complex.
- *Rhynchobatus djiddensis:* a recent IUCN Red List assessment also categorised this species as Critically Endangered due to declines of >80% over the last three generations. Declines in this species have been inferred from landing data in the Islamic Republic of Iran, Oman and United Arab Emirates, and fishing pressure is thought to be particularly high in east Africa where it is targeted alongside hammerhead sharks. As with *Rhynchobatus australiae*, the species' distribution may not be fully defined.

While it appears the meat from these species is primarily used domestically, the fins of these species are reported to be exported, and the factor driving the retention of these species when caught. Fins from species in the Rhinidae family have been identified in the "Qun chi" category of fins in Hong Kong Special Administrative Region (SAR) as having the highest value, and have also

been observed in markets in Singapore. Whole *Rhynchobatus* specimens were reported to be sold for USD 680 in the United Arab Emirates and Oman between 2010 and 2012.

The forms in which species are traded (fins, meat, skins) make it hard to differentiate between species without genetic analysis. There is conflicting information on whether Rhinidae fins are morphologically similar to those from the family Glaucostegidae (subject to a separate listing proposal, Prop. 43) and Pristidae (listed in Appendix I in 2007), once removed from the whole animals, particularly in the processed form.

Legislation and management for the Rhinidae is limited and varies by location and country. Where excluder devices have been used, *Rhynchobatus australiae* have been caught in lower numbers.

Analysis: Species of Rhinidae are found in coastal waters in the east Atlantic, Indian and western Pacific Oceans. The species are susceptible to many fishing gear types, and intensively utilized in their distributions. While meat appears to be utilized locally, fins from these species have been observed in international trade, in the highest value fin categories, which is presumed to drive the retention of the species as incidental catch. Localised declines have been reported across much of their ranges from landings data or catch rates, or inferred based on similar species and fishing pressure, in various locations.

In the most recent IUCN Red List assessments (to be published July 2019), eight of the species were said to have undergone declines of greater than 80% over the past three generations, and one species is considered possibly extinct (all nine are Critically Endangered), and therefore these species already meet the biological criteria for inclusion in Appendix I of a marked recent rate of decline. The final species (*Rhynchobatus palpebratus)* has undergone declines of 20–30% (Near Threatened) in the same time period. Over-harvest is identified as the main factor driving these declines. Therefore, it is likely that for all of the species in this family regulation of trade is required to ensure that harvest from the wild is not reducing populations to levels where survival might be threatened by continued harvest or other influences.

Due to the difficulties in differentiating the species in the form that they are traded in, particularly with the taxonomic confusion there is within this family, if any of the species are considered to meet the criteria then all other species in this family should be listed under look-alikes.

Other Considerations: Species of Rhinidae and Glaucostegidae (subject to a separate listing proposal, Prop. 43) are often landed and traded together. Therefore, if one of the proposals is accepted then the other family would meet the criteria in Annex 2bA of *Res. Conf. 9.24 (Rev. CoP17)*.

It is unclear for species in the Rhinidae family what proportion of animals survive if released after capture. It appears from limited information that initial survival after capture is high. In some instances, incidental catch levels are as high as target catches and therefore survival after potential release could be crucial in determining whether regulation is likely to have conservation benefits. In northern Australia, catches of large elasmobranchs have been reduced after the introduction of turtle exclusion devices (TEDs); *Rhynchobatus australiae* were caught in significantly lower numbers in nets with TEDs. The species in the family Rhinidae are subject to little or no management across their range; an Appendix-II listing could help support the improved management of these species in their range States.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS

As the taxonomy and with it the distributions of described species has changed in recent years (2016), information on species before this time may have related to previous understanding of the species and their distributions. We have not attempted to cross check to current taxonomy and distribution, and have presented data as reported.

Taxonomy
Wedgefish species in the family Rhinidae are shark-like batoid species possibly comprising ten species, in three genera (*Rhynchobatus*, *Rhynchorhina* and *Rhina)*. Trade information is often reported using generic terms such as "wedgefish" or "rhinobatids". Species of Rhinidae are also known to be caught and reported along Glaucostegidae (guitarfish) species using terms such as "guitarfish etc, nei".

Species-specific information has been difficult to collect, due to recent taxonomic revisions within the genus, and the difficulties identifying morphologically similar species. All Indo-West Pacific *Rhynchobatus* species were considered *Rhynchobatus djiddensis* prior to the late 1990s, when five separate species were either reinstated or newly described. In Australia, landings are reported as *Rhynchobatus* spp., comprised of a complex of three species: *Rhynchobatus australiae*, *R. laevis* and *R. palpebratus*. This makes assessing the threat to populations of each species a challenge.

Rhynchobatus
Rhynchobatus australiae
Rhynchotabus djiddensis
Rhynchobatus cooki
Rhynchobatus immaculatus
Rhynchobatus laevis
Rhynchobatus luebberti
Rhynchobatus palpebratus
Rhynchobatus springeri

Rhynchorhina
Rhynchorhina mauritaniensis

Rhina
Rhina ancylostoma

IUCN Global Category and Range
The species in the Rhinidae family have recently been reassessed for the IUCN Red List, with new assessments being accepted for publication in the July 2019 Red List update. Seven species have had their Red List category changed due to new datasets documenting global population reductions in Rhinidae and three species are assessed for the first time. Eight species have been assessed by IUCN as Critically Endangered (declines of >80% over the last three generations), with one assessed as Critically Endangered (Possibly Extinct) and one assessed as Near Threatened (declines of 20–30% over the last three generations).

White-spotted Guitarfish *Rhynchobatus australiae*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne et al., 2019a, accepted for publication in the July 2019 Red List update)

It is widespread in the Indo-West Pacific from Mozambique through the Western Indian Ocean, the Arabian Sea, Southeast Asia, and extending north to Taiwan Province of China, south to Australia (where it is wide-ranging across the north of the continent), and east to the Solomon Islands. The species' distribution may not be fully defined due to confusion with other members of the Rhynchobatus djiddensis *species-complex. It is found in the following FAO Fishing Areas: western and eastern Indian Ocean; northwest, southwest and western central Pacific. Range countries/territories are: Australia, Bahrain, Bangladesh, Brunei Darussalam, Cambodia, China, Djibouti, Egypt, Eritrea, India, Indonesia, Islamic Republic of Iran, Israel, Kenya, Kuwait, Malaysia, Maldives, Mozambique, Myanmar, Oman, Pakistan, Papua New Guinea; Philippines, Qatar, Saudi Arabia, Seychelles, Singapore, Solomon Islands, Somalia, Sri Lanka, Sudan, Taiwan Province of China, United Republic of Thailand, Timor-Leste, Viet Nam, Yemen (Kyne* et al.*, 2019a, accepted for publication in the July 2019 Red List update).*

Whitespotted Wedgefish *Rhynchobatus djiddensis*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne et al., *2019b, accepted for publication in the July 2019 Red List update)*

It is widespread in the Western Indian Ocean from South Africa to Oman and the Arabian/Persian Gulf, but it may not be present further east. However, the species' distribution may not be fully defined due to confusion with other members of the Rhynchobatus djiddensis *species-complex. It is found in the following FAO fishing areas: southeast Atlantic and western Indian Ocean. Range countries are: Bahrain, Djibouti, Egypt, Eritrea, Islamic Republic of Iran, Iraq, Israel, Kenya, Kuwait, Mozambique, Oman, Qatar, Saudi Arabia, Somalia,*

South Africa, Sudan, United Republic of Tanzania, United Arab Emirates and Yemen (Kyne et al.*, 2019b, accepted for publication in the July 2019 Red List update).*

Roughnose Wedgefish *Rhynchobatus cooki*
Critically Endangered (Possibly Extinct) A2bd (ver 3.1, assessed 2018) (Kyne et al.*, 2019c, accepted for publication in the July 2019 Red List update)*

It is known from specimens collected from fish markets in Singapore and Jakarta (Indonesia) in the Western Central Pacific. The exact provenance of these landings are unknown as fishers operate across the Indo-Malay Archipelago, so these may have been caught in areas such as the South China, Java or Andaman Seas. Its range occurs in the following FAO fishing area: western central Pacific (where it is thought possibly extinct). Range countries are: Indonesia, Indonesia (Java) and Singapore (Kyne et al.*, 2019c, accepted for publication in the July 2019 Red List update).*

Taiwanese Wedgefish *Rhynchobatus immaculatus*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne and Ebert, 2019, accepted for publication in the July 2019 Red List update)

It is known only from Taiwan Province of China in the north-west Pacific. It is found in the following FAO fishing area: northwest Pacific. Range is: Taiwan Province of China (Kyne and Ebert, 2019, accepted for publication in the July 2019 Red List update).

Smoothnose Wedgefish *Rhynchobatus laevis*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne and Jabado, 2019a, accepted for publication in the July 2019 Red List update)

It is widespread in the Indo-West Pacific; it was first described from India and has been widely confused with the Western Indian Ocean Rhynchobatus djiddensis *across its range from the Arabian Sea to the Western Pacific. It was previously thought to occur in East Africa, throughout South-east Asia, and across northern Australia, but it is now considered to occur in the Arabian Sea and the Bay of Bengal in the Indian Ocean, and off China and Japan in the Western Pacific. The species' distribution may not be fully defined due to confusion with other members of* the *R. djiddensis species-complex. It is found in the following FAO fishing areas: western and eastern Indian Ocean and northwest Pacific. Range countries/territories are: Bahrain, Bangladesh, China, India, Islamic Republic of Iran, Iraq, Kuwait, Oman, Pakistan, Qatar, Saudi Arabia, Sri Lanka, Taiwan Province of China and United Arab Emirates (Kyne and Jabado, 2019a, accepted for publication in the July 2019 Red List update).*

Lubbert's Guitarfish *Rhynchobatus luebberti*
Critically Endangered A2d (ver 3.1, assessed 2018) (Kyne and Jabado, 2019b, accepted for publication in the July 2019 Red List update)

It occurs in the eastern Atlantic from Mauritania to the Democratic Republic of the Congo and Angola. It is found in the following FAO fishing areas: southeast Atlantic and eastern central Atlantic. Range countries are: Angola, Benin, Cameroon, Democratic Republic of the Congo, Côte d'Ivoire, Equatorial Guinea, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Liberia, Mauritania, Nigeria, São Tomé and Príncipe, Senegal, Sierra Leone and Togo (Kyne and Jabado, 2019b, accepted for publication in the July 2019 Red List update).

Eyebrow Wedgefish *Rhynchobatus palpebratus*
Near Threatened A2d (ver 3.1, assessed 2018) (Kyne and Rigby, 2019, accepted for publication in the July 2019 Red List update)

The distribution is poorly-defined due to confusion with other members of the *Rhynchobatus djiddensis species-complex. Last* et al. *(2016) show it occurring only across northern Australia and southern Papua New Guinea, but the species description includes a single record from the Andaman Sea off Thailand. Additionally, two specimens have been reported from Taiwan Province of China. Together, these records suggest a more widespread occurrence in the Eastern Indian-Western Pacific region. It is found in the following FAO fishing areas: eastern Indian Ocean, western central and northwest Pacific. Range countries/territories are: Australia, Indonesia, Papua New Guinea, Taiwan Province of China and Thailand (Kyne and Rigby, 2019, accepted for publication in the July 2019 Red List update).*

Broadnose Wedgefish *Rhynchobatus springeri*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne, 2019, accepted for publication in the July 2019 Red List update)

This species occurs in the Indo-Malay Archipelago in the Eastern Indian and Western Central Pacific including the islands of Java, Sumatra, and Borneo as well as peninsular Thailand and Malaysia, the Gulf of Thailand, and the Philippines. The species' distribution may not be fully defined due to confusion with other members of the Rhynchobatus djiddensis *species-complex. It is found in the following FAO fishing areas: eastern Indian Ocean; western central Pacific. Range countries are: Cambodia, Indonesia, Indonesia (Java), Indonesia (Kalimantan), Indonesia (Sumatera), Malaysia, Philippines, Singapore, Thailand and Viet Nam (Kyne, 2019, accepted for publication in the July 2019 Red List update).*

False Shark Ray *Rhynchorhina mauritaniensis*
Critically Endangered A2d (ver 3.1, assessed 2018) (Kyne and Jabado, 2019c, accepted for publication in the July 2019 Red List update)

It is currently only known from a single location, the Banc d'Arguin of Mauritania in the eastern central Atlantic. The species is very poorly-known and its full range is not defined, although current records suggest that it is likely to be very range restricted. It is found in the following FAO fishing area: eastern central Atlantic. Range country is: Mauritania (Kyne and Jabado, 2019c, accepted for publication in the July 2019 Red List update).

Shark Ray *Rhina ancylostoma*
Critically Endangered A2bd (ver 3.1, assessed 2018) (Kyne et al.*, 2019d, accepted for publication in the July 2019 Red List update)*

It is widespread in the Indo-West Pacific from South Africa through the Western Indian Ocean, the Arabian Sea, South-east Asia, and extending north to Japan, south to Australia (where it is wide-ranging across the north of the continent), and east to New Caledonia. It is found in the following FAO fishing areas: western and eastern Indian Ocean, northwest, western central and southwest Pacific. Range countries/territories are: Australia, Bahrain, Bangladesh, Brunei Darussalam, Cambodia, China, Djibouti, Egypt, Eritrea, India, Indonesia, Islamic Republic of Iran, Japan, Kenya, Democratic People's Republic of Korea, Republic of Korea, Republic of Kuwait, Madagascar, Malaysia, Maldives, Mauritius, Mozambique, Myanmar, New Caledonia, Oman, Pakistan, Papua New Guinea, Philippines, Qatar, Réunion, Saudi Arabia, Seychelles, Singapore, Somalia, South Africa, Sri Lanka, Sudan, Taiwan Province of China, United Republic of Tanzania, Thailand, Timor-Leste, United Arab Emirates, Viet Nam and Yemen (Kyne et al.*, 2019d, accepted for publication in the July 2019 Red List update).*

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

Biology

Species in the family Rhinidae occur in shallow, inshore continental waters, often in muddy enclosed bays, in estuaries, off river mouths, and on coral reefs, but do not penetrate fresh water to any extent. They occur in depths from the intertidal down to at least 64 m and off large continental islands. They are slow but strong swimming bottom-dwellers, resting on soft mud, sandy or rough bottoms, but also swimming just above it or well off it near the surface (Compagno and Last, 1999).

All species are ovoviviparous as far as is known, with foetuses having large yolk sacks. They feed on benthic invertebrates (Compagno and Last, 1999). Estimated generation lengths vary from 10–15 years (Kyne et al.*, 2019e).*

Rhinidae *are considered to be low to medium productivity species (FAO, 2019).*

The biology for the species with most information is reported below:

Rhynchobatus australiae
Total length: 280 cm
Litter sizes: 7–19 embryos (mean=14)

Rhynchobatus djiddensis
Total length: 310 cm
Litter size= 4 pups/litter
Size at maturity: 150 cm TL (van der Elst, 1988)
Size at birth: 60 cm TL (van der Elst, 1988)

Population size
Data are not available to determine the precise population size of any species in the family Rhinidae.

Population trends
The IUCN Red List reports current population trends for all Rhinidae species as declining. Eight species have been assessed by IUCN as having declines of >80% over the last three generations, with one assessed as Critically Endangered (Possibly Extinct) and one with declines of 20–30% over the last three generations.

Southeast Asia and Oceania
Data from Indonesia indicates significant declines in catch rates in the target gillnet fishery for rhinids and rhynchobatids, of which *Rhynchobatus australiae* is a key part, indicating local population declines. Given its susceptibility to multiple gear types and evidence of local population declines, it is likely populations of *R. australiae* have been locally reduced throughout its range.

Landings data for "whitespotted wedgefishes" are available from Indonesia for 2005–2014 (10 years) (DGCF, 2015). This grouping likely includes Rhina ancylostoma, R. australiae, R. cooki, R. palpebratus *and* R. springeri *(this grouping may also include species in the Glaucostegidae family, but trends are likely to be representative). Landings declined by 74% over this period, which is the equivalent of 98–99% population reductions over the last three generations of wedgefishes and giant guitarfishes (Kyne* et al., *2019e).*

The Aru Islands (Indonesia) rhinid and rhynchobatid gill net fishery first began in the mid-1970s and rapidly expanded to reach a peak in 1987, with more than 500 boats involved. In subsequent years the catches declined very rapidly with only 100 boats fishing in this area in 1996. *The decline in catch rates and number of boats is based on a personal communication with F. Amir in the original text.* A similar fishery also exists in Merauke (south Papua, Indonesia) with gillnet boats operating in the Arafura Sea, close to Australian waters, and the frozen catch sent by boat to processing areas in Jakarta *(Blaber* et al.*, 2009)*.

A study of Australia's Northern Prawn Fishery (NPF) assessed its sustainability to elasmobranch incidental catch finding that Rhynchobatus djiddensis *and* Rhina ancylostoma *fishing mortality rate were well below the maximum sustainable mortality rate suggesting the population was in a good state (Zhou and Griffiths, 2008). C. Simpfendorfer* in litt., *2019 report that the assessment was for* R. djiddensis*, meaning it was likely a complex of* R. australiae, R. palpebratus *and* R. laevis.

Southern Asia
Significant declines of guitarfish, around 86% over a five-year timeframe have been documented on the west coast of India at a landing site in Tamil Nadu, despite increasing fishing effort *(Mohanraj* et al.*, 2009)*. This is likely to include *Rhynchobatus australiae* as a significant component of the catch, and possibly *R. djiddensis* depending on its full range. *The original paper reports* Rhynchobatus djiddensis *and* Rhina ancylostoma*, with declines of 87% and 86% respectively over the five-year period from trawl net fishing (Mohanraj* et al.*, 2009). It should however be noted that this data is from one landing site only in Chennai and care needs to be taken extrapolating when additional data are not available. However, Jabado* in litt., *2019 notes that fishing pressure is intense and increasing along the whole coast of India.*

In Pakistan data on wedgefish and guitarfishes (which includes all the family Rhinidae present in their waters—including *Rhynchobatus australiae, reported as "Guitarfishes, etc. nei"*) shows significant declines over one generation—from over 2,018 t *(reported as 2,185 t)* landed in the year 2000, to 403 t in the year 2011. *However, from 2014, 2015 and 2016 reported landings were 2,136, 2,266 and 1098 t respectively.*

Landings data for the "rhinobatid" category are available from Pakistan for 1993–2011 (19 years) covering the country's two coastal provinces (M. Gore unpublished data). This grouping likely includes Rhina ancylostoma*,* Rhynchobatus australiae*, and* Rhynchobatus laevis*. Data from the Sindh province showed a 72% decrease, from peak landings in 1999 to a low in 2011, and data from Balochistan province showed an 81% decrease from landings in 1994 to a low in 2011. These decreases are the equivalent of 94-99% population reductions over the last three generations of wegdefishes and giant guitarfishes (Kyne* et al., *2019e).*

Catch data for myliobatoid rays (this includes a variety of demersal rays but does not include rhinopristoids) which can be used to infer declines in wedgefishes and guitarfishes given overlapping distributions, habitat and susceptibility to capture in the same fishing gear are available from Maharashtra, western India for 1990-2004 (15 years) (Raje and Zacharia, 2009). The catch rate declined by 63% over this period (while fishing effort doubled), which is the equivalent of 86-95% population reductions over the last three generations of wedgefishes and giant guitarfishes (Kyne et al.*, 2019e).*

It is noted that wedgefishes and guitarfishes used to be quite abundant in commercial landings along the coast of Pakistan, however, catches of these species have substantially decreased, with almost all wedgefish having now disappeared from landings and rarely seen. *There is no quantitative data within this document to support this statement.*

North-western Indian Ocean
"Giant guitarfish" (a grouping which likely includes Rhina ancylostoma*,* Rhynchobatus australiae*,* Rhynchobatus djiddensis*,* Rhynchobatus laevis*) landings data are available from Iran for 1997–2016 (FAO, 2018). Landings declined by 66% over this period, which is the equivalent of 81-99% population reductions over the last three generations of wedgefishes and guitarfishes (30-45 years) (Kyne* et al., *2019e).*

Historical and current fishing pressures have driven declines in abundance of *Rhynchobatus australiae* and *R. djiddensis* in fisheries in the United Arab Emirates and Oman. *Rhynchobatus* species made up 56% of the landings of rhinopristoids, with *Rhynchobatus australiae* and *Rhynchobatus djiddensis* a significant component of that catch *(Rhynchobatus laevis were also included in these landings) between 2010-2012*. Individual specimens were reported as selling for as much as AED2500 (USD680), with landings noted to have declined in a short timeframe of less than 10 years, despite increasing fishing effort.

A study in 2017 concluded that all species of wedgefish have declined in the Arabian Sea and adjacent waters, and that populations of *Rhynchobatus australiae* and *Rhynchobatus djiddensis* (along with a sympatric species *Rhynchobatus laevis*) have suffered declines estimated between 50-80% over the last three decades. These species are now assessed as Endangered in this region due to intensive fishing pressure, that is likely to continue into the future and drive additional declines.

Eastern Africa
Although high quality data from the region are lacking, fisher, trader and local community survey work suggest declining population trends attributed to fisheries targeting both *Rhynchobatus australiae* and *Rhynchobatus djiddensis* for their high value fins.

Fishing pressure in East Africa is noted to be particularly high for *Rhynchobatus djiddensis*, where it is targeted alongside hammerhead sharks, due to the high value of their fins in export markets. Reports from artisanal fisheries in Mozambique indicate that fishing pressure has had a significant impact on local populations; many specimens observed by locally-based marine scientists that were caught were mature females, and numbers are reduced to very low levels on reefs where they had been abundant before long-line fisheries began locally in the early 2000's.

A survey of fishers and traders in Zanzibar, Tanzania, noted that *shark like rays* (potentially both *Rhynchobatus australiae* and *Rhynchobatus djiddensis*) are of particular interest due to the high value of their fins—noted as among the highest value by fin traders surveyed. Fishermen reported that they catch this species in high numbers; however, its numbers are declining, and it is now considered by some a rare species—one fisher stating that this was because so many people were catching it for its fins. *Fishers perceived changes in elasmobranch catch as the result of a range of factors such as overfishing, with an increased number of vessels and more efficient gear. Changes in value were perceived due to combined factors of lower catch and increased demand as well as a higher cost of living. However, some of the reasons cited by different fishers were contradictory, for example, some thought the demand for elasmobranch products had increased, whilst others thought it had decreased along with the sale value. This could result from the large variation in the number of years fishers had been fishing and requires further investigation. Some fishers commented that larger sharks had decreased in number whilst smaller sharks had remained the same. This may be a result of the typically greater vulnerability of larger elasmobranch species and corroborates the likely overexploitation and partial collapse of Zanzibar's elasmobranch fishery.*

In Madagascar (not shown as a range State in current referenceable range maps, but clearly landing the species), a premium price of up to MGA 400,000 (USD 204)/kg was paid for "tandraly" or giant guitarfish (Rhinidae spp. – likely *Rhynchobatus australiae* and *Rhynchobatus djiddensis*) fins, due to their high quality ceratotrichia (filaments of elastic protein). Two local collectors also stated that this species was decreasing.

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

Market trade
Recent research has, for the first time, revealed the scale of guitarfish trade *(only species from the family Rhinidae and Glaucostegidae have been identified so far)*. It was hypothesised that the fin trade category "Qun

Chi" in the Hong Kong, SAR shark fin market referred to this group. This was recently confirmed by a study which used DNA barcoding (n=19 fins) to identify the presence of multiple species, including *Rhynchobatus australiae*, *Rhynchobatus djiddensis*, *Glaucostegus cemiculus*, and *Rhina ancylostoma* (*Bloom, 2018*). A genetic analysis was undertaken of processed fin trimmings (by-products from preparing imported fins for consumption) purchased in Hong Kong SAR, the global hub of the shark fin trade, in 2014. This revealed the presence of multiple species in the family Rhinidae: *Rhynchobatus australiae*, *Rhynchobatus cf. laevis*, and *Rhynchobatus djiddensis*. Collectively, these made up at least 0.1% of the trimmings collected, ranking them in the top 20 most common among 86 species or species groups recorded in the fin trade. *Blue Shark* Prionace glauca *and Silky Shark* Carcharhinus falciformis *made up 34 and 10% of samples respectively, hence other species making up a smaller proportion.*

A recent study in Singapore, the world's secondary shark fin trade hub after Hong Kong SAR, collected 207 samples of shark and ray products that were on sale to the general public. Of the 106 products labelled as 'shark', 17% *(18% reported in paper)* were identified as *Rhynchobatus australiae*, with the species noted as being highly prized for its fins. This was the largest percentage of any 'shark' species found within the Singapore trade in the study.

The global trade demand for their high value fins drives wedgefish mortality in many fisheries and represents the principle threat to *Rhynchobatus australiae* and *Rhynchobatus djiddensis*, wherever they are found. *There is clear evidence that trade in* Rhynchobatus *fins is the major driver in the retention of these species by fishers, and hence declines experienced (Simpfendorfer in litt., 2019).*

North western Indian Ocean
Interviews with fishermen and traders in the United Arab Emirates (UAE) suggests that guitarfishes have replaced sawfish as the most sought-after species for the international fin trade market and are increasingly targeted and retained due to the high value of their fins.

Based on observations during field surveys in the United Arab Emirates, rhinopristoids were always landed and auctioned whole (Jabado, 2018). Fins were then removed at the auction sites and sold separately to the highest bidders. The meat of larger species, such as rhinids, was either cut into fresh fillets and sold for local consumption, or the body was sold whole and sent for further processing (meat usually cut into small square pieces, salted, dried, and processed for the export market). The most valuable species were wedgefishes with specimens over 2000 mm TL each selling for AED 2500 (USD 680). Conversely, individuals of Rhina ancylostoma *were the least valuable and whole specimens of over 1500 mm TL each sold for AED 50 (USD 14).*

Southern Asia
Fishing pressure that will directly impact both species is intense and increasing in the region. The number of trawlers operating in Gujarat waters has increased from ~6,600 boats in 2004 to ~11,500 boats in 2010, and about 2,000 trawlers operate in Pakistan shelf waters.

Rhina ancylostoma *were encountered in processing centres in Bangladesh (Haque* et al.*, 2018). It is not clear what volume were encountered.*

Rhynchobatus australiae are heavily exploited in Southeast Asia for their fins, which are considered some of the most valuable in trade. Much of their range occurs in areas of high fishing pressure and they are susceptible to capture both as target and incidental catch by trawl, net and longline gear.

Rhynchobatus australiae *is one of the most sought after elasmobranchs in southeast Asia (particularly Indonesia), with the dorsal fins and upper caudal fin considered to be of premium quality and fetch the highest prices (Chen, 1996). A set of fins from a single individual have been reported to have fetched up to Rp 900,000 or USD 396/kg (Chen, 1996). The skins and flesh are also of good quality.*

Eastern Africa
In Zanzibar, Tanzania, the price range stated in interviews for shark or guitarfish fins was 10–35,000 TZS/kg (USD 9-31,500/kg, 1 TZS reported as USD 0.9 in paper, however currency converter reports 1TZS= USD 0.0005 on 21/08/15, therefore giving a value of USD 0.005-17.5/kg) with Rhynchobatus australiae *fins considered the most valuable and highest quality for consumption.*

*A premium on price was given for "tandraly" or Giant Guitarfish (*Rhynchobatus djiddensis*) fins in Madagascar, with a kilogram of fins reaching MGA 400 000 (USD 204), as they contained high quality cercotrichia (Hopkins, 2011). Two local collectors also stated that this species was decreasing.*

In Mozambique a spear-fisher reported that fins are worth between 6-7000 MT/kg (it is not clear when this was reported, value is converted to USD based on conversion rate on 01/01/08, 1 MT=USD 0.04, therefore USD 0.24-280/kg), with sharks and guitarfish being the target species (Pierce et al.*, 2008)). Fins are apparently sold-on to Chinese buyers in Maputo.*

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

Rhynchobatus spp*. in the Arabian Sea and adjacent waters appear to be comprised of several species including the* Rhynchobatus djiddensis*,* Rhynchobatus laevis *and* Rhynchobatus australiae*. These have been documented from the Arabian Sea, Sea of Oman, and the Gulf, and are consistently confused in the literature and require further taxonomic work (Last* et al.*, 2016). Within this complex, species identification relies on relatively small differences in the number and pattern of white and black spots around the spiracles and gills, variations in fin size ratios, and the absence or presence of dark bars between the eyes (Last* et al.*, 2016; White* et al.*, 2014).*

The fins from *Rhynchobatus australiae* are distinctive, and dried, unprocessed (skin on) Rhinopristiform fins can be visually identified at least to the family level (i.e., Rhinidae, Glaucostegidae, Pristidae – the three families most regularly found in trade). This ability to visually identify the primary product in trade will aid in the implementation and enforcement of this proposed listing.

Intra-species variability in fin morphology within the family Rhinidae makes identification to a species level challenging. For example, the highly prized dorsal fins are morphologically similar (in size, shape and colour) for many of the species within the family Rhinidae.

Fins from the family Rhinidae are morphologically similar to those from the family Glaucostegidae (giant guitarfishes) and Pristidae (sawfishes), once removed from the whole animals. Although, as noted above, visual identification to the family level for unprocessed fins is possible, it makes identification of fins between these three groups that have been heavily processed (skin removed) challenging. *Jabado in litt., 2019 report that the fins from the family Rhinidae are not morphologically similar to those from the family Glaucostegidae, they are similar to each other in the family - but Rhina and Rhynchorhina are distinct (dark, spotted) - and the fins of the remaining Rhynchobatus are different from the giant guitarfishes (Glaucostegidae) because of the enlarged denticles present on those (vs lacking on Rhinidae) for the dorsal fins - and for the caudals (which are usually sold whole) Rhynchobatus have a distinct upper and lower caudal lobe whereas Glaucostegidae species don't.*

Genetic techniques can address fin identification challenges. With Sawfishes already listed at the family level on CITES Appendix I, monitoring, enforcement and compliance measures should already be in place for fins originating from the Order Rhinopristiformes.

Simpfendorfer in litt., 2019 report that there is morphological similarity between the species in Rhinidae. It is especially true in the form that they are traded in (fins, skins and meat) that can only be delineated with confidence using genetic techniques (Simpfendorfer in litt., 2019).

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

The SS states that given these species' large size, restricted habitat use, high fin value in international markets, and the *current and increasing fishing pressure* throughout their range, *Rhynchobatus australiae* and *Rhynchobatus djiddensis* are believed to be at a particularly high risk of a similar fate to the sawfishes (family Pristidae), that have been extirpated from almost all of their historic range and were consequently listed on CITES Appendix I.

Additional Information

Threats

The primary threat to these species is unsustainable and unregulated fisheries mortality throughout their range. Both *Rhynchobatus australiae* and *Rhynchobatus djiddensis* are caught by artisanal and commercial fisheries both as a target species and as incidental catch in demersal trawl, net, and longline fisheries – with retention incentivized due to the very high value of their fins in international trade. Their use of inshore habitat and susceptibility to multiple gear types makes them particularly vulnerable, and that is compounded as their range includes some of the world's most heavily fished coastal regions.

Their dependence on inshore habitats makes them highly susceptible to habitat loss and degradation. The inshore habitats used by species in the family Rhinidae, such as seagrass and coral reef ecosystems, are suffering significant reductions globally due to anthropogenic impacts. This additional threat only heightens the concern for these species' survival.

Rhynchobatus djiddensis *have been observed to be caught in protective shark nets in the KZN, South Africa (Dudley and Cavanagh, 2006). Between 1981-2000, 118 individuals were caught, of which 74% were released alive (Young, 2001). Survival of released individuals is unknown.*

It is a popular target species for shore anglers because of its fighting ability, although it is common practice to release the species. Catch rate (number of fish per angler hour) in competition shore angling in KZN showed an increase in the period 1977-2000, as did the mean mass of animal caught (Pradervand, 2003). By contrast anecdotal reports from anglers indicate that the species is less commonly caught than in previous years (Mann, 2003).

Conservation, management and legislation

Rhynchobatus australiae and *Rhynchobatus djiddensis* are subject to limited management globally, and with their inshore range are subject to the national laws of countries throughout their range, rather than those of regional fisheries bodies and agreements. It is assumed that the vast majority of international trade in their fins and other products is legal, but from widely unregulated fisheries. Where shark finning is banned, but still occurs these could be species illegally finned due to the exceptionally high value of their fins when traded internationally, and the comparatively low value of their meat.

Rhynchobatus djiddensis is listed on Schedule 1 of the Bangladesh Wildlife Protection Act, 2012, and on India's Wildlife Protection Act 1972, which prohibits the hunting, trade and any other form of exploitation of these species. Pakistan have species-specific regulations banning the catching of rhinids and rhynchobatids, specifically the Department of Fisheries of two of Pakistan's maritime provinces, Sindh and Balochistan, have banned the catching, landing and marketing of all guitarfish, and a complex of wedgefishes including *Rhynchobatus australiae* is managed in a mixed fishery in Australia.

Apart from these limited examples of management, species in the family Rhinidae are subject to little or no management across their range, including in hotspots of inshore fishing pressure where significant declines in their populations have been noted, such as Eastern Africa and the north-western Indian Ocean.

In the Australian state of Queensland Rhynchobatus spp. *have a trip limit of 5 for commercial fishers (Simpfendorfer in litt., 2019, White* et al.*, 2013).*

Rhynchobatus australiae is listed on CMS Appendix II.

Rhynchobatus australiae*,* Rhynchobatus djiddensis *and* Rhynchobatus laevis *are also listed on the Shark MoU as of December 2018 (Jabado in litt., 2019).*

The recreational line fishery in South Africa is managed by a bag limit of one/species/person/day for unspecified chondrichthyans, which includes Rhynchobatus djiddensis *(Dudley and Cavanagh, 2006).*

A study in the Northern Prawn Fishery has found that the use of turtle exclusion devices (TEDs) have resulted in significantly lower proportions of Rhynchobatus australiae *being caught in nets with TEDs than without (M. Campbell, in litt., 2019).*

Other comments

As species in the Rhinidae family are often being caught in multispecies net fisheries, they are likely to continue being caught. The species survival probability after capture is therefore important when determining whether implementation will have conservation benefits. Rhynchobatus djiddensis *is taken as incidental catch by demersal prawn trawlers in central KZN, primarily in the summer and at a rate of 123 to 231 per year (Fennessy, 1994). Most (82%) of those caught in a sample of 100 trawls were alive and released, although subsequent survival is not known.*

Rhynchobatus djiddensis *(probably* Rhynchobatus australiae*) was found to be one of the four most commonly caught elasmobranchs in the incidental catch of the trawl fisheries (prawn and fish) in northern Australia, with approximately 10% dying in the trawl net (Stobutzki* et al.*, 2002). Survival increased with the length of the individual (Stobutzki* et al.*, 2002). Since the introduction of Turtle Exclusion Devices (TEDS) in some northern*

Australian trawl fisheries, catches of large elasmobranchs have been reduced (Brewer et al.*, 1998) and thus* Rhynchobatus australiae *are probably caught in lower numbers.*

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Inclusion of the following three species belonging to the subgenus *Holothuria (Microthele)*: *Holothuria (Microthele) fuscogilva*, *Holothuria (Microthele) nobilis* and *Holothuria (Microthele) whitmaei* in Appendix II

Proponents: European Union, Kenya, Senegal, Seychelles and United States of America

Summary: The Class Holothuroidea, commonly referred to as sea cucumbers or beche-de-mer in its dried, traded form (a delicacy prepared from the dried body wall thought to have medicinal properties), contains 1,743 species. The genus *Holothuria* contains more than 20 subgenera, of which the subgenus *Holothuria (Microthele)* contains four species. Three species within the subgenus are commonly referred to as teatfish due to their lateral protrusions, whilst the fourth species *Holothuria fuscopunctata* is known as a trunkfish and lacks teats. The presence of teats (in the three species which display them) differentiate this group from other sea cucumbers, even in dried form, and it is only these three species subject to this listing proposal:

Holothuria fuscogilva: Varies in colour from a dark colour with light spots to a light colour with dark spots, with large lateral protrusions (teats) along its flanks. Length and weight vary from 28–57 cm and 2.4–3 kg based on location. This species can be found on reef slopes, sandy areas and seagrass beds between 0–50 m in depth. This species occurs throughout the Indian and Pacific Oceans.

Holothuria nobilis: Black in colour with white blotches on the sides of the animal and 6–10 large lateral protrusions (teats) along its flanks. Length varies from 14–60 cm and weight from 0.23–3 kg based on location. This species can be found on shallow coral reef habitats, seagrass beds and sandy substrates between 0–40 m in depth. This species only occurs in the African and Indian Ocean region. *Holothuria nobilis* also includes a yet undescribed species which is likely to be separated from *H. nobilis* named *Holothuria (Microthele) sp. "pentard"*. This yet undescribed species has an average length of 30 cm and weight of 1.7 kg and has been found to prefer sandy substrates at a depth between 10–50 m.

Holothuria whitmaei: Uniformly black dorsally and grey ventrally with 5–10 large lateral protrusions (teats) along its flanks. Length of live specimens varies from 23–54 cm and weight is an average of 1.8 kg. This species inhabits shallow waters between 0–20 m and is found in coral reef flats and slopes and sandy seagrass beds. This species is only found in the Pacific Ocean.

Very little is known about generation length and recruitment in Holothurians, but the IUCN Red List assessments suggest that species within this subgenus could live from up to 12 years to several decades.

Trends in populations are derived from density estimates, but due to the large ranges of the three species proposed, very little evidence of overall population trends exists. The FAO Expert Advisory Panel noted that there is a general negative trend in populations in all three species throughout their ranges and that many populations have lower densities than the recommended threshold density (10 per ha) for healthy populations. From surveys in specific sites:

- densities of *Holothuria nobilis* in Sri Lanka were less than one individual per hectare (2010).
- in Zanzibar, *H. nobilis* was not located outside of protected areas, and inside protected areas was found at densities of 1.2 individuals per hectare (2010).
- declines in densities in previously fished areas of 80% over five years in Australia (1998–2005) and 83% over 16 years in Egypt (2000 – 2016) were observed for *H. whitmaei*.
- in the same studies, *H. fuscogilva* densities decreased by 86% over five years in Australia (1998–2005) and 94% over 16 years in Egypt (2000–2016).

Sea cucumbers from the family Holothuridae have been harvested in the Indo-Pacific region for over 1,000 years. In the 1980s harvesting of sea cucumbers increased to feed the demand for beche-de-mer in Asian markets. Trade data for individual species are rarely available as trade is often reported

using a generic "sea cucumber" name. Annual global capture of sea cucumbers showed a six-fold increase in the 1980s and from 1990 onwards has been steadily increasing to a weight of 31,000 t in 2016.

One species-specific example of trade comes from the Seychelles, where capture data between 2001 and 2016 totalled 1,700 t of *Holothuria fuscogilva* and 180 t of *H. nobilis*. Complementary density data showed a 54% decline in *H. fuscogilva* density and a 73% decline in *H. nobilis* density between the two survey periods in 2003–2004 and 2011–2013.

Holothuria fuscogilva was assessed as Vulnerable on the IUCN Red List in 2010; the population is estimated to have declined by 30–50% since the 1960s. Both *H. nobilis* and *H. whitmaei* were assessed as Endangered on the IUCN Red List (2010) with declines since the 1960s estimated at 60–70% in at least 80% of its range for *H. nobilis* and 60–90% in most of its range for *H. whitmaei.*

Sea cucumber fisheries are not regulated in several countries, although some have employed various measures. Australia and Egypt both employed closed fishing areas and India employed a total ban on sea cucumber fishing, but these are not widely adopted strategies and there have been issues with implementation. Limited-access fisheries have also restricted the number of vessels/harvesters in a given area. Total Allowable Catches (TACs) or quotas have also been established in Australia and Papua New Guinea. Minimum catch sizes are implemented in Australia, Papua New Guinea, Fiji and Tonga. However, large areas of these species' ranges are not protected or regulated.

Analysis: *Holothuria fuscogilva, H. nobilis* and *H. whitmaei* are the target of fisheries driven by the international trade of beche-de-mer mainly to Asian markets. Information on the productivity and recruitment of all three species is relatively unknown, but threshold densities are thought to be required to ensure successful reproduction.

Although very little information on species-specific trade is available, all three species have been observed in markets. The only specific data on the impact of fishing, over a period of 11 years in the Seychelles, showed declines in density of 54% for *Holothuria fuscogilva* and 73% for *H. nobilis.* The FAO Expert Advisory Panel Report noted that both historic and recent declines have been observed in the densities of all three species which are consistent with the indicative guidelines for inclusion in Appendix II of commercially exploited aquatic species suggested in the footnote to Annex 5 of *Resolution 9.24 (Rev. CoP17)*. These declines are from studies in limited areas of the ranges of these three species, however fishing pressure is likely over much of their range. The IUCN Red List assessments have estimated the overall declines as: *H. fuscogilva* 30–50% since the 1960s, *H. nobilis* 60–70% in at least 80% of its range, and *H. whitmaei* 60–90% in the majority of its range. Many of the densities considered in the FAO Panel's report were below the "rule of thumb" threshold for healthy breeding populations.

As international trade is likely to be driving the majority of fishing for these species, it would appear that regulation of trade is required to ensure that harvest from the wild is not reducing population to a level where survival might be threatened by continued harvest or other influences.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

The Class Holothuroidea, commonly referred to as sea cucumbers (or beche-de-mer in its dried, traded form) contains 1,743 species, while the genus *Holothuria* contains more than 20 subgenera, of which the subgenus *Holothuria (Microthele)* contains four species. Three species within the subgenus are commonly referred to as teatfish due to their lateral protrusions whilst the fourth species *Holothuria fuscopunctata* is known as a trunkfish and lacks teats.

Holothuria (Microthele) fuscogilva
Holothuria (Microthele) nobilis currently includes what is likely to become *Holothuria (Microthele) sp. "pentard"*
Holothuria (Microthele) whitmaei

Holothuria (Microthele) *nobilis* taxa seems to be considered as a group of species where Holothuria sp. "pentard" is a form that is currently being described.

Range
Holothuria (Microthele) fuscogilva: Madagascar, Easter Island, from South China to Lord Howe Island, French Polynesia, Réunion, New Caledonia, Scattered Islands, Wallis and Futuna, Mayotte, Solomon Islands, Australia, India, Zanzibar, Tanzania, Madagascar, Philippines, Kiribati, Tonga, Fiji, Papua New Guinea, Sri Lanka, Indonesia, Cook Islands, Egypt, Vanuatu, Kenya, Somalia, Soudan, Eritrea, Yemen, Saudi Arabia, Hawaii (United States of America), Viet Nam, Seychelles, Malaysia, Singapore, Guam, Micronesia, Jordan, Comoros, Djibouti, Samoa, American Samoa, Brunei Darussalam, Christmas Island, Cocos Islands (Keeling), Mozambique, Tuvalu, Marshall Island, small islands away from the United States of America, Northern Mariana Islands, Oman, Palau, Timor-Leste, Norfolk Island, Tokelau, Pitcairn Islands, Nauru, Niue.

Holothuria (Microthele) nobilis: India, Maldives, Mayotte, Reunion, Kenya, Zanzibar, Tanzania, Egypt, Madagascar, Eritrea, Mauritius, Sri Lanka, Seychelles, Mozambique, Sudan, Yemen, Somalia, Israel, Comoros, Jordan, Djibouti *(Holothuria (Microthele) sp. "pentard"*: Comoros, Nosy Be Island (Madagascar), Seychelles, Zanzibar (Tanzania), Maldives, Sri Lanka.

Holothuria (Microthele) whitmaei: Australia, Hawaii, French Polynesia, South China to Lord Howe Island, 31° S (Australia), New Caledonia, French Polynesia, Wallis and Futuna, Kiribati, Viet Nam, Malaysia, Philippines, Tonga, Fiji, Papua New Guinea, Cook Islands, Indonesia, Vanuatu, Cambodia, Singapore, Thailand, Tuvalu, Hawaii (United States), Samoa, American Samoa, Guam, Micronesia, Northern Mariana Islands, Nauru, Niue, Tokelau.

IUCN Global Category
Holothuria (Microthele) fuscogilva: Vulnerable A2bd (2013) ver. 3.1
Holothuria (Microthele) nobilis: Endangered A2bd (2013) ver. 3.1
Holothuria (Microthele) whitmaei: Endangered A2bd (2013) ver. 3.1

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I
Population size and trends
The mean density of each teatfish was calculated in several studies:
– For the species *Holothuria (Microthele) nobilis*, the mean density varies from 0.12 to 10 individuals per ha.
– For the species *Holothuria (Microthele) fuscogilva*, the density of its populations does not exceed 40 individuals per ha.
– For the species *Holothuria (Microthele) whitmaei*, density does not exceed 12 individuals per ha in the Pacific (and is much lower in other locations).
– In 2008, the density of *H. fuscogilva*, *H. nobilis*, and *H. "pentard"* was calculated in two regions of Sri Lanka: less than one individual per hectare for all of these species.

The size of teatfish populations may vary by location, by year and by sampling method; biases may therefore appear.

The FAO Expert Advisory Panel described a "rule of thumb" threshold density of 10 ind. ha^{-1} which describes a baseline threshold for "healthy" teatfish stocks (FAO, 2019). The Expert Advisory Panel also performed an in-depth review of literature relating to the three proposed species in 16 countries within their range (up to 25 sites per species). Analysis from the Advisory panel show how populations at different sites for each species compare to the "rule of thumb" density:

Holothuria fuscogilva: The panel reviewed literature for 25 different sites and concluded that H. fuscogilva densities were above the threshold at three sites, at or below the threshold at 14 sites and that there was insufficient evidence at eight sites.
Holothuria nobilis: The panel reviewed literature for nine different sites and concluded that H. nobilis densities were above the threshold at one site, at or below the threshold at six sites and that there was insufficient evidence at two sites.

Holothuria whitmaei: The panel reviewed literature for 15 different sites and concluded that H. whitmaei densities were above the threshold at three sites, at or below the threshold at seven sites and that there was insufficient evidence at five sites.

Overall, teatfish populations are depleted or over-exploited in most range countries. According to an evaluation of the IUCN Red List published in 2013, populations of *Holothuria nobilis*, *H. whitmaei* and *H. fuscogilva* are declining.

Populations of Holothuria nobilis *have been estimated to have declined by between 60% and 70% in at least 80% of its range since the 1960s in the IUCN Red List assessment (Conand* et al., *2013a).*

Global declines of Holothuria fuscogilva *were estimated by the IUCN Red List to be between 30% and 50% since the 1960s* (Conand *et al.,* 2013).

Declines of between 60% and 90% have been estimated by the IUCN Red List for Holothuria whitmaei *since the 1960s (Conand* et al., *2013b). There is currently no IUCN Red List assessment for* H. "pentard" *as the species has not yet been described. It should be noted that the reported Red List assessment declines for the three species listed are estimates and should be treated with caution.*

The FAO Export Advisory Panel noted "a general negative trend in population measures across their range, and low or depleted abundances when compared to recognized baselines for "healthy" stocks" (FAO, 2019).

***Holothuria nobilis*:** In Madagascar, stocks are assumed to be depleted as very few specimens have been seen in recent years, particularly in areas that have been heavily exploited. In Egypt, this species has almost completely disappeared because of fishing. In Tanzania, this species previously comprised a very small percentage of the total of sea cucumber species but dominated the catch. In the Chagos Marine Protected Area, populations have also declined over the past 4 or 5 years due to illegal fishing. In the Seychelles catches for this species were relatively stable from 2003 to 2006, with a peak of 10,371 individuals and then decreased in 2007 and 2008 to 5,687 individuals. This species has also been depleted in Mozambique, India, Sri Lanka, the Red Sea, the Maldives and possibly Kenya due to overfishing. On the basis of these references, it is estimated that there has been at least 60% to 70% decline in more than 80% of its distribution.

The percentage decline noted above is also the figure given in the Red List assessment for Holothuria nobilis (Conand *et al.,* 2013a).

In Sri Lanka densities of Holothuria nobilis *in two survey sites were observed to be less than 1 ind. ha^{-1} (Dissanayake & Stefansson, 2010).*

A survey of four sites in Zanzibar located Holothuria nobilis *in one protected area at a density of 1.2 ind. ha^{-1}, in the three other areas (where collection is permitted),* H. nobilis *was absent* (Eriksson *et al.*, 2010).

The FAO expert panel studied the literature at nine different locations/countries looking at population trends for Holothuria nobilis*. They concluded that there was insufficient population trend data to determine if the Appendix II criteria had been met, but from snapshot density estimates showed that at six locations,* H. nobilis *densities fell below the "rule of thumb" density for a healthy population (FAO, 2019).*

Holothuria whitmaei: In Saipan, Northern Mariana Islands, a fishery targeted this species but stopped in 1997 due to a decline in the CPUE (Catch per unit effort). In the Marshall Islands and the Cook Islands, this species is rare. In Tonga, sea cucumber stocks are depleted. In Papua New Guinea, sea cucumber stocks are depleted, with low densities of commercial holothurians (21 per ha) and comparisons with historical catch data show that this species has been grossly over-exploited. In the Solomon Islands, the species was observed in low densities. This species has been over-exploited in the Torres Strait since the 1990s, and its population has decreased by 80% in the Great Barrier Reef in recent decades. In Ashmore Reef, populations of this species were considered to be severely depleted in 2000. In the Philippines, this species is considered as over-exploited in view of the decrease in the number of exports. The species is also over-exploited in Indonesia.

Data collection on Ashmore reef (Australia) has seen population densities of Holothuria whitmaei *decrease by 80% between the year 2000 (0.5 ind. ha^{-1}) and 2005 (0.1 ind. ha^{-1}) (Ceccarelli* et al., *2011). It should be noted that this area was designated as a Marine Protected Area in 1983 but is surrounded by areas which are heavily fished.*

In the Abou Ghosoun area of Egypt, Holothuria whitmaei *abundance has decreased 83% in 16 years between 2000 and 2016 (Hasan, 2019).*

In Tonga, Holothuria whitmaei *occurrence was observed to be decreasing between 1984 and 2004 and even after a seven-year moratorium on fishing, occurrence observations did not increase to original levels* (Friedman et al., *2011). Some caution should be taken with these results as a variety of data collection methods were used during the different time-periods of the survey. Between 2003 and 2009 in Fiji, both occurrence and density values decreased when Reef Benthos transect methods were used, no difference was observed in the two time periods when Manta surveys were performed (Friedman* et al., *2011). Further declines in density from the 2009 level in Friedman* et al., *(2011) have been reported for* H. whitmaei *in 2013, but these declines are on a much more local scale (Jupiter and Vave, 2013).*

The FAO expert panel looked at population trends at 15 different sites for Holothuria whitmaei*. They concluded that at three sites there had been a greater than 80% recent decline in population densities, three of the sites had seen no decreases in population density and at nine sites population trends could not be ascertained. At two of the sites (in Australia) where no declines were observed, sea cucumbers have been protected through a cessation of fishing of more than 17 years (FAO, 2019).*

***Holothuria fuscogilva*:** This species has been depleted in South-east Asia and parts of the South Pacific (about 30% of its distribution). It is considered mainly over-exploited in East Africa (40% of its distribution). Shallower waters are more severely affected. In the Cook Islands, *H. fuscogilva* is rare. In some countries of South-east Asia, populations of *H. fuscogilva* are considered severely depleted, as in Indonesia and the Philippines. In New Caledonia, the species is also considered as depleted.

Data collection on Ashmore reef (Australia) (Ceccarelli et al., *2011) showed that in 1998,* Holothuria fuscogilva *densities were 1.4 ind. ha^{-1}, but reduced to 0.2 ind. ha^{-1} between 2005 and 2006. This represents a decline of 86% in the densities of* H. fuscogilva *on Ashmore reef.*

In the Abou Ghosoun area of Egypt, Holothuria fuscogilva *abundance has decreased 94% in 16 years between 2000 and 2016 (Hasan, 2019).*

In Sri Lanka densities of Holothuria fuscogilva *in two survey sites were observed to be less than 1 ind. ha^{-1} (Dissanayake & Stefansson, 2010).*

Between 2003 and 2009 in Fiji, both occurrence and density values of Holothuria fuscogilva *decreased when Reef Benthos transect methods were used, no difference was observed in the two time periods when Manta surveys were performed (Friedman* et al., *2011). Further declines in density from the 2009 level in Friedman* et al., *(2011) have been reported for* H. fuscogilva *in 2013, but these declines are on a much more local scale (Jupiter & Vave, 2013). Declines in occurrences of* H. fuscogilva *were also recorded in Tonga between 1984 and 2004 (Friedman* et al., *2011). Some caution should be taken with these results as a variety of data collection methods were used during the different time-periods of the survey.*

The FAO expert panel reviewed population density data at 25 sites/countries for Holothuria fuscogilva*. It was concluded that population densities had recently declined by more than 80% at two sites/countries, two sites had seen density declines of less than 30% and one site had not seen a recent decline in population density. There was insufficient time series data to make a determination of a decline at 20 of the sites/countries, but of these 20, 14 sites/countries were found to have snapshot densities of at or below the 'rule of thumb' threshold for healthy populations (FAO, 2019).*

A visit to a holothurian processing plant, conducted at a workshop on the Indian Ocean Holothurian fisheries in Tanzania (Zanzibar), allowed the observation that teatfish (*Holothuria nobilis*, *H. fuscogilva* and *H.* "pentard") which have a high commercial value, accounted for only a tiny fraction of the large number of drying products. In addition, most individuals were small. This suggests that populations of highly valued species are decreasing both in size and total landings, which explains why catches increasingly concern low- and medium-value species.

The FAO Expert Advisory Panel found limited evidence of population increases, although one example from Australia presented data that suggest that it required 17 years for stocks of Holothuria whitmaei *to recover to pre-fishing levels. For further information on population recoveries, please refer to the FAO report from the Expert Advisory Panel (FAO, 2019).*

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

The main threat to teatfish populations is overfishing in order to meet the demand for beche-de-mer, teatfish are also used for biomedical research. Sea cucumbers are sedentary animals that are particularly vulnerable to over-exploitation because they are large in size, easy to collect due to their shallow area of occurrence, and do not require sophisticated fishing techniques. Strong fishing pressure causes a decrease in species biomass density and populations are unable to replenish once they have fallen below critical mass. To reproduce, teatfish release their gametes in water and the success of the fertilisation depends on the proximity of the individuals during the spawning period (and therefore on the density of the population). Due to the reduction in population density caused by fishing, individuals may be unable to reproduce successfully, the distance between males and females being too large.

Trade statistics rarely differentiate individual species, all sea cucumber species are often referred to as beche-de-mer. The legal trade is highly lucrative and an important source of income for many developed and developing countries; it is also one of the oldest forms of trade in the Pacific islands. It aims to meet the needs of Asian markets such as China, where it is mainly consumed as a delicacy. Most sea cucumbers are imported into Asia, mainly via Hong Kong Special Administrative Region (SAR), Singapore and Taipei, from which they are re-exported. The market is mainly in dried tropical cucumbers of all varieties, with a small amount of skinless and frozen sea cucumbers sent by air freight. In the Pacific region, the main producing countries are Papua New Guinea, Solomon Islands, Fiji and Australia, while in South Asia the main countries of production and/or export are Sri Lanka, Maldives and India. Traditional fishers from Indonesia, Papua New Guinea and New Zealand may also have access to marine resources within the Australian Fishing Zone or contiguous waters under bilateral arrangements.

The skinless imports suggested above are imported from New Zealand (Ferdouse, 2004) which is not a range State for any of the teatfish species. Therefore, if teatfish are not exported skinless, they would maintain the "teats" which make them identifiable.

According to FAO statistics a total of 317,263 t of sea cucumber commodities have been imported by 68 different countries/territories between 1990–2016 (which includes commodities which are: dried, salted, in brine, frozen, live, fresh, chilled, prepared or preserved). The top ten importers are shown in Table 1.

Table 1: *Top ten importers of sea cucumber (not species specific) commodities between 1990–2016* (FAO, 2019).

Country/territory	*Weight imported between 1990–2016 (tonnes)*
Hong Kong SAR	*155,346*
Taiwan Province of China	*25,984*
China	*22,174*
Singapore	*22,074*
Japan	*19,882*
Korea, Republic of	*17,290*
United States of America	*17,176*
Malaysia	*16,845*
Spain	*7,271*
Bulgaria	*3,578*

A total of 212,463 t of sea cucumber commodities have been exported by 71 different countries/territories between 1990–2016. The top ten exporting countries are shown in Table 2. It is unclear why there are such large discrepancies between reported figures for total imports and exports: these data should be treated with caution.

Table 2: *Top ten exporters of sea cucumber (not species specific) commodities between 1990–2016 (FAO, 2019).*

Country	*Weight exported between 1990-2016 (tonnes)*
Philippines	*27,527*
Malaysia	*25,024*

China	*17,540*
Canada	*16,295*
Singapore	*15,925*
Indonesia	*10,577*
United States of America	*7,912*
Japan	*7,131*
Iceland	*5,848*
Maldives	*4,868*

The global fishery for sea cucumbers has increased dramatically over the last 25 years. In the 1980s and 1990s, more and more countries began to export Holothurians and world production increased. The global catch of sea cucumbers was estimated at 25,000 t (live) in 1983. The catch consisted mainly of tropical Indo-Pacific species. Global captures increased three-fold between 1985 and 1986 and doubled during the 1987–1989 period, in response to increased demand in Asian markets. In 1989, catches totalled 90,000 t worldwide, broken down as follows: 78,000 t from the South Pacific and South-east Asia. The sea cucumber fishery continued to increase, with a global harvest of 120,000 t in the early 1990s.

More recently global captures have steadily increased between 1990 and 2016, from a low of 12,140 t in 1993 to a high of 31,287 t in 2016 (Figure 1). Data in Figure 1 show a fairly even annual capture rate in the Pacific Ocean whereas annual captures of sea cucumbers have been recently decreasing (2008–2016) in the Indian Ocean and increasing in the Atlantic Ocean and adjacent seas. Reductions in captures from the Indian Ocean fit with the theory that Holothuria *fisheries are "boom and bust" industries that over-exploit resources until they are depleted (Anderson* et al., *2011; Purcell* et al., *2013). Seven of the top ten countries with the highest weight of sea cucumber captures between 1990–2016 are range States of* H. fuscogilva, H. nobilis *(including* H. "pentard") *and/or* H. whitmaei *(Indonesia, Madagascar, Sri Lanka, Philippines, Papua New Guinea, New Caledonia and the Solomon Islands).*

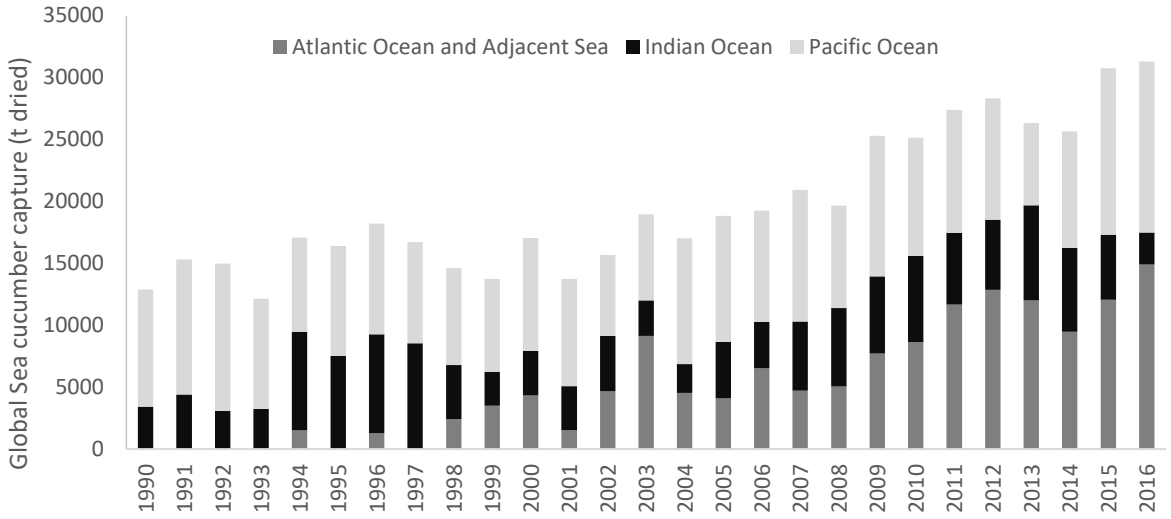

Figure 1*: Global sea cucumber capture data between 1990 and 2016* (FAO, 2019)*. Species specific data were available for* Apostichopus japonicus *so it was removed from the presented data set as it can be confirmed that they are not from the genus* Holothuria*.*

In the longer term, the global sea cucumber fishery has grown from 4,300 t in 1950 to 23,400 t in 2000, falling to 18,900 t in 2001 (fresh or frozen, frozen, dried, salted or in brine, canned). The increase is likely due to the combination of several factors: new sea cucumber producing countries, more exploited species, increased fishing effort from deep-sea stocks, and finally, gradual expansion of fishing areas. Some countries have experienced dramatic declines in landings due to over-exploitation of wild populations.

Hong Kong SAR import statistics show an increase in the number of countries exporting dried, salted or brine cucumbers: 25 countries in 1989, 49 in 2001 and 78 in 2005. In 2005, eight countries exported more than 1,000 t

of sea cucumbers each to the Hong Kong SAR, six countries exported between 500 and 1,000 t, 10 countries between 150 and 500 t, and the remaining 54 countries recorded lower catches to 150 t. The main exporting countries to the Hong Kong SAR were Indonesia, the Philippines, Papua New Guinea, Singapore and Fiji.

It is estimated, however, that the available trade figures would underestimate the total volume of world trade, given that trade chains are complex, incomplete export data and individual species rarely differentiated in trade statistics. FAO trade figures on world exports are low due to the lack of data published/*provided* by exporting countries.

Despite over-exploitation, global catches have increased due to the change of species composition and the opening of new fishing zones. However, it is also possible that catches are better reported than before and/or that more species are harvested.

Further examination of Holothuroidea *fisheries suggests that the majority of fisheries follow a boom and bust pattern of high exploitation, followed by a collapse, with fisheries developing further away from target markets in Asia to feed demand. High value species (such as Holothuria nobilis, H. fuscogilva and H. whitmaei) are exploited first in shallower, near-shore areas and then spatial expansion occurs in off-shore areas until the resource is depleted and lower value species are targeted (Anderson* et al., *2011).*

Based on an analysis of Purcell *et al*., (2013), the current Holothurian fisheries in the Indo-Pacific regions are predominantly over-exploited or depleted.

Species level capture data has been reported by the Seychelles for both Holothuria fuscogilva *and* H. nobilis *since 2001 (FAO, 2019) and during that period the Seychelles has captured a total of 184 t of* H. nobilis *and 1,724 tonnes of* H. fuscogilva *(Figure 2). During data collection in 2003–2004, a survey estimated densities of* H. fuscogilva *and* H. nobilis *at 0.63 and 1.98 ind. ha-1 (Aumeeruddy & Conand, 2008). Data were collected again in 2011–2013 and reported densities of 0.29 ind. ha-1 and 0.53 ind. ha-1 for* H. fuscogilva *and* H. nobilis *respectively (Koike, 2017). These results show a recent decline in density of 54% for* H. fuscogilva *and 73% for* H. nobilis *in the Seychelles.*

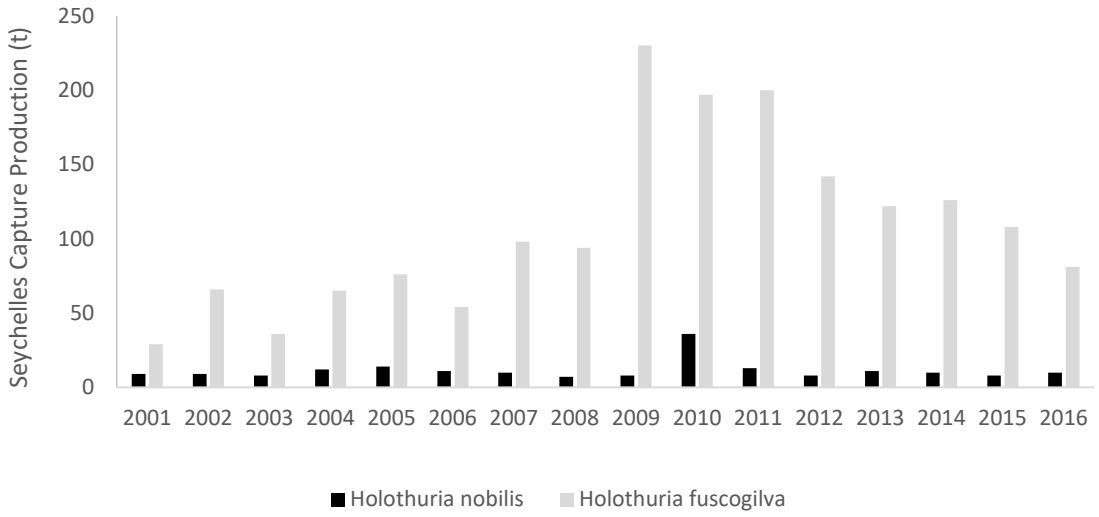

Figure 2: *Capture production figures (in tonnes) for Seychelles for* H. nobilis *and* H. fuscogilva *between 2001 and 2016* (FAO, 2019).

Inclusion in Appendix II to improve control of other listed species

A) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Successful reproduction of sea cucumbers is density-dependent, due to their benthic nature and reproduction via broadcast spawning, fertilisation of eggs can be greatly impaired by low population densities (Bell et al., *2008). This phenomenon has been referred to as the Allee effect (Courchamp* et al., *1999) and can cause progressive decline as population growth decreases leading eventually to extirpation. It is not known what the critical density for specific Holothurian species is to prevent extirpations, but there are examples of* H. nobilis *populations not recovering in Australia, even two years after commercial fishing in the area had been stopped (Uthicke* et al.,

2004). Declines in H. nobilis *populations were also observed in the Chagos islands between 2006 and 2010, even after sea cucumber harvesting had reduced (Price* et al., *2013).*

Additional Information

Threats

Despite the commercial importance of teatfish, there is still much to learn about their biology, ecology and the dynamics of their populations. The lack of scientific information thus constitutes an indirect threat, since it is essential to establish comprehensive management plans capable of ensuring the conservation of these species and sustainable harvesting schemes. The species *Holothuria* sp. "*pentard*" which has not yet been described, demonstrates current gaps in taxonomy even for large commercial species. Habitat degradation and loss also contribute to the decline of teatfish. These species are found in coral reefs that are degraded by climatic oscillations (e.g. El Niño), ecological disasters (e.g. tsunamis), and many other anthropogenic degradations, including (e.g. the use of explosives and poison), as well as coastal pollution and sedimentation.

Conservation, management and legislation

While Holothurian fisheries are still not regulated in a number of countries, other countries have adopted management measures at various levels to avoid overfishing. In general, these measures include specific areas of permitted and prohibited harvesting, licensing, quota, seasonal harvesting, rotating harvesting and other strategies. In several countries, sites were closed soon after the opening of the fishery due to over-exploitation and rapid biological or commercial disappearance. At some sites, the taking of teatfish species is prohibited because of their rarity.

Historically, management of sea cucumber fisheries is based on tenure systems owned by local communities, particularly in traditional fishing areas. However, with the expansion of this activity into non-traditional areas, loss of ancestral crops and increased demand, commercial fisheries are often poorly managed, resulting in the implementation of an inventory once depletion has already begun. In most developing countries, new fisheries are starting up under open access regimes, and management plans or regulations such as prohibitions (e.g. closure of an area) or fishing seasons are introduced in order to try to mitigate the decline of the resource.

In the tropics, fishing is done at a small-scale but is of great socio-economic importance. Management measures have been taken in some tropical countries. In most of teatfish range countries, government management of sea cucumber fisheries is in place. Unfortunately, such management is generally poorly applied, probably due to poor human resources and other resources, capacity to implement and control regulations, effective scientific monitoring mechanisms and adequate management measures taking scientific information into account.

The different types of management measures taken for sea cucumbers are detailed in the SS and summarised as follows:

– Prohibited fishing zones: around the world, fishing exclusion zones are recognised for the benefits they bring to exploited species. There are a few rare examples for sea cucumbers. In Egypt, prohibited fishing areas had a greater diversity and density of commercial species of sea cucumber. In Australia, *Holothuria whitmaei* densities were 75% higher in prohibited fishing areas than in fishing areas. Prohibited fishing areas can be beneficial especially when they have been established and approved in conjunction with actors such as fishermen. However, their success depends largely on the continued support of fishing communities, the effectiveness of enforcement measures and the fight against fraud, and tangible benefits for local actors. Selection criteria for the establishment of prohibited fishing areas should take into account the type, size, shape and number of habitats, as well as the characteristics of other prohibited fishing areas.

– Complete closure of fisheries: in India, in 2001, all species of commercialised Holothurians were listed in annex I of the Wildlife Protection Act (1972), which prohibits all fishing activities. This decision was aimed at promoting the recovery of over-exploited populations; nevertheless, illegal fishing continues, and most stocks are or remain severely depleted. Despite its potential benefits for wild populations, a total ban on sea cucumber fishing has significant socioeconomic consequences and has not proved to be effective in practice. For fishermen such a prohibition, if not accompanied by an alternative, means a significant loss of income and may incite them to fish illegally. Such situations may be even more detrimental to sea cucumbers and unfavourable to humans because they cannot enforce biological thresholds and pay a fair and equitable price.

– Limited access: in general, limited access is a licensing system whereby the number of fishers or vessels involved in fishing is limited. This management tool can curb the competition between fishermen and help maintain a fishery. This system also improves compliance with management measures and can help ensure that economic benefits accrue to local communities. Granting land rights to fishermen's co-operatives can help manage open access fisheries. This management approach appears to be effective in developed countries

where there are other alternatives for sea cucumber fishers who have been displaced (e.g. in Australia). However, in traditional systems, this procedure is difficult to apply since all fishermen have equal rights to exploit their resources. Moreover, this procedure may, as such, prove to be binding on the managing authorities fishing, and even social disturbances and conflicts. Fishermen's co-operatives should be organised so that licences are only granted to people whose main source of income is the sea cucumber fishery and not to any member of the co-operative. In Fiji, only indigenous fishermen are allowed to fish for sea cucumbers.

– Quotas: quotas or total allowable catches (TACs) are the maximum number of individuals or biomass that can be exploited each year during a fishing season or fishing expedition in certain areas, etc. In the sea cucumber fisheries on the eastern coast of Australia a TAC for the species *Holothuria fuscogilva* was introduced after the collapse of the *H. whitmaei* fishery in 1999. The TAC for *H. fuscogilva* is examined each year while *H. whitmaei* is banned from fishing. In Australia, in the Northern Territory, a TAC of 127 t was set for *H. fuscogilva*. It is combined with a mixture of input and output controls, including temporal and spatial management tools, size and gear restrictions. In Papua New Guinea, a quota has been set for each province but is often exceeded.

– Minimum catch sizes: the minimum catch sizes are based on size at maturity to ensure reproduction of the stock at least once before entering the fisheries. This can help prevent a population collapse due to recruitment failure. In addition, this management tool facilitates the targeting of large individuals who reach higher prices in the market. However, the size and weight of sea cucumbers are largely dependent on the amount of live and processed individuals in the water, which can pose problems. For many commercial species, biological information is lacking to determine the minimum harvest size. This management tool is also used in Australia, Papua New Guinea, Fiji and Tonga, with other regulatory methods such as quotas. Nevertheless, the minimum sizes set vary by country, region and species. For example, on the west coast of Australia, the minimum landing size was set at 15 cm for all commercial sea cucumber species, while in the western region this minimum size varies by species. It is important to improve the training of fishermen so that they avoid taking too small individuals. It is also possible that sea cucumbers rejected because of their small size are sold on the black market at lower prices.

Given the importance of densities for healthy breeding populations, ensuring fishing does not deplete densities to below this level could be a useful management measure, and should not be technically difficult.

Artificial propagation/captive breeding

To protect their Holothurian populations from overfishing, countries have developed new methods to produce beche-de-mer. These measures have gained importance since methods of reproduction and rearing of larvae and juveniles have been developed for some commercial species. However, access to technology for juvenile production is not sufficient to initiate stock rebuilding and enhancement programmes.

The Sandfish *Holothuria scabra* (not a teatfish, but a member of the same genus) has been identified as one of the most promising species of sea cucumbers for aquaculture and has been subjected to captive breeding experiments in Australia, India and Viet Nam. Unfortunately, the results to date with other experimental species in aquaculture enterprises have not been positive, especially for species *H. nobilis* and *H. fuscogilva*. Ranching operations have also been developed in the Northern Territory of Australia, since 2015, with further juvenile releases planned for 2019.

More recent data suggest that aquaculture production of sea cucumbers (almost entirely made up of Apostichopus japonicus*), has accounted for more sea cucumber production than capture fisheries since 2011, but this drastic increase in aquaculture production of sea cucumbers has not reduced the annual amount of sea cucumbers captured from the wild (Figure 3). Even though the majority of sea cucumbers produced from aquaculture do not come from the* Holothuria *genus, recent studies have seen successful hatchery efforts that allowed a single technician to produce 18,800 juvenile* Holothuria scabra *(Militz* et al., *2018). Furthermore, some Pacific island countries have completed successful research trials of hatchery and release techniques and are at a stage to scale up, with the Federated States of Micronesia having a project for hatchery-based releasing of* H. nobilis *(Pickering & Hair, 2012).*

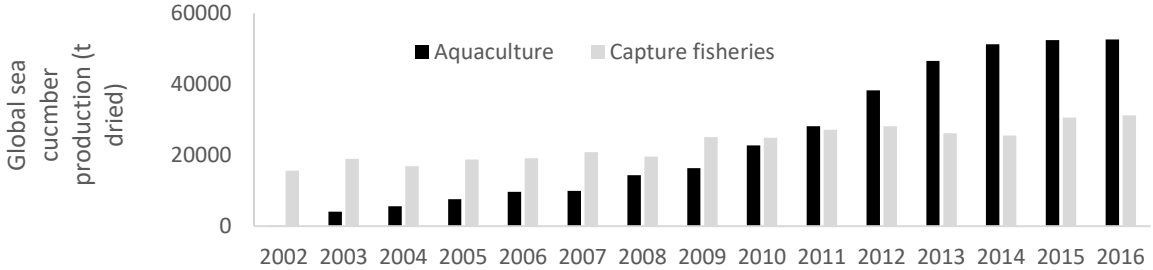

Figure 3: *Annual global aquaculture production and wild captures of sea cucumbers* (FAO 2006–2019). *Data for aquaculture production of* Apostichopus japonicus *were converted from fresh weight to dry weights using the conversion factor 0.04 (Skewes et al., 2004).*

Implementation challenges (including similar species)

*Although one species within the subgenus has been omitted from the proposal (*Holothuria fuscopunctata*), this species is not easily confused with the other members of the subgenus as it lacks the protrusions (teats) that make* H. fuscogilva, H. nobilis *and* H. whitmaei *so recognisable. A guide was produced in 2012 listing and explaining the commercially important sea cucumbers of the world (Purcell* et al., *2012). The guide gives the diagnostic features and photographs of each commercially important species (including teatfish) in both their live and processed forms.*

Researchers of these species are now separating out teatfish specimens during market surveys to investigate price comparisons of different species (*Purcell* et al., *2018*)*, suggesting that it is possible to identify them to a species level in their processed form.*

Other comments

One study in the Philippines has shown that the market chain for sea cucumbers is multi-layered and can involve up to six layers (Brown et al., *2010). As trade chains for sea cucumbers are reportedly complicated, strong traceability systems will be required in the event of an Appendix II listing, such as the labelling scheme used in the caviar industry or the tagging system used for crocodile skins (Mundy and Sant, 2015).*

A socio-economic study in two communities in Papua New Guinea showed that households are not financially poorer when sea cucumber harvesting bans are imposed (Purdy et al., *2017). During sea cucumber collection moratoria, households increased the diversity of the species collected to maintain their income. These results suggest that CITES Appendix II trade regulation will not have a significant impact on livelihoods.*

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Inclusion of Ornamental Spiders *Poecilotheria* spp. in Appendix II

Proponent: Sri Lanka and United States of America

Summary: There are currently 15 recognized species of ornamental spiders in the genus *Poecilotheria*, with four described since 2006. Eight species are endemic to India, five are endemic to Sri Lanka, and two species occur in both countries. The IUCN Red List categorises two *Poecilotheria* species as Critically Endangered, three as Endangered, one species as Vulnerable, one species as Least Concern and one species as Data Deficient. The remainder are yet to be assessed. *Poecilotheria* spiders live in forested areas including some species in teak and banana plantations. They live in pre-existing holes or cavities in trees or behind loose bark and have been found in crevices of buildings located nearby to forested areas. The main threat to *Poecilotheria* species appears to be habitat loss and fragmentation.

Due to the cryptic nature of *Poecilotheria,* their nocturnal habits and sensitivity to vibrations by human surveyors they are difficult to study. Therefore, current and historical population estimates and status for these species are lacking and population trends are unknown. Although many of these species' habitats are known to be declining, knowledge on the distribution of some species continues to increase as new localities are recorded extending their known range.

Poecilotheria are currently protected from wild harvest in Sri Lanka, however they can be legally collected in India except within protected areas. This genus of tarantulas is popular in the pet trade due to their coloration and size, with the USA and Europe the main destinations. US import data revealed 20,000 live specimens imported between 2008 and 2017, the majority (97%) were reported as captive-bred mostly coming from European countries. According to some experts, most species are considered easy to breed and the number of individuals bred in captivity is likely enough to sustain the overall demand of those species. However, morphological and genetic diversity is said to be highly sought after in the hobby and there are some concerns over ongoing offtake from the wild. Imports into the USA of wild-caught *Poecilotheria* totalled 643 over the same time period and included at least 10 species: *P. metallica* (253), *Poecilotheria spp.* (124)*, P. rufilata* (69), *P. tigrinawesseli* (42), *P. regalis* (38)*, P. formosa* (30), *P. fasciata* (26), *P. ornata* (26), *P. striata* (16), *P. miranda* (14) and *P. subfusca* (5). Few wild sourced specimens where reported as originating in a range State and most were imported from Europe.

Poecilotheria metallica (Indian endemic) was the most commonly imported species into the USA (7,900 live, of which 253 were wild) and was assessed in 2008 by IUCN as Critically Endangered because its range was limited to <100 km^2 which was declining and severely fragmented.

Poecilotheria regalis (Indian endemic) was one of the species most commonly imported into the USA (1,700 live, of which 38 were wild) and was assessed in 2008 by IUCN as Least Concern as it was widely distributed in India and although its available habitat was known to be shrinking and faced several threats, it was considered one of the most abundant of all *Poecilotheria* species.

Poecilotheria hanumavilasumica (first described in 2004) was assessed as Critically Endangered in 2008 due to a limited distribution (<6 km^2) and continuing decline in area, quality, populations and number of mature individuals. In 2015, this species was discovered in Sri Lanka, extending its known range. A total of 114 live specimens (all reported as captive-bred) were imported into the USA between 2008 and 2017. Smuggling of adults and juveniles from India has been reported.

Several other threatened species were reported in trade, most of which was reportedly captive-bred, although some limited wild trade was reported and for some species international trade was said at the time to be a threat although it is not known if this is still the case.

Some species of *Poecilotheria* appear to be morphologically distinct, such *as P. metallica* and could be easily identified by enforcement authorities. However, other species such as

P. hanumavilasumica closely resemble other species and could pose challenges in enforcement. The ventral leg markings of the species are the primary identifiers of most species, leg-banding patterns appear to be relatively conserved with little intra-specific variation between individuals and the taxonomy of the group is unresolved.

Analysis: Information on the historical and current wild population sizes of *Poecilotheria* species is lacking and therefore population trends are unknown, although most populations are believed to be fragmented with limited ranges. *Poecilotheria* are currently protected in Sri Lanka, but can be legally collected in India outside of protected areas. Available information indicates that the main threat faced by *Poecilotheria* species is habitat loss and fragmentation.

Of the species which appear in international trade, at least one, *P. regalis* is widespread and considered by IUCN to be Least Concern. The USA is thought to be one of the main markets (as well as Europe). Based on US import data, trade in wild specimens is low and it is unlikely the species meets the criteria for inclusion in Appendix II.

Other species such as *P. hanumavilasumica* and *P. metallica* have restricted ranges and declining habitat and may already meet the biological criteria for inclusion in Appendix I, although much of the trade into the USA is reported to be from captive-bred sources and therefore it is unclear what impact trade is having on these species in the wild. It would seem to be precautionary to list these two species in Appendix II.

Species within this genus are distinguished through their leg-banding patterns. *P. metallica* could be easily identified by enforcement authorities, whereas *P. hanumavilasumica* closely resembles other species including *P. fasciata* and *P. striata*. It is also likely that the taxonomy will continue to evolve. Thus, if Parties consider *P. hanumavilasumica* and *P. metallica* to meet the criteria for inclusion in Appendix II, then it would be deemed appropriate to list the genus in order to facilitate implementation.

Other Considerations: If the proposal is rejected, the range States could consider an Appendix-III listing for their species. In this case stipulating a zero-export quota for wild specimens with the listing for Sri Lanka would reflect that export from there is illegal.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Species and description	Range	IUCN Global Category	Sri Lanka National Assessment
Poecilotheria chaojii (Mirza, Snap & Bhosale, 2014)	India	Not Assessed	Not Applicable
Poecilotheria fasciata (Latreille, 1804)	Sri Lanka	Not Assessed	Endangered (2012)
Poecilotheria formosa (Pocock, 1899)	India	Endangered (2008)	Not Applicable
Poecilotheria hanumavilasumica (Smith, 2004)	India & Sri Lanka	Critically Endangered (2008)	Not Applicable
Poecilotheria metallica (Pocock, 1899)	India	Critically Endangered (2008)	Not Applicable
Poecilotheria miranda (Pocock, 1900)	India	Endangered (2008)	Not Applicable
Poecilotheria ornata (Pocock, 1899)	Sri Lanka	Not Assessed	Endangered (2012)
Poecilotheria rajaei (Nanayakkara, Kirk, Dayananda, Ganehiarachchi, Vishvanath & Kusuminda, 2012)	Sri Lanka	Not Assessed	Not Applicable
Poecilotheria regalis (Pocock, 1899)	India	Least Concern (2008)	Not Applicable

Poecilotheria rufilata (Pocock, 1899)	India	Endangered (2008)	Not Applicable
Poecilotheria smithi (Kirk, 1996)	Sri Lanka	Not Assessed	Critically Endangered (2012)
Poecilotheria striata (Pocock, 1895)	India	Vulnerable (2008)	Not Applicable
Poecilotheria subfusca (Pocock, 1895)	Sri Lanka	Not Assessed	Endangered (2012)
Poecilotheria tigrinawesseli (Smith, 2006)	India	Data Deficient (2008)	Not Applicable
Poecilotheria vittata (Pocock, 1895)	India & Sri Lanka	Not Assessed	Endangered (2012)

Some of the assessments were last conducted in 2008 and may now be out of date.

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influence

Data on *Poecilotheria* population sizes are incomplete due to the cryptic nature of the genus. *Current and historical population estimates and distributions for Poecilotheria species are lacking and therefore population trends for these species are unknown (USFWS, 2018). A study in 2000 (Molur, 2003) in India found specimens of Poecilotheria in habitats including teak plantations, banana plantations, primary evergreen forest, secondary moist and dry deciduous forest, highly degraded secondary forest and human settlements.*

There is a lack of data on the distribution and biology of *Poecilotheria;* however, it is assumed that many species in the genus have decreasing populations based on habitat loss and degradation. *Recent studies have attempted to gain a better understanding on Poecilotheria population and distribution. In Sri Lanka one study looked at the occurrence of species across sites (selected for their likelihood of being appropriate habitat) using a capture and release method and found only three of 71 sites where no species were observed. In the other 68 sites a total of 112 sightings were made;* P. fasciata *(61),* P. ornata *(18)*, P. subfusca *(9),* P. smithi *(2),* P. pederseni *(now* P. vittata *(7)) and* P. hanumavilasumica *(3) were recorded. This survey recorded two tarantulas which had not been previously described and considered likely to be new species (Nanayakkara* et al.*, 2012). The known distribution for some species such as* P. smithi*,* P. striata *and* P. hanumavilasumica *are larger than previous estimates (Nanayakkara* et al.*, 2013, Siliwal* et al.*, 2013 & Nanayakkara* et al.*, 2015).* Poecilotheria hanumavilasumica *is currently categorised as Critically Endangered by IUCN (2008) due to its limited distribution which is now known to extend across two range States. Although it was noted that the collection of females and the killing of males by locals could have serious consequences on the populations, more studies are necessary to determine the impacts of harvests and habitat change on Poecilotheria densities (Molur, 2003).*

Poecilotheria are considered to be affected by habitat loss and fragmentation. Collection of these spiders from the wild may have a significant effect on wild populations; the loss of individuals from a single location or population could significantly decrease the genetic diversity of the species and make the survival of that population more vulnerable to natural or anthropogenic threats. *It appears that some species may adapt better than others to changing habitats, for example* Poecilotheria regalis *was observed in mixed vegetation, primary and secondary forests but did not adapt well to degraded habitats (Molur* et al.*, 2004).*

The 15 species of Poecilotheria listed on the World Spider Catalog (2019) (which match those in the SS) have not been molecularly researched to determine if some of these species are synonymous and thus could potentially extend a senior synonym's species range by the combination of two or more species (R. West, in litt., 2019). Surveys have also found Poecilotheria species which could not be confidently assigned to any of the known species, which artificially increases the range of current described species (S. Henriques, in litt., 2019).

Trade

Poecilotheria are valued in the pet trade with main destination markets in Europe and America. Captive-bred specimens have consistently made up most of the trade over those declared as wild-origin. Between 2006 and 2017, the USA imported 22,918 live individuals of *Poecilotheria spp. Poecilotheria metallica, P. regalis, P. ornata* and *P. rufilata* were the most commonly traded species and a large majority were imported from European countries. From 2013-2017, the most recent 5-year period with complete data, the number of declared imports of *Poecilotheria spp.* was 16,510 and the number of declared exports was 145. The demand for specimens of this genus has risen considerably over the past three decades.

Analysis of USA import data between 2008 and 2017 reported a total of 20,036 live specimens imported into the USA for the purpose of commercial trade (76 were imported for other purposes) (see Figure 1 and Table 1):

Species	Range	IUCN Global Category	Imports to USA "C/F"	Imports to USA "W"
Poecilotheria metallica	*India*	*CR*	*7697*	*253*
Poecilotheria species			*3770*	*124*
Poecilotheria regalis	*India*	*LC*	*1662*	*38*
Poecilotheria ornata	*Sri Lanka*	*NA*	*1348*	*26*
Poecilotheria rufilata	*India*	*E*	*1149*	*69*
Poecilotheria tigrinawesseli	*India*	*DD*	*890*	*42*
Poecilotheria miranda	*India*	*E*	*824*	*14*
Poecilotheria formosa	*India*	*E*	*509*	*30*
Poecilotheria fasciata	*Sri Lanka*	*NA*	*455*	*26*
Poecilotheria striata	*India*	*V*	*364*	*16*
Poecilotheria smithi	*Sri Lanka*	*NA*	*344*	
Poecilotheria vittata (trade reported as P. pederseni)	*India & Sri Lanka*	*NA*	*248*	
Poecilotheria hanumavilasumica	*India & Sri Lanka*	*CR*	*114*	
Poecilotheria subfusca	*Sri Lanka*	*NA*	*69*	*5*

Table 1: Imports to the USA of live spiders reported as Captive-born or bred and wild sourced between 2008-2017.

- *The main exporting countries of live specimens for the purpose of commercial trade were Germany (17,009), Poland (1,584) and the Netherlands (587).*
- *Few of the wild-sourced spiders were reported with one of their ranges States as the Country of Origin, and most had come from European countries.*
- *During this period imports peaked in 2017 when 5,112 live specimens were imported, a relatively large proportion of which were wild (6%) and comprised of a mixture of species.*

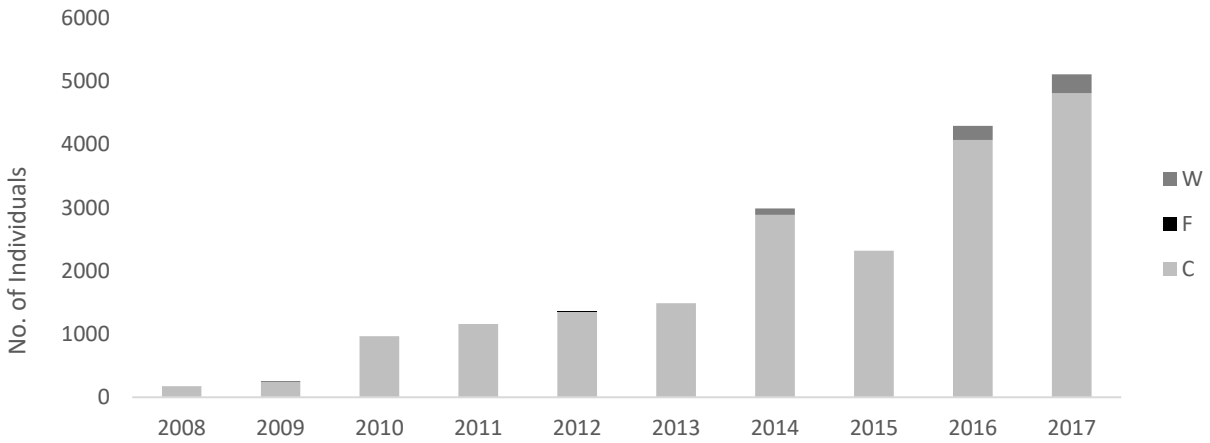

Figure 1: Imports of live *Poecilotheria spp.* into the USA reported in LEMIS data between 2008-2017

Spiders from National Parks and other protected areas are believed to be illegally collected and sold on the international market. Collectors often try to capture gravid females and future progeny from wild populations, then misclassifying them as captive reared. There are indications that illegal smuggling from Sri Lanka occurs for introduction into the pet trade, but few detailed accounts are available. Collection (legal/illegal) of spiders for the pet trade is having a negative effect on wild populations in India. *There is little evidence of illegal collection and trade of Poecilotheria species to supply the international pet trade. The Supporting Statement provides anecdotal information on the illegal collection of Poecilotheria spp. in India however, notes that this genus is not currently protected under Indian legislation. In addition, the Supporting Statement notes that commercial collection and exportation of Poecilotheria from Sri Lanka is prohibited but does not provide examples or information of the scale of illegal collection that is taking place in this range State (S. Henriques & E. Cooper, in litt., 2019).*

A documented case of wild-caught species used for parental captive stock was for the species P. regalis *(Gabriel* et al.*, 2011a), which is found in India and was assessed by IUCN (2014) as Least Concern.* Poecilotheria spp. *are not a protected species in India and therefore collection of individuals outside of protected areas is legal.*

The following species (in order of trade levels) are those that are most in trade and/or threatened (all US import data is from LEMIS):

Poecilotheria metallica: Assessed as Critically Endangered in 2008, *because its range was limited to <100 km² in a single location (*despite surveys conducted in adjacent areas*) and continuing decline in habitat quality (Molur* et al.*, 2008). The species was rediscovered in 2001, after 102 years from the last observation in the wild. The habitat where the species occurs is completely degraded and under intense pressure from the surrounding villages as well as from insurgents (Molur* et al.*, 2008). A total of 7,900 live specimens were imported into the USA between 2008 and 2017: 253 of which were wild and 7,647 captive-bred. An incident of smuggling was recorded in 2002 when two Europeans took a few specimens out of the country and advertised them for sale on the internet. There are also reports available of other such incidents since then (Molur* et al.*, 2008). Since 2001 it is said to have been successfully bred in captivity (Dye, 2012) and appears easier to breed than other species of Poecilotheria (although there have been documented cases of cannibalism when rearing spiderlings (Gabriel, 2011b)).*

Poecilotheria regalis: Assessed as Least Concern in 2008, *as it is widely distributed in India, throughout northern Western Ghats and present in a few locations in the Eastern Ghats (Molur* et al.*, 2008) Is considered to have the widest distribution of* Poecilotheria spp.*and possibly the species with the longest and most extensive captive breeding history (S. Henriques, in litt., 2019). The known and inferred extent of occurrence for* P. regalis *is estimated to be >50,000 km^2 and area of occupancy is likely to be >2,000 km^2 (Molur* et al.*, 2008). This species has been found to utilize various wooded habitats, from medium to old-growth teak plantations, primary forest in addition to severely degraded forest and banana plantation (although based on one specimen). It appears to have adapted to changing habitats but does not do well in degraded habitats (Molur* et al*, 2004). Appears to be one of the most traded Poecilotheria species. A total of 1,700 live specimens were imported into the USA between 2008 and 2017: 38 of which were wild and 1,662 captive-bred.*

Poecilotheria ornata: *Assessed as endangered in the Sri Lanka National Red List (2012) and is estimated to occupy <500 km^2 of its range (MOE, 2012). It is found in the plains and hills of the lowland wet zone in southwestern Sri Lanka (Nanayakkara, 2014, unpublished data). This species is now considered to be one of the easier of the Poecilotheria species to breed in captivity (Gabriel, 2005a). This species was introduced to the hobby from a parental stock of approximately 16-19 individuals which were exported to Germany, most of the captive stock has been produced from these original specimens (Gabriel, 2012). A total of 1,374 live specimens were imported into the USA between 2008 and 2017: 26 of which were wild and 1,347 were captive-bred and 1 captive-born.*

Poecilotheria rufilata: Assessed as Endangered in 2008. *The species is limited to fewer than five fragmented locations in the southern Western Ghats, and the area of occupancy is estimated to be less than 2,000 km^2. Population information is not available, but it is said to be decreasing. The species suffers from the threats of habitat degradation and loss and is sometimes killed by humans, in addition to collection for the international pet trade (Siliwal* et al.*, 2008). A total of 1,218 live specimens were imported into the USA between 2008 and 2017: of which 69 were wild and 1,149 captive-bred. There are conflicting reports on how easy the species is to breed: with some considering that it requires a continuous influx of wild individuals (Siliwal* et al.*, 2008), and others reporting that can be easily bred in captivity (Gabriel, 2012).*

Poecilotheria tigrinawesseli: Assessed as Data Deficient in 2008. *This species is endemic to India and recorded from only six locations in the Eastern Ghats (Siliwal* et al.*, 2008). The extent of occurrence encompassing known and inferred distribution is more than 20,000 km^2 and the area of occupancy is unknown. This species suffers from the threat of habitat loss which is considered a general threat in the Eastern Ghats. A total of 932 live specimens were imported into the USA for the purpose of commercial trade between 2008 and 2017: 42 of which were wild and 890 captive-bred.*

Poecilotheria miranda: *P. miranda was assessed as Endangered in 2008 as it is recorded only from a few locations in the Chhota Nagpur region (area of occupancy <less than 2,000 km²) (Siliwal* et al.*, 2008). Threats include severe fragmentation, loss of habitat, loss of habitat quality, and indiscriminate collections by several European pet traders. A total of 838 live specimens were imported into the USA between 2008 and 2017: 14 of which were wild and 824 captive-bred.*

Poecilotheria formosa: *Assessed as Endangered in 2008 as it was known from only three sites in two areas of the highly degraded Eastern Ghats. The species has a small range and its extent of occurrence (less than 5,000 km²) and area of occupancy (less than 500 km²) was limited due to past and continuing decline in the extent, area and quality of habitat. The population is additionally affected by factors including human interference in the form of persecution, collection for pet trade and other developmental activities (Molur* et al.*, 2008). Collection for*

international trade is considered an additional pressure. A total of 539 live specimens were imported into the USA between 2008 and 2017: 30 of which were wild and 509 captive-bred.

Poecilotheria fasciata: *Assessed as endangered in the Sri Lanka National Red List (2012) (MOE, 2012). It is found to occur in forests below 200m elevation in Sri Lanka's dry and intermediate zones north of Colombo and is estimated to occupy an area of <500 km^2 of its range (MOE, 2012). It has been documented to occupy coconut plantations however, this colonization was considered a result of a drought and pressure on their traditional habitat located adjacent to the coconut plantation (Smith* et al.*, 2001).* Poecilotheria fasciata *are one of the easiest Poecilotheria species to breed in captivity (Gabriel, 2005a). A total of 481 live specimens were imported into the USA between 2008 and 2017: 26 of which were wild and 455 captive-bred.*

Poecilotheria striata: Assessed as Vulnerable in 2008. *This species has a small range in the Western Ghats and adjoining Palghat gap, and the area of occupancy is estimated to be less than 2,000 km^2 (Siliwal* et al.*, 2008). It is found to occur in both natural forests and plantations, however, informal observations estimate that densities are higher in natural forests. Threats include habitat loss and degradation as well as indiscriminate collection for the pet trade. A total of 380 live specimens were imported into the USA between 2008 and 2017: 16 of which were wild and 364 captive-bred.*

Poecilotheria smithi: *Assessed as critically endangered in 2012 under Sri Lanka's National Red List assessment (MOE, 2012) and found from the Central Province of Sri Lanka (Kirk, 1996). A total of 344 live specimens were imported into the USA between 2008 and 2017: all reported to be captive-bred.*

Poecilotheria vittata: Poecilotheria vittata *has been assessed as endangered in the Sri Lanka National Red List (2012) (MOE, 2012) and is reported to be present in both India and Sri Lanka, although exact distribution is unknown (Dhali* et al.*, 2016).* Poecilotheria vittata *was removed from the synonymy with* Poecilotheria striata *and is now considered to be the parent synonym of* Poecilotheria pederseni *(Gabriel & Gallon, 2013). All reported trade of* P. vittata *into the USA are reported as* P. pederseni*. A total of 248 live specimens were imported into the USA between 2008 and 2017: all of which were reported to be captive-bred.*

Poecilotheria hanumavilasumica: *Assessed as Critically Endangered in 2008 (requires updating) due to small distribution, and continuing decline in the quality and extent of available habitat, populations and mature individuals. This species was first described in 2004 (Smith, 2004) and was believed to be limited to a few tamarind, casuarina and mix dry deciduous tree and palm plantations (all on private land) on the island of Rameshwaram and on the mainland close to the island (Siliwal* et al.*, 2008). In 2015 Nanayakkara* et al. *(2015) discovered* P. hanumavilasumica *present on the island of Mannar in Sri Lanka, extending the known range for this species. With the new localities discovered* P. hanumavilasumica *has been recorded from eight subpopulations and <15 severely fragmented locations (Nanayakkara* et al.*, 2015). Although not found extensively in pet trade, a few adult males and females along with subadults and juveniles were taken out of the country (Siliwal* et al.*, 2008). A total of 114 live specimens were imported into the USA 2008 and 2017: all were reported as captive-bred.*

Poecilotheria subfusca: *Assessed as endangered in 2012 under Sri Lanka's National Red List Assessment (MOE, 2012) and is endemic to Sri Lanka. It is found to occur in Central Province of Sri Lanka (Smith* et al.*, 2002). The species suffers from threats of habitat loss and degradation due to intensive agricultural activity and illegal firewood collection (Smith* et al.*, 2002). A total of 74 live specimens were imported into the USA between 2008 and 2017: 5 of which were wild and 69 captive-bred.*

Much of what is known about the biology of the genus has been observed in captive specimens, and large knowledge gaps remain. Though many *Poecilotheria* specimens are declared captive-bred and many pet traders advertise their spiders to be captive-bred, according to the Supporting Statement the low reproductive rate of this genus (Poecilotheria females in general have low reproductive rates, producing an average of only 100 eggs per yearly egg sac) and the restricted gene pool of the captive population have made attempts to breed them in high numbers unsuccessful. *However, there has been both published documentation on observations of Poecilotheria breeding in captivity and Poecilotheria species are commonly bred in captivity by amateur hobbyists as well as vendors and are available as captive-bred young in the pet trade in the USA, Europe and elsewhere (USFWS, 2018). It appears that the captive breeding of Poecilotheria is generally successful and the whole genus has been bred, most species are considered easy to breed (J. Wu, in litt., 2019). Although the number of individuals bred in captivity is likely enough to sustain the overall demand of some of the species, logistical and genetic issues might make sustainable captive reared populations of some species unfeasible or unpractical while sought after morphological and genetic diversity makes wild parental stock appealing to breeders (S. Henriques, in litt., 2019).*

Additional Information

Threats

The main threats to this genus include habitat loss, habitat fragmentation, and collection for the pet trade. Due to their specialised habitat needs *Poecilotheria* spiders are affected by habitat loss through deforestation which directly reduces or eliminates the availability of trees that *Poecilotheria* rely on for reproduction, foraging and protection. Due to the limited dispersal ability of this genus, forest loss is likely to result in spider mortality through intentional killing when encountered by loggers, or through physical trauma caused by falling trees. *The primary threat to* Poecilotheria spp*. is habitat loss and fragmentation (USWFS, 2018).*

Conservation, management and legislation

Sri Lanka and the USA submitted a proposal for the listing of the genus in Appendix II at CoP11, but the proposal was rejected based on the lack of information on international trade and populations. It was also noted that the main threat faced by some of the species seemed to be habitat destruction and human encroachment. *Sri Lanka stated their intention to list the genus in Appendix III (USWFS, 2018) but it does not appear this has been done.*

India: The Indian Wildlife Protection Act of 1972 (WLPA) provides for the protection of animals and plants to ensure the ecological and environmental security of India. This legislation establishes protected areas, regulates trade in wildlife parts and products, and provides a framework to regulate hunting and to protect and manage wildlife habitats. The WLPA does not currently list any *Poecilotheria* as protected. Protected areas established under the Act; such as National Parks, prohibit use of wildlife resources and collection in these areas would be illegal.

Sri Lanka: The primary legislation for the protection of wildlife and plant species is the Fauna and Flora Protection (Amendment) Act, No. 22 of 2009, part of the Fauna and Flora Protection Ordinance of Sri Lanka. The FFPA affords protection of habitat and wildlife through designation of protected areas and provides protection for certain species outside of protected areas. Under the FFPA, commercial collection and export of *Poecilotheria* species is prohibited. Though Sri Lanka has a legislative framework for the conservation of this genus, enforcement is said to be weak.

Poecilotheria is currently not protected by international legislation. *In the USA, five species of this genus were listed as Endangered under the United States Endangered Species Act in 2018;* P. subfusca*,* P. ornata*,* P. vittata*,* P. smithi *and* P. fasciata*, all of which are found in Sri Lanka. It is unlawful for any person subject to the jurisdiction of the USA to import into and export from the USA any listed species, or its parts or products, posses, sell or deliver listed species taken in violation of the Endangered Species Act. A person would require a permit to import or export any specimen of Poecilotheria (USFWS, 2018).*

Several populations of *Poecilotheria spp.* are found in or near protected areas in India*.* The first invertebrate sanctuary in India has been proposed by the Zoo Outreach Organization and WILD for the protection of *P. hanumavilasumica.* The proposed Rameswaram Tiger-Spider Sanctuary would protect the largest population of the species in its 6 km^2 range.

Artificial Propagation/captive breeding

A hobbyist describes the difficulties of breeding *Poecilotheria,* specifically *P. smithi* and *P. subfusca* in captivity in 2005 (Jacobi, 2005). *However, there has been more recent evidence that there has been much success in breeding this genus in captivity (USFWS, 2018*). Most of what is known on the biology and ecology of this species has been attributed to captive breeding. *All Poecilotheria (except for* P. ornata *and potentially* P. metallica*) can be kept with their mother or reared together in social groups (Gabriel, 2012). There have been documented cases of hybridization within captive breeding activities that result in different colour variants of the parental stock (Gabriel, 2012).*

The following species are said to be bred in captivity: P. regalis *(Gabriel, 2005a, Gabriel, 2011a),* P. smithi *(Gabriel* et al.*, 2005b),* P. metallica *(Dye, 2012),* P. formosa *(Gabriel, 2011b),* P. ornata *(Gallon, 2012, Gabriel, 2005a),* P. subfusca *(Bustard, 1995),* P. rufilata *(Gabriel, 2012),* P. fasciata *(Gabriel, 2005a),* P. miranda *(Gabriel, 2011b) and* P. tigrinawesseli *(Gabriel, 2011b).*

Implementation challenges (including similar species)

Unresolved taxonomy as well as similarity in appearance of many *Poecilotheria* species mean for implementation purposes the entire genus should be listed. *Some species within this genus appear to be morphologically distinct such as* P. metallica *and could be easily identified by enforcement authorities. However, some species such as* P. hanumavilasumica *closely resembles other species and is recognised to have one of the smallest and most vulnerable ranges (S. Longhorn, in litt., 2019). The ventral leg markings of the species are the primary identifiers of each species, leg-banding patterns appear to be relatively conserved with little intra-specific variation between*

individuals of captive-bred stock as the majority of these individuals are derived from a small sample of wild populations (Morra, 2013). Distinction between species' leg bands can be hard to discern, particularly in juvenile individuals and the taxonomy of the group is unresolved.

Identification of juveniles at the species level is not reliable and furthermore egg identification at the genus level is also considered to be highly questionable (S. Henriques, in litt., 2019). Unlike most tarantulas, Poecilotheria exhibit a degree of sociality and spiderlings often remain with the female until maturity (USFWS, 2018).

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Inclusion of Mindoro Peacock Swallowtail *Achillides chikae hermeli* in Appendix I

Proponent: European Union and Philippines

Summary: This proposal recommends the inclusion of Mindoro Peacock Swallowtail *Achillides chikae hermeli* in Appendix I, and the adoption of a new standard taxonomic reference for Papilionidae (swallowtail butterflies) in the Philippines.

This Mindoro Peacock Swallowtail was discovered in 1992 on Mindoro Island in the Philippines and upon its discovery it was named as *Papilio hermeli*. A very similar butterfly the Luzon Peacock Butterfly was already known from Luzon Island, adjacent to Mindoro. The Luzon Peacock Butterfly currently designated *Papilio chikae* was included in Appendix I in 1987. It is one of three species in the genus *Papilio* currently listed in the Appendices. Some taxonomists consider these two populations to be subspecies of the same species. Some also consider these and other species of East Asian swallowtails to belong to a separate genus *Achillides,* which otherwise is considered a subgenus of *Papilio*.

The Proponents recommend the adoption of Page and Treadaway (2004) as a standard taxonomic reference for Papilionidae in the Philippines, a source that recognises *Achillides* as a separate genus, and that considers the two populations to be subspecies of the same species (*Achillides chikae chikae* and *Achillides chikae hermeli*). Having consulted the Animals Committee Nomenclature Specialist, the Proponents state that the adoption of Page and Treadaway (2004) would have the effect of changing the current listing of *Papilio chikae* in Appendix I to *Achillides chikae chikae*.

They also propose including the Mindoro population, considered as *Achillides chikae hermeli* by Page and Treadaway (2004), in the Appendices on the basis that it closely resembles *Papilio chikae*. Although the lookalike criterion in Annex 2bA of *Res. Conf. 9.24 (Rev. CoP17)* allow for listing in Appendix II on that basis, the proponents seek to list this taxon in Appendix I, citing as justification *Res. Conf. 12.11* (*Rev. CoP17*), which recommends that in the case of subspecies:

> b) where there are identification difficulties, the problem be approached by either including the entire species in Appendix I or Appendix II or by circumscribing the range of the subspecies warranting protection and listing the populations within this area on a country basis.

Achillides chikae hermeli is found in two separate mountain masses (Mt. Halcon and Mt. Baco) on the island of Mindoro in the Philippines. Page and Treadaway (2004) described it as being observed from 1800-2400 m on Mt Halcon. Its habitat of montane forests has been decreasing and is fragmented in some parts. The taxon was considered to be rare but with a 'probably stable' population soon after it was described. It has not yet been assessed by IUCN. As a species, *Achillides chikae* (including *Papilio chikae* and *A. c. hermeli*) is said to be a very local butterfly with a tendency to concentrate in certain localities.

All swallowtail butterflies are protected in the Philippines, with trade managed through permits of which none have been issued for this taxon, and therefore all trade is presumably illegal. *Achillides* are considered popular among collectors, naturalists and researchers, and *P. chikae [A. c. chikae]* is reported to be amongst the most beautiful and desirable. Both *A. c. hermeli* and *Papilio chikae [A. c. chikae]* have been found offered for sale online within the Philippines and non-range States. Instances of illegal trade have been noted with indications that specimens of *P. chikae [A. c. chikae]* have been traded under the name "*P. hermeli" or A. c. hermeli.* Although there are distinguishing features between the two taxa, these may not be easily apparent to enforcement officers.

Analysis: *Achillides chikae hermeli* has been observed for sale online (although numbers appear relatively low) within the collector trade, and as the taxon is protected all trade is thought to be illegal. Specimens of *P. chikae* have been traded under the name *P. hermeli* or *A. c. hermeli*. Although there

are distinguishing features between the two taxa, these may not be easily apparent to enforcement officers, and it would appear that listing *A. c. hermeli* in Appendix II would ensure more effective control of trade in the taxon currently listed as *P. chikae.*

The Proponents have recommended the adoption of Page and Treadaway (2004) as the CITES standard reference for Papilionidae in the Phillippines as recommended in paragraph 2 d of *Res. Conf. 12.11 (Rev. CoP17) Standard nomenclature.* If this is adopted, the taxon currently listed in Appendix I as *P. chikae* would become *A. c. chikae*.

Under *Res. Conf. 12.11 (Rev.* CoP17). It is recommended in the case of subspecies where there are identification difficulties that the problem be approached by either including the entire species in Appendix I or Appendix II or by circumscribing the range of the subspecies warranting protection and listing the populations within this area on a country basis. Given that *A. c. hermeli* is considered to meet the criteria in Annex 2b as a lookalike, listing both the subspecies in Appendix I, and thus resulting in the full species *Achillides chikae* would appear to be in line with the recommendations in *Res. Conf. 12.11 (Rev. CoP17)*.

Achillides chikae hermeli is endemic to the Philippines with a restricted distribution on Mindoro Island and decreasing habitat and remaining natural forest reported to be fragmented and largely confined to higher altitudes. There is little population information and it was considered rare although with a population that was 'probably stable' soon after it was described. It may be that the species also meets the biological criteria for listing in Appendix I in its own right.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Papilio chikae was described in 1965 and was listed in Appendix I in 1987.
Achillides chikae hermeli was first described as *Papillio hermeli* in 1992 and is the taxon considered under this proposal. This taxon is not currently listed in the Appendices.

The Proponents recommend the adoption of Page and Treadaway (2004) Papilionidae in the Philippines as the standard reference for *Achillides chikae hermeli* which, if adopted, would treat *Papilio chikae* as *Achillides chikae chikae.* The only subspecies of *Achillides chikae* considered under Page and Treadaway are *A. c. hermeli* and *A. c. chikae*

Range

Philippines.

IUCN Global Category

Not assessed.

Biological criteria for inclusion in Appendix I

A) Small wild population

Papilio chikae hermeli [now Achillides chikae hermeli] was reported to be 'very rare' by Treadaway (1995), Treadaway and Schröder (2012) *and Condamine et al. (2013). There is no quantitative data on population size.*

B) Restricted area of distribution

Achillides chikae hermeli is endemic to the island of Mindoro in the west-central Philippines and inhabits montane forest at altitudes above 1800 m a.s.l. Until 2012 *Achillides chikae hermeli* had been reported from Mt. Halcon only (2,580 m above sea level). *Page and Treadaway (2004) described it as being observed from 1800-2400 m on Mt Halcon.* However, in 2012, the subspecies was also reported from Mt. Baco (2,488 m above sea level), representing an extension of its known range. *As a species Achillides chikae (including Papilio chikae and A. c. hermeli) is said to be a very local butterfly with a tendency to concentrate in certain localities (Page and Treadaway (2004)).*

Gonzales *et al.* (2000) reported that according to historical accounts, Mindoro was once covered entirely by rainforests, however, based on estimates made in 1988, only around 870km^2 of tropical forest remained in 1999.

In 1990, the remaining forests were reported to be largely montane forest located on Mt. Halcon and Mt. Baco and in 2012, the remaining natural forest on Mt. Halcon were reported to be fragmented and largely confined to higher altitudes. *Of the habitat designated as a strict protected zone on Mt. Iglit-Baco, a total of 294km^2 (36%) is >1,000m above sea level (Mallari, et al., 2016).*

C) Decline in number of wild individuals
Whilst the subspecies has not been assessed by the IUCN, Treadaway (in: Danielsen and Treadaway, 2004) assessed the subspecies as vulnerable, using the categories and criteria of the IUCN (Version 2.3) (IUCN, 1995), and considered the population to be 'probably stable'.

Danielson and Treadaway (2004) cautioned that knowledge of the distribution, status and biogeography of Philippine butterflies is limited, and considered it uncertain whether existing records and assigned conservation status "accurately reflect the true distribution and status" of the taxa included in the study.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Legal Trade
The CITES Trade Database records a total of 17 bodies of P. chikae imported, 10 of which came from the Philippines. Most had source code I or O.

Illegal
No permits for trade in *A. c. hermeli* have been approved by the Government of the Philippines. Despite being nationally protected in the Philippines, both *A. c. hermeli* and *Papilio chikae [A. c. chikae]* have been found offered for sale on the internet. A short survey of online trade in *A. c. hermeli* and *Papilio chikae [A. c. chikae]* was undertaken on 10th October 2018, with 18 offers of sale found from sellers within the Philippines, Europe, US, Canada, Taiwan PoC, Thailand and the Russian Federation. Sale of specimens of *Papilio chikae* offered as *hermeli* have been discussed on an online hobbyist website. *A total of three specimens offered as* hermeli *have been observed for sale on eBay and prices were ranging between $50-$70 (eBay, 2019).*

In recent years, EU Member State Authorities have come across multiple cases where traders offered to import into the EU specimens of *Papilio chikae [A. c. chikae]* mislabelled as *Papilio chikae hermeli [A. c. hermeli],* apparently exploiting the inability of customs officers to distinguish the taxa during border controls, for example:

- Two specimens of *Papilio chikae chikae [A. c. chikae]* falsely described as *hermeli* were confiscated by UK police in 2018 and are in the process of being formerly seized by UK Border Force.
- Illegal trade in *Papilio chikae [A. c. chikae]* under the name of *Papilio hermeli* was also reported by the Swedish Management Authority.

One Philippine trader reported that 300-500 individuals of "*Papilio chikae"* could be sourced annually.

The magnitude of the illegal trade is unclear. The impact of international trade on *Papilio chikae [A. c. chikae] and Achillides chikae hermeli* is unknown.

Additional Information

Threats
Achillides are considered popular among collectors, naturalists and researchers. *Papilio chikae [A. c. chikae]* (which *A. c. hermeli* closely resembles) was reported to be among the most beautiful and desirable members of Papilionidae. Due to its slow flight and attraction to decoys, it was considered to be easily captured.

Specific threats to *Achillides chikae hermeli* have not been documented. However, threats to the Iglit-Baco Mountains Important Bird Area (IBA) were reported to include cattle ranching, upland farming and collection of firewood, resulting in rapid deforestation, both within and outside of Mt. Iglit-Baco National Park.

Conservation, management and legislation
National
All swallowtail butterflies in the Philippines are protected under the Wildlife Resource Conservation and Protection Act of 2001, and any collection and trade must be managed through a permitting system.

Republic Act 7586 (1992) prohibits the hunting, destruction, disturbance or possession of any plants or animals, or derived products, from protected areas without a permit from the Management Board; violations are subject to a fine of P5,000-500,000 (ca. USD 96 – 9,600), or imprisonment for between 1-6 years, or both.

Executive Order No. 247 (1995) establishes the regulatory framework for the prospecting (collection, research and use) of biological and genetic resources, their byproducts and derivatives, for scientific, commercial and other purposes, and requires collectors or researchers to obtain Prior Informed Consent (PIC) from the local communities concerned, and to enter into a research agreement whit the Philippine Government.

The management of ancestral domain of the Mangyans in Mt. Halcon is governed by the Indigenous Peoples' Rights Act of 1997 (Republic Act No 8371), which mandates that management of ancestral domains lies primarily with Indigenous Cultural Communities/Indigenous Peoples (ICCs/IPs), with participation from government agencies.

International
Papilio chikae hermeli [A. c. hermeli] is at present not subject to the provisions of international conventions, treaties or regulations. It's close relative *Papilio chikae* has been included in CITES Appendix I since 1987.

Papilio chikae hermeli [A. c. hermeli] is present in Mt. Iglit-Baco National Park. It also occurs at Mt. Halcon which has been identified as an area with a high value of both irreplaceability and vulnerability and recommended that the area be prioritised for conservation action.

Artificial Propagation/captive breeding
No information on captive breeding facilities in the Philippines is known.

Implementation challenges (including similar species)
Species of *Papilio [Achillides]* are morphologically similar, with black wings covered with bright green scales. The "Illustrated identification guide to insects protected by the CITES and Wildlife Conservation Law of Taiwan, P.O.C." published by the Council of Agriculture in 2000, advises that *Papilio chikae [A. c. chikae]* is distinguishable from *Achillides chikae hermeli* by a less developed lunate pattern on its hindwing. The males appear almost indistinguishable. The difficulties in identifying specimens of the subspecies were highlighted by UK enforcement officers, and it was noted that in many cases specimens are unset and with the wings closed, hiding the distinguishing blue bands of *Papilio chikae.*

Potential benefit(s) of listing for trade regulation
The inclusion of Achillides chikae hermeli in Appendix I would avoid a split-listing with Papilio chikae that is already listed on Appendix I, and avoid any issues of lookalikes when implementing CITES.

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Inclusion of Riverside Swallowtail *Parides burchellanus* in Appendix I

Proponent: Brazil

Summary: The Riverside Swallowtail *Parides burchellanus* is endemic to Brazil. It is found along river margins and in riparian gallery forests within the Cerrado region, a relatively scarce habitat that is influenced heavily by anthropogenic factors. The species was assessed in 2018 as Endangered, considering the discovery of a subpopulation around the Serra da Canastra National Park, and is considered critically endangered in the Brazilian List of Threatened Species.

The species occurs in three districts of eastern Brazil; Distrito Federal, Minas Gerais and Goiás, and is known from no more than four distinct areas or subpopulations. These subpopulations are spatially restricted, and the population overall is severely fragmented and in decline. Individuals have a limited ability to disperse and move only a few hundred metres along rivers, there is no natural connection between the four known subpopulations. The overall population size is unknown, but known subpopulations are very small, with numbers reaching up to 50 individuals, but more commonly around 30 individuals.

The species' area of occupancy is currently estimated to be 120 km² based on known localities and is unlikely to extend beyond 500 km^2. The Cerrado habitat, which *P. burchellanus* relies on exclusively, is under threat from ongoing habitat loss and degradation: Cerrado habitat is estimated to have lost more than half of its original vegetation for farming for crops, cattle-raising activities, energy generation and urbanisation. The only known larval host plant, *Aristolochia chamissonis* is sparsely distributed in small sections along streams associated with fragile and vulnerable environments, which are under threat. The life history traits of *P. burchellanus*, including high habitat specificity and low resilience, make it highly vulnerable to extrinsic factors. Local extinctions have been observed, caused by habitat degradation.

Specimens of *P. burchellanus* appear within international trade with specimens offered for sale online at high prices. *Parides burchellanus* is protected under Brazilian legislation and the capture of specimens is prohibited, therefore trade observed for this species is assumed to be illegal.

Analysis: The total population is unknown for *Parides burchellanus,* although subpopulations are estimated to be very small and persist in only four localities that are spatially restricted, severely fragmented and in decline. Some localised extinctions in subpopulations have been observed. Overall population trend data are unavailable however some subpopulations are inferred to be declining due to habitat degradation. Low resilience to extrinsic factors such as habitat loss and flooding events have been observed for this species. There is evidence for international trade in pinned specimens of *P. burchellanus,* which is believed to be illegal. Although little is known of the population overall, it seems likely that the restricted distribution and fragmented range, very small subpopulations, threats to the habitat and vulnerability due to its specialised niche requirements mean that *Parides burchellanus* meets the criteria in Annex 1 of *Res. Conf. 9.24 (Rev. CoP17).*

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS), text in italics is based on additional information and/or assessment of information in the SS.

Range
Brazil

IUCN Global Category
Endangered *B2ab(iii) (2018) ver 3.1*.

Biological criteria for inclusion in Appendix I

A) Small wild population

The species is categorised under Brazil's National Red List assessment as critically endangered (André et al., *2018).*
Occurrences are clustered into four disjunct areas that are so far apart that genetic exchange is not possible (Grice et al., *2018). Parides burchellanus occurs in spatially limited and fragmented colonies and known populations generally have up to 50 individuals, more commonly fewer than 30 (André* et al., *2018).* The current known subpopulations of the species include Planaltina (Distrito Federal), Planaltina de Goiás (upper Maranhão River), the region around Brumadinho, near Belo Horizonte (State of Minas Gerais) and the area around Serra da Canastra National Park (State of Minas Gerais), *and are discussed in more detail below*:

Brumadinho (State of Minas Gerais)

It is estimated that the local population size in Brumadinho is 10–40 mature individuals over an area of 1.5 km x 30 m. *In 2002, a population discovered along the Catarina River in Brumadinho, Minas Gerais was described as "vigorous" and consisted of more than 50 adults (Mielke* et al., *2004). In 2012 three resident populations were identified in Brumadinho (Beirão* et al., *2012).* This population is particularly vulnerable due to habitat loss and degradation.

Serra da Canastra National Park (State of Minas Gerais)

In 2013, several populations were observed along the São Francisco and Araguari rivers in the State of Minas Gerais, within the municipalities of Campos Altos, Perdizes, Araxá, Ibiá, São Roque de Minas, Tapiraí, Medeiros, Vergem Bonita and Tapira (Grice et al., *2018). In 2015, the species was recorded at a new locality in south-western Minas Gerais, in areas of the upper São Francisco and Araguari River systems, including the Parque National da Serra da Canastra and from the preliminary data collected the number of mature individuals was estimated to be 50 (Bedê* et al., *2015). As there were no previous records of this species present at this locality, data to determine whether this population is in decline are lacking.*

Planaltina (Distrito Federal)

In Planaltina it is estimated the local population size is 10–40 mature individuals over an area of 2.2 km x 30 m. This population is particularly vulnerable due to habitat loss and degradation, especially near the city of Brasília.

Planaltina de Goiás (State of Goiás)

In Planaltina de Goiás Parides burchellanus is found in the upper Maranhão River (André et al., *2018).*

B) Restricted area of distribution

This species exists along river margins and in riparian gallery forests in the Cerrado region in Brazil at altitudes between 800 m and 1,100 m.

This species has a restricted distribution and its current area of occupancy is estimated to be around 120 km^2 based on known localities, and unlikely to extend beyond 500 km^2 in total. Individuals were found in only seven out of 36 localities within its potential area of occupancy. This species moves only a few hundred metres along the rivers being unable to cross large distances. There is no natural connection between the three known subpopulations.

This species exhibits high levels of foodplant and habitat specificity and the only reported larval host plant Aristolochia chamissonis (found in shaded patches of undisturbed stream and riverbank habitats) is sparsely distributed (Bedê et al., *2015) and associated with fragile and vulnerable environments (André* et al., *2018), which could explain* P. burchellanus' *limited distribution.*

There has been a reduction in its Extent of Occurrence (EOO) due to the local extinction of the populations formerly known from the North of São Paulo and Southwest Minas Gerais. Localised reduction of its Area of Occupancy (AOO) is also occurring due to habitat degradation in two of the three known localities. *Extensive surveys have been conducted throughout the species' range and found that the species was absent from several sites that had apparently suitable habitats and presence of the host plant* A. chamissonis *(Beirão* et al., *2012). Of the total 63 surveyed sites, which were selected due to the availability of suitable habitat,* P. burchellanus *was only found to be present in seven sites (Beirão* et al., *2012).*

C) Decline in number of wild individuals

The Cerrado habitat in which the species is found is under immense pressure due to development and transformation of habitat and it has been estimated that the Cerrado has already lost more than half of its original vegetation to farming for crops, cattle-raising activities, energy generation and urbanisation (WWF Brazil, 2011). Furthermore, it is estimated that only 6% of this land area is protected (Françoso et al., *2015).*

There are historical records of this species in areas that have now been altered including; Sobradinho, Anápolis, Carmo do Rio Claro, Matosinhos (Rio das Velhas), Uberaba (Farinha Podre) and Batarais and Bauru. However, due to flooding events, habitat alteration and loss, these populations are now locally extinct (Bedê et al., *2015). There have been no recent records in any of these locations (André* et al., *2018).*

There is evidence that this species has experienced an historical rate of decline and has been extirpated from previously known localities within its range, partly due to the species' low resilience to extrinsic factors, such as flooding and deforestation (Bedê et al.*, 2015). Due to a lack of population data, it is unclear if this species is currently experiencing ongoing population decline (M. Bohm* in litt., *2019). The National Red List assessment (2018) infers that the best studied population located in Brumadinho is declining due to the degradation of gallery forest (André* et al., *2018).*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

In Brazil, this species is listed as critically endangered and its commerce is forbidden. The Brazilian Administrative Authority (IBAMA) has no record of authorized exports in the last ten years. However, the offer for sale of pinned specimens (the only commodity known in international trade) has been observed outside Brazil, on international market sites.

According to the figures provided in the Supporting Statement a total of 19 specimens were observed for sale in international markets in December 2018, with no mention as to whether they were wild-sourced. Advertisements were observed in Spain and Argentina and prices ranged from USD 455 to over USD 3,330 per specimen. This species can be seen advertised online for very large sums of money (between USD 900 and USD 2,800 per specimen) (N. M. Collins in litt., *2019). Considering the high prices that specimens of this species demand within international trade, indiscriminate collection is likely to occur unless safeguards are put in place (M. Bohm,* in litt., *2019).*

Additional Information

Threats

Fragmentation and change of its natural habitat and deforestation of gallery forests, mainly as a result of agriculture, fire and urbanisation, represent the main threats. Pollution of streams is also a strong threat, as the occurrence of the species is associated with water courses.

Parides burchellanus only occurs in the riparian forest, flying near the river and rarely along the forest edges, they have low dispersal rates and high habitat specificity, and some populations have also been observed to exhibit male-biased sex ratios (Beirão et al., *2018). The ecology and life history of this species make it vulnerable to the removal of individuals from the population to supply international trade (M. Bohm,* in litt., *2019).*

Conservation, management and legislation

The capture of these specimens is subject to Brazilian legislation, which allows only research and conservation actions authorised by the environmental agencies (IBAMA). This species is not currently protected by international legislation.

Several actions for conservation were developed from 2010 to 2015 under the National Action Plan for the conservation of Lepidoptera Threatened with Extinction (e.g. conservation of its habitat and water courses, as well as proposals for the creation of protected areas in Brumadinho, etc.). This species is present in three protected areas. *Some localities that contain populations border the Parque Nacional da Serra da Canastra however the level of protection afforded by proximity to this national park is likely to be insufficient (Grice et al., 2018).*

Artificial propagation/captive breeding

There are currently no *ex-situ* breeding programmes for *Parides burchellanus.* Belo Horizonte Zoo had a conservation programme in which it developed *ex-situ* reproduction techniques for *Parides burchellanus,* but the programme was discontinued in 2012. Reintroduction and population supplementation actions in the former areas of occurrence were considered difficult because the habitat is in an advanced stage of degradation.

Other comments

If this proposal to list Parides burchellanus is accepted, regulation of other species may need to be considered in the future—for example, Parides panthonus (also native to Brazil, as well as Guyana, French Guiana and Suriname) is known to be offered for sale internationally and looks similar to P. burchellanus (N. M. Collins, in litt., *2019).*

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Inclusion of Trumpet Trees *Handroanthus* spp*., Tabebuia* spp*.* and *Roseodendron* spp. in Appendix II with annotation #6

Proponent: Brazil

Summary: *Handroanthus*, *Tabebuia* and *Roseodendron* are genera of Bignoniaceae distributed from southern USA to Argentina and Chile, including the Caribbean. There are currently 106 recognised species across the three genera (30 in *Handroanthus*, 73 in *Tabebuia* and 3 in *Roseodendron*). The three genera were previously recognised as belonging to a single genus (*Tabebuia*) but were split in 2007 based on genetic studies, and new species continue to be described. There is considerable confusion in the taxonomy and nomenclature of the three genera with differing names used in the literature and in reported trade.

Species within these genera produce a very hard, heavy and durable wood that is used locally in the construction of houses and bridges, flooring, decking and handicrafts. Internationally it is one of the preferred timbers for decking. The wood is marketed with the same common name (ipê); distinguishing between species and between genera is reportedly difficult even at the microscopic level, and there are no identification guides covering all species.

Handroanthus timbers are some of the most valuable in the market, with prices in Brazil reported to be as high as those achieved historically by Big-leaf Mahogany *Swietenia macrophylla* before commercial exploitation of the latter species was prohibited in the country. Due to their natural low densities, growth rates and shade-intolerant seedlings, Ipê species appear to be particularly vulnerable to logging, even at substantially reduced intensities. Various species have been widely planted throughout the Americas for commercial plantations, reforestation and urban landscaping.

Although no estimates for the global trade in ipê exist, ITTO members reported exports totaling approximately 271,000 m^3 sawn wood (96% from Brazil) and 5,000 m^3 logs (all from Suriname) from 2011-2015. Brazil reportedly exports ipê to 60 countries, the principal importers being the USA and European countries. Trade from Brazil accounted for 93% of ipê sawn wood and ca. 87% of ipê flooring imports by the USA from 2008-2017. All ipê timber production in Brazil derives from natural populations. Potentially high levels of illegal harvest have been reported in the country, and there are concerns over inappropriate management measures including overestimation of sustainable offtakes, although it is unclear what proportion of illegally harvested timber enters international trade.

Handroanthus serratifolius
Of ipê exports reported by Brazil from 2010-2016, 70% (ca. 180,000 m^3) were of *H. serratifolius.* Of the exports of this species, 75% were reported as decking, 16% as sawn wood and the remainder as flooring, clapboards and “other”. The USA and European countries were the major importers. Although annual production of *H. serratifolius* in Brazil increased by 150% from 2012-2017, reaching 220,000 m^3 in 2017, exports of this species decreased from 36,000 m^3 in 2012 to 16,000 m^3 in 2016. In the years for which both production and export figures are available for *H. serratifolius* in Brazil (2012-2016), export volumes were ~16% of production volumes. While this may indicate that domestic use exceeds international trade, a 2008 study reported a relatively low processing efficiency for ipê (42%) suggesting potentially high levels of wastage during processing of exported products. The average yield of this species is estimated at 2.4 m^3/ha. Exploitation in some regions of Brazil has reportedly resulted in significant declines of *H. serratifolius*, with no evidence of long-term population recovery. The species is considered threatened in both Peru and Venezuela; relatively low levels of legal and illegal international trade in the species are reported by Peru, but it is unclear whether this trade has contributed to the reported declines.

Handroanthus impetiginosus
Like *H. serratifolius*, populations of *H. impetiginosus* in parts of Brazil have reportedly suffered significant declines through overexploitation, although reported exports of the species from Brazil were relatively low (1,665 m^3 from 2010-2016). Exports of *H. impetiginosus* are also reported by

Venezuela (20,491 m³ from 2007-2017). The species was categorised globally as Least Concern on the IUCN Red List in 1998, although exploitation was considered to have contributed to population declines, particularly in Brazil. The species is currently categorised as near threatened in Brazil, threatened in Mexico and endangered in Peru.

Other species
Other species reported in international trade include *H. capitatus* (6,000 m^3 sawn wood exported from Suriname from 2011-2015), *H. heptaphyllus* (5,000 m^3 sawn wood exported from Guyana from 2011-2015), *Roseodendron donnell-smithii* (183 m^3 sawn wood and 510 roundwood pieces exported from Mexico from 2010-2012), and *Tabebuia rosea* (exports from Venezuela totalling 29,637 m^3 from 2007-2017 and seizures destined for international export totalling 66 m^3 from 2013-2018). It is not clear whether international trade presents a threat to these species. Deforestation for land clearance is reportedly a threat to certain species in parts of their ranges, such as *H. chrysanthus* in Colombia and *T. rosea* in Mexico, while in other areas reforestation programs are underway.

The proponents seek to include the genera *Handroanthus, Tabebuia* and *Roseodendron* in Appendix II with annotation #6 (logs, sawn wood, veneer sheets and plywood).

Analysis: *Handroanthus*, *Tabebuia* and *Roseodendron* are genera of New World trees comprising over a hundred species, with new species still being described. The timbers of certain species are in high demand both domestically and internationally, and are reportedly some of the most valuable on the market. Woods of the three genera are marketed with the same common name (Ipê); distinguishing between the species and genera is reportedly difficult even at the microscopic level. The most highly traded species based on reported data are *H. serratifolius* and *H. impetiginosus,* which occur in a number of countries from Mexico to Argentina.

While global data on trade are not available, Brazil appears to be the main exporter of ipê, the majority of which is of *H. serratifolius* with 15 other species also exported. There are also reports of illegal ipê harvest and trade taking place in the country. Overexploitation in some areas has reportedly resulted in significant population declines of *H. serratifolius* and *H. impetiginosus* which, like other species in these genera, appear to be particularly vulnerable to logging since they do not regenerate easily. On this basis, *H. serratifolius* and *H. impetiginosus* may meet the criteria for inclusion in Appendix II in Annex 2a of *Res. Conf. 9.24 (Rev. CoP17)*. The remaining species in all three genera would therefore meet the criteria for inclusion in Annex 2b, based on the reported identification difficulties as well as taxonomic and nomenclatural uncertainties.

If this proposal is adopted, it is not clear whether the proposed annotation #6 (logs, sawn wood, veneer sheets and plywood) would cover the main commodities that first appear in trade and drive the demand. Decking and flooring accounted for more than three-quarters of Brazil's reported exports of *H. serratifolius* from 2010-2016, and Brazilian legislation currently prohibits the export of unfinished wood of native species (although large quantities of sawn wood are also reportedly imported into the USA from Brazil). None of the parts and derivatives defined in *Res. Conf. 10.13 (Rev. CoP15*) *Implementation of the Convention for timber species* currently explicitly covers flooring or decking, or refers to the HS code that seems most relevant (HS44.09). It may therefore be necessary to create a new annotation to include "Wood" as defined in HS44.09 and amend *Res. Conf. 10.13* accordingly. Proposal 53 seeks to amend the annotation for *Pericopsis elata* to read "Logs, sawn wood, veneer sheets, plywood, and transformed wood", with transformed wood defined as HS code 44.09, and if that proposal is adopted the same annotation could be applied in this case. This may be considered to be expanding the scope of the proposal, but would be in line with the guidance in *Res. Conf. 11.21 (Rev. CoP17) Use of annotations in Appendices I and II*.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Originally, all species of *Handroanthus*, *Tabebuia* and *Roseodendron* were included in the genus *Tabebuia.* In 2007 it was proposed that *Tabebuia* be divided into three genera. Currently 30 species are recognised for *Handroanthus*, 73 species for *Tabebuia* and 3 species for *Roseodendron*, however new species continue to be described and there is still confusion over nomenclature.

Range

Handroanthus, *Tabebuia* and *Roseodendron* are distributed from southern USA to Argentina and Chile, including the Caribbean.

IUCN Global Category

(Note that certain species assessed on the IUCN Red List are considered synonyms according to the taxonomy referenced in the proposal (the Plant List), details below.)

17 species of Tabebuia/Handroanthus *have been assessed on the IUCN Red List, as follows:*

Endangered:
T. conferta (assessed 2017),
T. elongata (assessed 1998)

Vulnerable:
T. anafensis (considered a synonym of T. myrtifolia in the Plant List; assessed 1998),
T. arimaoensis (assessed 1998),
T. bibracteolata (assessed 1998),
T. dubia (assessed 1998),
T. furfuracea (considered a synonym of T. bibracteolata in the Plant List; assessed 1998),
T. hypoleuca (assessed 1998),
T. jackiana (assessed 1998),
T. lapacho (considered a synonym of H. lapacho in the Plant List; assessed 1998),
T. oligolepis (considered a synonym of T. shaferi in the Plant List; assessed 1998),
T. palustris (assessed 2008),
T. polymorpha (assessed 1998),
T. shaferi (assessed 1998),
T. striata (assessed 1998)

Near Threatened:
T. platyantha (assessed 1998)

Least Concern:
T. impetiginosa (considered a synonym of H. impetiginosus in the Plant List; assessed 1998).

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

Trade in ipê

Most of the *Handroanthus* species are slow-growing heliophytes and require large areas of forest with little competition from other plants to reach the canopy; they are said to be some of the most vulnerable to logging in Amazonian forests because of their natural low density and low growth rates.

It is not possible to maintain timber production at current harvest levels on 30-yr cutting cycles, or even at substantially reduced logging intensities with the current log-and-leave reduced impact (RIL) model, and there appears to be no reason to believe that populations of ipê subjected to logging are any more resilient than those of mahogany (M. Schulze, in litt., *2019). A study by Richardson & Peres (2016) in Pará found no evidence that the post-logging timber species composition and total value of forest stands recovers beyond the first cut, suggesting that the commercially most valuable timber species (including ipê) become predictably rare or economically extinct in old logging frontiers.*

Woods of the three genera are marketed with the same common name (ipê). Most of the *Handroanthus* species produce a very hard, heavy and durable wood which is used locally in the construction of houses and bridges, pavements, decks, exterior woods and handicrafts. Its dark and dense wood is highly valued for residential decks in the USA; *it is described as the internationally preferred timber for decking (Brancalion* et al., *2018). The value of Ipê processed into flooring or decking can reach USD 2,500/m^3 at export ports (Greenpeace, 2018).*

No estimates for the global trade in ipê exist. However, ITTO biennial reports (ITTO, 2015 & 2017) include the following trade in ipê involving ITTO member countries over the period 2011-2015 (trading partner countries were not specified):

- *Exports of ~5,000 m^3 logs (all* H. serratifolius *from Suriname);*
- *Exports of ~271,000 m^3 sawn wood (259,000 m^3 of unspecified species from Brazil (96%), 6,000 m^3* H. capitatus *from Suriname, 5,000 m^3* H. heptaphyllus *from Guyana, 1,000 m^3* H. serratifolius *from Suriname), see Figure 1 below;*
- *Imports of ~2,000 m^3 sawn wood by Brazil (species unspecified).*

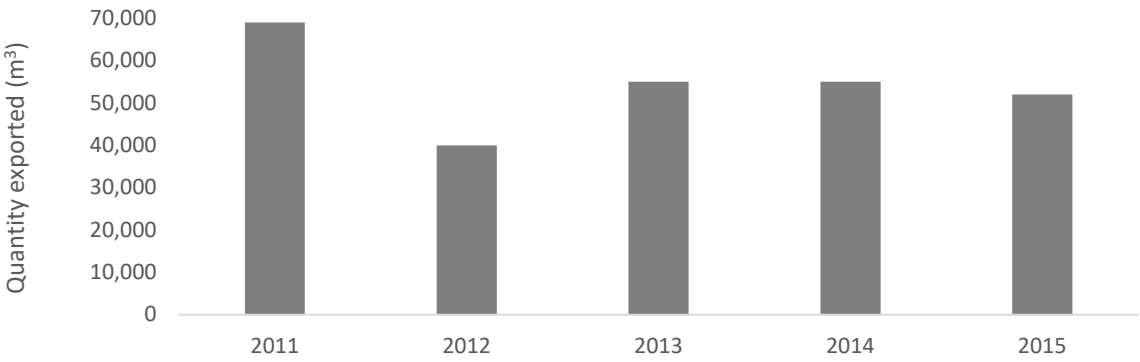

Figure 1: *ITTO reported exports of ipê sawn wood over time from 2011-2015 (ITTO, 2015 & 2017).*

From 2008-2017, the USA reported the import of 276,016 m^3 ipê sawn wood (HS code 44.07.29.01.21: Ipê (Tabebuia spp.) wood sawn or chipped lengthwise, sliced or peeled, whether or not planed, sanded or end-jointed, of a thickness exceeding 6 mm). The top exporter of ipê sawn wood was Brazil (93% of imports), followed by Guyana (3%) and Bolivia (1%). Reported imports of ipê sawn wood over this period were at their highest in 2008, declined by more than 50% in 2009 and subsequently increased overall (see figure below). In 2017 the USA also reported the import of 70,562 m^2 and 1,362 m^3 Ipê flooring (HS code 44.09.22.05.25: ipê (Tabebuia spp.) wood (including strips and friezes for parquet flooring, not assembled) continuously shaped (tongued, grooved, rebated, chamfered, V-jointed, beaded, moulded, rounded or the like) along any of its edges, ends or faces, whether or not planed, sanded or end-jointed). The top exporter was Brazil (87% of imports reported in m^2) followed by Paraguay (11% of imports reported in m^2) (USITC DataWeb, 2019). USA reported imports of ipê sawn wood over the period 2008-2017 were at their highest in 2008, declined by over half in 2009 and subsequently increased overall (see Figure 2).

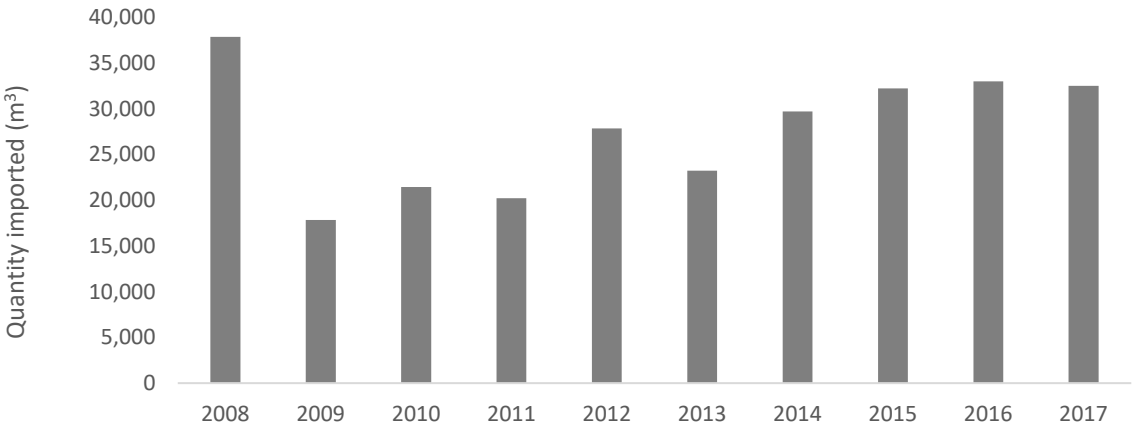

Figure 2: *Imports of ipê sawn wood (HS code 44.07) reported by the USA from 2008-2017 (USITC DataWeb, 2019).*

Available data on international exports indicate that Brazil is the most significant exporter. *Handroanthus* are said to be some of the most expensive species in the market in Brazil, *with prices as high as those achieved historically for Big-leaf Mahogany (*Swietenia macrophylla*) before commercial exploitation was prohibited in Brazil in 2001 (Brancalion* et al., *2018);* the authors suggest that Ipê is the "new Big-leaf Mahogany*". The average price of Ipê sawn wood exports in the period 2011-2015 was USD 824/m^3 (ITTO, 2015 & 2017). There was a 500% increase from 1998 to 2004 in ipê timber exports from the Brazilian Amazon (Schulze* et al.*, 2008).* Ipê exports from Brazil are reported to have been almost 256,000m^3 between 2010 and 2016, 70% of which were reportedly *H. serratifolius* (see table below). *Most of the ipê timber harvested in Brazil is reported as H. impetiginosus and H. serratifolius (Schulze et al., 2008).* Ipê wood is exported from Brazil to 60 countries; the main importers are the USA (28% of exported volume, *not clear over what time period*) and European countries. In Brazil all ipê wood comes from natural forests, since there are no plantations in the country*. The export of unfinished timber of species native to Brazil (i.e. destined to be processed abroad) is prohibited according to Normative Instruction 15/2011 (IBAMA, 2011), amended by Normative Instruction 13/ 2018 (IBAMA, 2018).*

Between 2010 and 2016, 16 species of ipê were reportedly exported from Brazil (see Table 1).

Table 1: ipê exports from Brazil by species from 2010-2016, as reported in the Supporting Statement (*table below uses the accepted names from the Plant List*):

Species	Volume (m^3)
Handroanthus serratifolius	180,110
Handroanthus capitatus	2,887
Handroanthus incanus	2,243
Handroanthus impetiginosus	1,665
Handroanthus heptaphyllus	1,565
Tabebuia ochracea	1,439
Handroanthus vellosoi	1,436
Handroanthus albus	1,373
Handroanthus chrysotrichus	898
Handroanthus barbatus	316
Tabebuia cassinoides	223
Tabebuia aurea	128
Handroanthus umbellatus	115
Handroanthus chrysanthus	50
Tabebuia angustata	24
Tabebuia roseoalba	24
Tabebuia spp. (assumed to include *Handroanthus spp.)*	61,227

Illegal trade in ipê

In the study by Brancalion *et al.* (2018), ipê were by far the species with the strongest evidence of potential fraud in logging permits in Brazil. *Only 61% of the 152 trees identified as* Handroanthus spp. *in logging permits were confirmed during field checking of logged forests. Ipê trees are quite easy to identify so misidentification can be attributed to fraud. In addition, the diameter of real ipê trees was frequently overestimated. Inventing trees, duplicating tree numbers, exceeding allowed harvest rates, and felling trees in forbidden zones were also observed, although less frequent. In all of the management areas investigated, there was evidence that the ipê timber volume reported on the permit could not have been produced from that area alone while following harvest regulations. The apparent substantial overestimation of timber volumes in logging licenses has the potential to significantly impact species like Ipê (Brancalion* et al., *2018).*

*Greenpeace-Brasil (2015), in collaboration with SEMA and Brazil's Public Prosecutor's Office, carried out a systematic review of all 1,325 extant management plans in Pará between 2006 and 2013 to assess the extent to which timber laundering occurred. In total, 746 (56%) plans listed ipê (*H. serratifolius*) in their inventories and approximately 14% overestimated volumetric offtakes (3,000 m^3 per concession or 60% above the species average of 2.4 m^3/ha). Subsequent in situ inspections revealed a range of fraudulent activities including illegal timber laundering through illegitimate plans where authorised areas showed no signs of logging. Electronic credit documents were crediting timber well in excess of what had been authorised by management plans before round logs were transferred to sawmills exporting timber worldwide (Greenpeace, 2015). Greenpeace-Brasil published a further report in 2018 highlighting weaknesses in the timber licensing regime in Pará with indiscriminate and*

illegal logging of ipê, driven by the high value of processed products of ipê wood (decking and flooring) in international markets (Greenpeace, 2018).

Brazil reported seizures of ipé destined for international export totalling 350 m^3 in 2016 and 475 m^3 in 2018.

Handroanthus serratifolius
Range: Antigua and Barbuda, Bahamas, Barbados, Bolivia, Brazil, Colombia, Cuba, Dominica, Dominican Republic, French Guiana, Grenada, Guadeloupe, Guatemala, Guyana, Haiti, Honduras, Jamaica, Martinique, Mexico, Montserrat, Netherlands Antilles, Nicaragua, Panama, Paraguay, Peru, Puerto Rico, St Kitts and Nevis, St Lucia, St Vincent and the Grenadines, Suriname, Trinidad and Tobago, USA, Venezuela, Virgin Islands (US); Exotic – India, Kenya (Orwa et al., *2009).*

Brazil: Forest inventories conducted in the state of Pará, in the Brazilian Amazon, recorded densities of between 0.2 and 0.4 *H. serratifolius* trees/ha with DBH ≥ 50 cm. The average yield of this species is estimated at 2.4 m^3/ha. The wood production of *H. serratifolius* in Brazil was 220,000 m^3 in 2017, which was 150% higher than in 2012.

As noted above, available data on international exports indicate that Brazil is the most significant exporter of ipé. Exports of *H. serratifolius* from Brazil from 2010 to 2016 totalled over 180,000 m^3, which corresponds to 70% of total exports of ipê for that period (255,723 m^3). Of these exports 75% were reported as decking, 16% sawn wood and the remainder flooring, clapboards or "other". The main importers were the USA (29%), France (17%) and Belgium (10%).

Production and export volumes of *H. serraifolius* from Brazil are summarised in Figure 3 below *(production data not available prior to 2012; export data not available for 2017)*. The smallest volume was exported in 2010 (~10,000 m^3) and the largest in 2012 (36,000 m^3); export volumes then decreased up to 2016. Reported exports of *H. serratifolius* from Brazil were around 16% of total production over the period 2012-2016 (~135,000 m^3 and 855,000 m^3 respectively) and around 8% of reported production in the country in 2016 (~16,000 m^3 and 194,000 m^3 respectively). *Schulze et al. (2008) cite a processing efficiency of 42% for ipê and indicate that only 36% of processed ipê wood meets export standards; the gap between production and exports is therefore at least partly accounted for by inefficient and poor quality processing, the latter potentially indicative of a domestic market for lower quality products.*

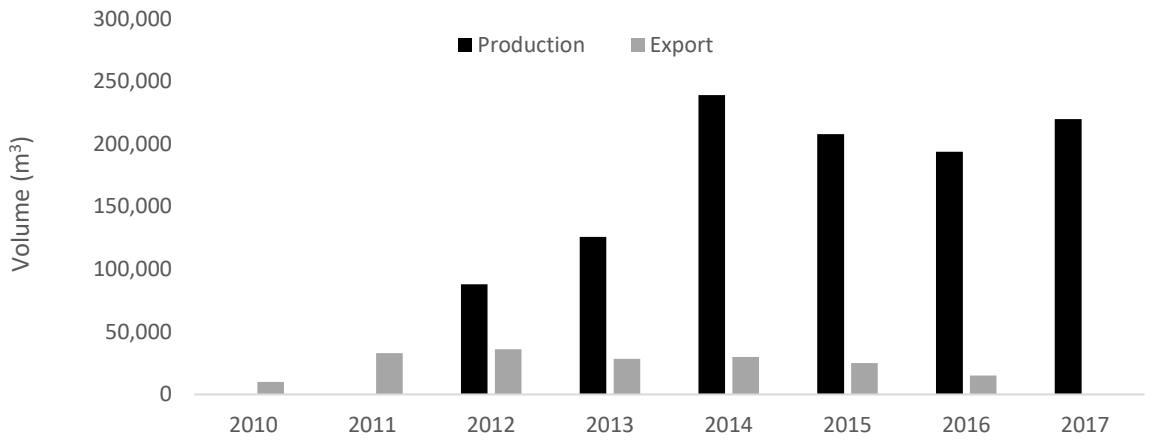

Figure 3: Production and export volumes of *H. serraifolius* from Brazil reported in the Supporting Statement from 2010-2017 *(production data not available prior to 2012; export data not available for 2017).*

A 2008 study found that all populations of *H. serratifolius* in the forests of north-eastern Brazil showed drastic population declines, with no evidence of long-term population recovery. At best, only one individual of one of the two species *H. serratifolius* and *H. impetiginosus* in each 10 ha could reach adult size, and this could take a century or more. The authors suggest that cutting cycles applied for *H. serratifolius* in the western Brazilian Amazon may not be sustainable. *Shade intolerant seedlings, size class distributions skewed towards large adults, and poor colonisation of logging gaps make populations of this species unlikely to rebound from logging (Schulze, 2003).*

Bolivia*: Bolivian forest inventory data indicate that* H. serratifolius *occurs at relatively high densities (0.45 trees/ha (Justiniano* et al., *2000). However, population structures are skewed towards large, very old adults, meaning few young trees are present in stands. This is a pattern typical for light-demanding long-lived timber*

species in the Amazon, including mahogany, and is frequently cited as a major limitation for sustainable timber production (Schulze et al., *2008).*

Peru: *H. serratifolius* is assessed nationally as threatened. During recent decades *H. serratifolius* has entered the market as the preferred wood for decking. The country reported exports of this species totalling 1,131 m^3 from January 2016- March 2018. The biggest importers were China, the Dominican Republic, the USA and France. The country reports Ipê seizures destined for international export of 119 m^3, 14.96 kg and 4,738 pieces between 2011 and 2017 (species unspecified).

Suriname: *ITTO biennial reports for the period 2011-2015 include exports from Suriname of 5,000 m^3 logs and 1,000 m^3 sawn wood of* H. serratifolius*, destination unspecified (ITTO, 2015 & 2017).*

Venezuela: *H. serratifolius* is assessed nationally as threatened. The species has shown a marked decrease in its natural populations as a consequence of the popular demand for wood for the manufacture of handicrafts in the states of Lara and Falcón. *No exports of H. serratifolius from the country are mentioned*.

Handroanthus impetiginosus
Range: Argentina, Bolivia, Brazil, Colombia, Costa Rica, El Salvador, French Guiana, Guatemala, Honduras, Mexico, Nicaragua, Panama, Paraguay, Peru, Suriname, Venezuela (IPNI, 2019).

Brazil: As with *H. serratifolius*, all populations of *H. impetiginosus* in the forests of north-eastern Brazil showed drastic population declines in a 2008 study, with no evidence of long-term population recovery. *Shade intolerant seedlings, size class distributions skewed towards large adults, and poor colonisation of logging gaps make populations of this species unlikely to rebound from logging (Schulze, 2003).* Exports of *H. impetiginosus* from 2010-2016 totalled 1,665 m^3. *H. impetiginosus is assessed as near threatened in Brazil with overexploitation reported as a threat (CNCFlora, 2012).*

Bolivia: *As with H. serratifolius, Bolivian forest inventory data indicate that* H. impetiginosus *occurs at relatively high densities (2.5 trees/ha; Justiniano* et al., *2000). However, population structures are skewed towards large, very old adults, meaning few young trees are present in stands. This is a pattern typical for light-demanding long-lived timber species in the Amazon, including mahogany, and is frequently cited as a major limitation for sustainable timber production (Schulze* et al., *2008).*

Mexico: Assessed nationally as threatened.

Peru: Assessed nationally as endangered.

Venezuela: From 2007 to 2017 Venezuela exported 20,491 m^3. During this period, exports decreased from a maximum of 5,570 m^3 in 2007 to a minimum of 23 m^3 in 2017.

Handroanthus capitatus
Range: Bolivia, Brazil, Colombia, French Guiana, Guyana, Peru, Suriname, Trinidad and Tobago, Venezuela (IPNI, 2019).

Suriname: *ITTO biennial reports for the period 2011-2015 include exports from Suriname of 6,000 m^3 sawn wood of H. serratifolius, destination unspecified (ITTO, 2015 & 2017).*

Handroanthus chrysanthus
Range: Belize, Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Guyana, Honduras, Mexico, Nicaragua, Panama, Peru, Trinidad and Tobago, Venezuela (IPNI, 2019).

Ecuador: In a study of forest species most used in the southern region of Ecuador between 2014 and 2015, *H. chrysanthus* was one of the most commercialised species. Data reported through the Provincial Department of the Environment of Guayas show exports of 270 m^3 in 2012. The population has recovered mainly due to management actions, including a declared ban on exploitation in 1978.

Mexico: Assessed nationally as threatened.

Handroanthus heptaphyllus
Range: Argentina, Bolivia, Brazil, Paraguay (IPNI, 2019).

Guyana: *ITTO biennial reports for the period 2011-2015 include exports from Guyana of 5,000 m³ sawn wood of H. heptaphyllus, destination unspecified (ITTO, 2015 & 2017).*

Roseodendron donnell-smithii
Range: Colombia, El Salvador, Guatemala, Honduras, Mexico, Venezuela; Introduced: Ecuador, Puerto Rico (IPNI, 2019).
Mexico: *Described as an important timber yielding species in Mexico with high potential to trade on the international market (Colín-Urieta et al., 2015).* From 2010-2011 Mexico exported 183.4 m³ sawn wood to Guatemala, and in 2012 exported 510 roundwood pieces to Honduras. *In 1989 it was reported to be exported from Guatemala and Mexico to the USA (Little and Skolmen, 1989).*

Tabebuia cassinoides
Range: Brazil (IPNI, 2019).
Brazil: *This endemic species is assessed nationally as endangered (CNCFlora, 2012) noting that its habitats are severely fragmented and that although protected by conservation units, it is widely exploited for timber. It has light wood considered the second best wood in the world for pencil production, and is also used to make clogs, musical instruments, toys and for crafts. Currently, there are few viable subpopulations for commercial exploitation (CNCFlora, 2012).*

Tabebuia rosea
Range: Belize, Colombia, Costa Rica, Ecuador, El Salvador, French Guiana, Guatemala, Guyana, Honduras, Mexico, Nicaragua, Panamá, Venezuela. Introduced to: Brazil, Cayman Islands, Cuba, Dominican Republic, Gambia, Jamaica, Leeward Is., Puerto Rico, Trinidad-Tobago, Venezuelan Antilles, Windward Is. (IPNI, 2019).

Mexico: *T. rosea* was highly exploited in the Yucatan Peninsula, so its stocks are low. Cutting cycles in Mexico have favoured the renewal of all important commercial species, except *T. rosea*. *Exploitation is reportedly primarily for local use (E. Martínez Salas,* in litt., *2019)*.

Venezuela: From 2007 to 2017 there were exports of 29,637 m³. The country reports seizures destined for international export totalling 66 m³ between 2013 and 2018.

Other species
H. chrysotrichus: critically endangered in Bolivia
H. guayacan: threatened in Costa Rica
H. lapacho: threatened in Argentina, endangered in Bolivia
H. pulcherrimus: highly threatened across its whole distribution in South America
H. arianeae, H. riodocensis, H. spongiosus: endangered in Brazil *(CNCFlora, 2012)*
H. botelhensis, H. cristatus, H. selachidentatus: near threatened in Brazil *(CNCFlora, 2012)*
T. conferta: endangered in Haiti
T. obovata: threatened in Cuba, endangered in Dominican Republic
T. palustris: threatened in Colombia, Costa Rica and Panama
T striata: threatened in Colombia and Panama
T. bullata, T. crispiflora, T. dominguensis, T. maxonii, T. ophiolithica, T. paniculata, T. ricardii, T. vinosa, T. zanonii: endangered in the Dominican Republic
T. heterophylla, T. lepidota, T. pulverulenta, T. sauvallei: critically endangered in Cuba
T. bibracteolata, T. clementis: endangered in Cuba
H. billbergii, T. arimaoensis, T. bahamensis, T. berteroi, T. calcicola, T. caleticana, T. densifolia, T. elongata, T. inaequipes, T. pinetorum, T. polymorpha: threatened in Cuba
T. jackiana: vulnerable in Cuba
T. brooksiana, T. elegans: near threatened in Cuba.

Additional Information

Threats
Over-exploitation is considered a threat, as well as habitat loss. Based on satellite imagery, the total area deforested in the Brazilian Amazon in 2004 was 2.8 million ha. The annual rate of deforestation between 2012 and 2013 was 5.9 million ha, an increase of 28% compared to 2011. Deforestation was driven mainly by the demand for agricultural land. In Colombia, the reduction of forest fragments caused by the expansion of areas for agricultural and livestock use has restricted *H. chrysanthus* to the drier transition zone of xerophytic scrub vegetation in the south of the country. In Michoacán, Mexico, natural populations of *T. rosea* have decreased considerably due to anthropogenic factors, mainly deforestation for the construction of human settlements combined with obtaining wood, contributing to the reduction of their habitat.

Conservation, management and legislation
There are broad bans/restrictions on timber logging/export in several range States, examples include:
*- **Brazil:** the export of unfinished timber of native species (i.e. destined to be processed abroad) is prohibited according to Normative Instruction 15/2011 (IBAMA, 2011), amended by Normative Instruction 13/ 2018 (IBAMA, 2018).*
*- **Bolivia:** export of unprocessed forestry products is subject to restrictions and highly regulated, mainly through forest certification (from 1996 onwards, last updated 2016; Forest Legality Initiative, 2016).*
*- **Ecuador:** there is an export ban on roundwood, except in limited quantities for scientific and experimental purposes, and semi-finished forest product exports are allowed only when "domestic needs and the minimum levels of industrialization have been met" (from 2005 onwards, last updated 2016; Forest Legality Initiative, 2016).*
*- **Peru:** there is an export ban on logs and forest products "in their natural state" except when they originate from nurseries or forest plantations, and if they do not require processing for final consumption (from 1972 onwards, last updated 2016; Forest Legality Initiative, 2016).*

In Ecuador, in vitro cultures are being used to recover endangered species of *Handroanthus*. Populations of *H. chrysanthus* and *H. billbergii* have been recovered mainly by management actions, among them the declared ban on exploitation in 1978. In 2006, Venezuela decreed a ban on exploitation of *T. spectabilis*.

Artificial propagation/captive breeding
Various species in this genus are grown in nurseries for forest plantations, reforestation and urban planting throughout the Americas. In Panama, *T. rosea* was experimented with to reforest degraded areas and the yield was good in all sites. In Venezuela there are plantations of this species in the Barinas and Monagas States. In Jamaica *T. rufescens* and *T. rosea* are commonly cultivated. In Mexico there are 62,736 ha of plantations of ipê of several species, mostly *T. rosea* (48,748 ha) which is managed in commercial plantations, as well as being used to enrich secondary forests and degraded paddocks. There are no ipê plantations in Brazil.

Implementation challenges (including similar species)
Due to the great similarity between the woods of the different species and genera, they are marketed with the same common name (ipê). The quantitative characteristics of the wood anatomy can present variations between individuals of a species and even within the same individual, so distinguishing species is very difficult. *A clear differentiation of the individual species is not possible on either the macroscopic or microscopic level (G. Koch,* in litt., *2019).* There are no identification guides for all species of the three genera. *While the proposal outlines distinguishing characters between the three genera, a clear differentiation between the genera* Handroanthus *and* Tabebuia *is not considered possible, especially for non-experts. The species of the genus* Roseodendron *may be distinguishable from* Handroanthus *and* Tabebuia *on the microscopic level (with relevant reference samples and expertise). However, based on the very complex taxonomy of the three genera with many synonyms, a clear and assured differentiation may be difficult in practice (G. Koch,* in litt., *2019).*

The following species are commonly confused with ipê in trade, although they can be differentiated using microscopic characters (G. Koch, in litt., 2019). None of these species are currently included in the CITES Appendices; all are present in range States where the three proposed genera are also present:
Acosmium spp.*: occur in Bolivia and Brazil (Rodriguez & Tozzi, 2009)*
Leptolobium spp*. (Lapachillo, lapachin): occur from Mexico to Argentina (Rodriguez & Tozzi, 2012)*
Dicorynia guianensis Amsh.*: occurs in Suriname and French Guiana (ITTO, 2019)*
Dicorynia paraensis *Benth. (Angélique, basralocus): occurs in Brazil, Colombia and Venezuela (Canteiro, 2018)*
Dipteryx odorata *(Aubl.) Willd.: occurs in Brazil, Colombia, French Guiana, Guyana, Suriname and Venezuela (Requena Suarez, 2017)*
Dipteryx alata *Vog. (Cumarú): occurs in Brazil (World Conservation Monitoring Centre, 1998).*

Potential risk(s) of a listing
Trade shifts to other non-listed species, as appears to have affected these genera following the listing of mahogany (Brancalion et al., *2018).*

Other comments
It is not clear whether the proposed annotation #6 (logs, sawn wood, veneer sheets and plywood) would cover the main commodities that first appear in trade and drive the demand. The end use of much of the export (at least to the USA) is reported to be decking and flooring. The USA uses two HS codes for ipê imports: HS44.07 (wood sawn or chipped lengthwise, sliced or peeled, whether or not planed, sanded or end-jointed, of a thickness exceeding 6 mm) and HS44.09 (wood (including strips and friezes for parquet flooring, not assembled) continuously shaped (tongued, grooved, rebated, chamfered, V-jointed, beaded, moulded, rounded or the like) along any of its edges, ends or faces, whether or not planed, sanded or end-jointed). The definition of sawn wood

in Res. Conf. 10.13 (Rev. CoP15) Implementation of the Convention for timber species *already refers to HS44.07, but no part or derivative defined in Res. Conf. 10.13 explicitly explicitly covers flooring or decking, or refers to HS44.09 (the HS code that seems most relevant). Proposal 53 seeks to amend the annotation for* Pericopsis elata *to read "Logs, sawn wood, veneer sheets, plywood, and transformed wood", with transformed wood defined as HS code 44.09, and if that proposal is adopted the same annotation could be applied for the listing of these taxa. This may be considered to be expanding the scope of the proposal, but would be in line with the guidance on annotations in Res. Conf. 11.21 (Rev. CoP17) paragraph 6, which recommends that "i) controls should concentrate on those commodities that first appear in international trade as exports from range States; these may range from crude to processed material; and ii) controls should include only those commodities that dominate the trade and the demand for the wild resource." Ensuring the most appropriate annotation is adopted would avoid the need to amend it through a further proposal at a subsequent CoP.*

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Inclusion of Mulanje Cedar *Widdringtonia whytei* in Appendix II

Proponent: Malawi

Summary: The Mulanje Cedar *Widdringtonia whytei* is a slow growing, coniferous tree in the cypress family, endemic to the Mount Mulanje Massif in south-eastern Malawi, which occurs over 650 km^2. It can reach a height of 40 m and over 1 m in diameter, taking 80–100 years fully to mature. Much of its habitat is found in the Mount Mulanje Forest Reserve. It historically grew at 1500–2200 m above sea level. There are around 70 ha of plantations on Zomba Mountain and another 80 ha in the large timber plantations of the Viphya Plateau, which may include a mix of *W. whytei* and *Widdringtonia nodiflora.*

Widdringtonia whytei faces numerous threats, the most serious of which are changing fire regimes, fuelwood collection, illegal logging, invasive tree species and conifer aphids.

Widdringtonia whytei was assessed as Critically Endangered in 2011 on the basis that threats were likely to cause a decline of more than 80% by 2030. In 2014, a Forest Department survey found 38,138 mature, living *W. whytei* (with a further 25,609 standing dead individuals) but by 2017 only seven mature *W. whytei* trees were found, all of which had been felled by 2018. There are no reproductively mature trees on the Mulanje Mountain. The remaining population is thought to comprise of seedlings that have been planted since 2017 as part of a major restoration project. Due to low regeneration and recruitment, the success of the project will not be known for years to come. Plantation forestry has been conducted in other areas of Malawi with limited success.

The export of native hardwood logs has been banned since 2008 and *W. whytei* is listed as a protected species in Malawi. Licences were only meant to be issued for salvage logging of dead trees, but illegal logging that targeted the remaining large, living trees escalated throughout the period 2007–2018. While *W. whytei* has been commercially exploited for over a century, it is unclear whether international trade or national utilisation has driven the recent decline.

The proposal is to list the species *W. whytei* in Appendix II without annotation.

Analysis: There are no mature *Widdringtonia whytei* trees remaining in its natural habitat, the last remaining seven having been felled by 2018. The species can be considered to be commercially extinct in the wild and therefore meets the biological criteria for listing in Appendix I already. Seedlings planted since 2017 are unlikely to mature for tens of years and therefore any trade in this species from the wild is not likely in the near future. Appendix II listing is therefore unlikely to have any significant conservation impact.

It is possible that trade in *W. whytei* from plantations may occur as plantation forestry has been attempted for over a century with limited success.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

Widdringtonia whytei *or Mulanje Cedar is one of four species in the genus* Widdringtonia (Widdringtonia nodiflora, Widdringtonia schwarzii, Widdringtonia wallichii *and* Widdringtonia whytei*).*

Widdringtonia whytei was long thought to be the same species as the more widespread *Widdringtonia nodiflora*, which has a more narrow and multi-stemmed growth form. However, genetic analysis at the University of Cape Town has conclusively shown that they are distinct species. Both species grow on Mount Mulanje.

Range

Malawi

IUCN Global Category
Critically Endangered *A4acde + B2ab (i,ii,iii,iv,v) (2011) ver 3.1*

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

Biology
Widdringtonia whytei is a coniferous tree in the cypress family that can reach 40 m in height and over 1 m in diameter. The tree is wide-crowned and often branchless up to about 21 m. Its bark is spongy and thickens with age, splitting along longitudinal cracks. *Widdringtonia whytei* wood has copious amounts of aromatic resin whose resistance to fungal rot, decay and insects make its yellow-white timber highly valuable. Its basic density is 385–430 kg/m^3.

Widdringtonia whytei grows relatively slowly, particularly in comparison to the non-native conifer species with which it now competes on the mountain. It takes 80–100 years fully to mature but will eventually become a dominant or co-dominant tree in its ecosystem if canopy out-competition, natural or human disturbances do not intervene.

Population size
Widdringtonia whytei is endemic to the Mount Mulanje massif in south-eastern Malawi, whose total area covers approximately 650 km^2. Much of this area is gazetted as the Mount Mulanje Forest Reserve (MMFR) and known locally as *chilumba mu mlengalenga*, "island in the sky". *Widdringtonia whytei* historically grew across the upper reaches of the MMFR (1500–2200 m above sea level).

The most recent ecological baseline survey, conducted in 2017, found that "Mulanje Cedar is considered to be practically extinct on Mulanje mountain. It is very likely that the small number of sizeable standing individuals that remain will be gone before the end of 2017." This survey found only seven mature individual *Widdringtonia whytei* trees, clustered together in a single inaccessible gully (for purposes of the survey, mature individuals were those capable of producing seed). Out of 34 20 m^2 plots surveyed, 25 plots had fewer than 20 living individuals, most of which were young seedlings recently planted by a restoration project in the Forest Reserve. Eight of these plots had no living individuals. As of 2018, the seven reproductively mature standing individuals had been felled.

Population trends
The IUCN Red List assessment (assessed in 2011) stated that a population decline of over 80% was likely by 2030 and considers the species Critically Endangered. The actual decline has been even more drastic. In 2007, densities of between 41 and 131 stems/ha of trees above 5 cm diameter, and 78,159 m^3 of standing live volume were reported. In 2014, a Forest Department survey found 38,138 mature, living cedar trees (and a further 25,609 standing dead trees). But by 2017, field surveys found only seven mature living trees. *This decline was caused by lawlessness, resulting in illegal harvesting (T. Chanyenga and D. Nagoma,* in litt., *2019).*

Trade
Widdringtonia whytei timber is highly valued in Malawi, as its light-to-moderately-heavy yellow-brown wood is easy to work with and is decay- and insect-resistant. It is a highly desirable species for construction and boat building, and has been commercially harvested for over a century. The British colonial regime established Mount Mulanje Forest Reserve largely to manage its extraction.

The Malawi Department of Forestry currently licenses only the utilisation of dead Widdringtonia whytei *trees and is reviewed annually, averaging 20 pit-sawyers in 2007 (Bayliss* et al., *2007).* As recently as 2007, the government of Malawi issued harvesting licences for cedar to build 400 plank boats. The species is also preferred for furniture, roof shingles (weathering to a silver-grey), panelling, flooring and handicrafts such as walking sticks sold to tourists. An oil called "Mulanje tar" can be distilled and used as a preservative. *Widdringtonia whytei* has been traded as logs, sawn wood, or smaller dimensional pieces of timber salvaged from dead trees. Derivative products can also be made from its resin. *It is reported to be prized for use in making furniture, chests and curios sold to tourists (Hecht, 2008).*

Widdringtonia whytei *has become a prestige wood with a high scarcity value: one large tree might have fetched a price of £1000 (USD 1300, GBP 1=USD 1.3 calculated on 28/01/19) in 1992 (Chapman, 1992, reported in Pauw and Linder 1997) (valued at £2033 in 2018 (USD 2643, GBP 1=USD 1.3 calculated on 28/01/19)).*

More recently, stockpiled wood is listed on a "Rare Woods" website in South Africa priced at R46,120/m^3 (USD 3,385/m^3, ZAR 1=USD 0.07 calculated on 28/01/19), available in 38 mm and 50 mm thickness (Rare Woods SA, 2018).

Additional Information

Threats

Changing fire regime: *Widdringtonia whytei* regeneration is linked to periodic fire that kills off shade canopy and exposes new mineral soil. But these natural fires are usually caused by lightning during the rainy season and are less damaging than the now-frequent fires caused by humans for various reasons. Many are set during the hot dry season to renew grasses for local livestock grazing. Hunting flush fires and crop waste fires are also common and can spread out of control, particularly on the drier northeastern slopes of the mountain adjacent to communities. Loggers set fires to expose remaining pieces of cedar in already-cut areas. Such fires damage or kill the adult trees and, critically, also have a heavy impact on regenerating seedlings.

Invasive tree species: Mexican Pine *Pinus patula*, originally planted by the colonial administration as a nurse crop for *Widdringtonia whytei*, turns out to grow much faster and shade it out. *Pinus patula* is also a pioneer species co-adapted with fire. It has taken over areas of the mountain to the exclusion of other species including *Widdringtonia whytei*.

Conifer aphids: The Giant Cypress Aphid *Cinara cuppresivora*, originally from Europe and North America was first recorded in Malawi in 1986. This aphid attacked and killed many *Widdringtonia whytei* trees in the 1980s and 1990s, before a parasitic wasp was released as a biological control. It has not been eradicated and remains a factor in the high mortality levels.

Other threats: Deposits of around 30 million t of bauxite have been found over about one third of the Mulanje plateau. While large-scale open cast mining has not yet occurred, the possibility remains and would completely destroy any cedar forest habitat in these areas. Most recently, rare earth minerals needed by computer and cell phone technologies have been found on the plateau.

Conservation, management and legislation

Native hardwood log exports have been banned since 2008. There is no legal export of *Widdringtonia whytei.*

Widdringtonia whytei was listed as a Protected Species in a revision to the National Parks and Wildlife Protected Species Act which was revised and published in September 2017. Under this Act one could receive a jail sentence of up to 30 years if found to be illegally possessing *W. whytei*.

Previous plantation or restoration attempts have been seriously hampered by a poor understanding of ecology, pathology and horticulture, although the tree has been successfully planted at a small scale elsewhere in Malawi as well as in botanic gardens. The “Save our Species” project planted 220,000 seedlings between 2003 and 2010, but almost all were lost to wildfire. Low seedling survival has been a consistent problem in the nursery as plants attain 5–10 cm, and survival dropped off again once they were out planted to the natural habitat. *A similar pattern of poor survival has been found with planting the Cederberg Cedar* Widdringtonia wallichii *in the South Western Cape of South Africa (N. Allsopp,* in litt., *2019).*

Currently an initiative funded by the Darwin Initiative and implemented by Botanic Gardens Conservation International, in partnership with the Mulanje Mountain Conservation Trust and the Forestry Research Institute of Malawi, is attempting a major *Widdringtonia whytei* replanting project within appropriate habitat of MMFR. Ten community nurseries were established around the base of the mountain and apparently traditional authorities were consulted and involved in project implementation. The project trained nursery supervisors and extension staff in horticultural protocols, refinement of which is ongoing. Trial plots were also set up across Malawi to “test growth limits and identify optimal growing conditions for Mulanje Cedar”. The project has planted 325,000 seedlings and its goal is to plant another 250,000 on the mountain by 2019 while selling 250,000 seedlings commercially to remove pressure from the Mount Mulanje population.

Artificial propagation/captive breeding

Plantation forestry of *Widdringtonia whytei* has been attempted for over a century with limited success. The species’ growth is far slower than exotic conifers like *Cupressus lusitanica* and *Pinus patula*. It also appears that in some cases *Widdringtonia whytei* and *W. nodiflora* have been mixed together in these plantations. There are 66 ha of plantation on Zomba Mountain and another 76 ha in the large timber plantations of the Viphya Plateau.

It is reported to be found in various botanical gardens around the world (Kenya, Tanzania, Indonesia and New Zealand) (Shaw & Smith, 2017).

Implementation challenges (including similar species)

Widdringtonia whytei was long thought to be the same species as the more widespread *W. nodiflora*, which has a more narrow and multi-stemmed growth form. However, genetic analysis at the University of Cape Town has

conclusively shown that they are distinct species. Both species grow on Mount Mulanje. Widdringtonia nodiflora *is widespread in southern Africa and assessed as Least Concern (ver 3.1, assessed 2011), although it can be readily coppiced it is probably only used for firewood locally, with no commercial use reported for this species (Farjon, 2013).*

Traditional chiefs have significant authority over harvest rights in their villages or territories. Their decisions may not always be in line with the national government's policies.

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Deletion of North Indian Rosewood *Dalbergia sissoo* from Appendix II

Proponents: Bangladesh, Bhutan, India and Nepal

Summary: North Indian Rosewood *Dalbergia sissoo* is a fast-growing perennial tree, native to Afghanistan, Bangladesh, Bhutan, India, Islamic Republic of Iran, Iraq, Myanmar, Nepal and Pakistan, and is also widely introduced, especially in Africa and Asia. In some regions it is considered invasive. The population size is not known, and although disease has impacted both wild and cultivated populations in a number of range States, the species' high regeneration and growth rate provides resilience to this threat. In Bangladesh, India, Nepal and Pakistan the species is widely cultivated, and has also successfully naturalised within some new areas, following afforestation programmes. *Dalbergia sissoo* is primarily harvested for its timber, which is used for a wide range of products including handicrafts and furniture. It has become one of the most widely utilised plantation tree species in the Indian subcontinent where it is economically important for its value in forestry, agroforestry and horticulture.

The genus *Dalbergia* was listed in Appendix II at CoP17 (2016) with annotation #15, except for the species already listed in Appendix I. It was argued at the time of the proposed listing that only some *Dalbergia* species met the criteria in Annex 2a, but enforcement and customs officers who encountered specimens of *Dalbergia* products would be unlikely to be able to distinguish between the various species of *Dalbergia* reliably so the whole genus should be listed. In 2017 the predominant commodities of *D. sissoo* reported in international trade were carvings (~5.8 million kg) and wood products (735,000 items plus ~80,000 kg), and most were reported as pre-Convention (although there was some trade reported as from artificially propagated and wild sources). The majority of trade was from India, and European countries (particularly Germany) and the USA were the major importers.

Many experts acknowledge that, without the use of technology, it is difficult for non-experts readily to identify *Dalbergia sissoo* once made into finished products, and these appear to be the predominant form in which *D. sissoo* is traded. While technological methods to identify *D. sissoo* exist, they require expertise and/or equipment not currently available on a global scale.

A proposal to amend annotation #15 has also been submitted (CoP18 Prop. 52). Should this be accepted, trade in some items, including products containing less than 500 g of wood and musical instruments, would be exempted from controls. This may have a significant impact depending on what proportion of India's carvings contain less than 500 g of wood; India raised particular concerns over the impact that the listing of *Dalbergia sissoo* has had on their handicraft industry.

Analysis: Wild populations of *Dalbergia sissoo* are found over a large range and in general there is no evidence that they are declining due to trade. The species is of significant economic importance in several range States, particularly India and Pakistan, where large volumes of trade are sourced from plantations. While the species does not meet the Appendix II listing criteria in Annex 2a of *Res. Conf. 9.24 (Rev. CoP17)*, differentiating this species in trade from all other *Dalbergia* species does, at present, remain a major implementation challenge. While methods exist to differentiate *D. sissoo* from other members of the genus in trade, these require expertise and technology not currently widely available globally. The species therefore still meets the criteria in Annex 2bA in that "the specimens of the species in the form in which they are traded resemble specimens of a species included in Appendix II under the provisions of Article II, paragraph 2 (a), or in Appendix I, so that enforcement officers who encounter specimens of CITES-listed species are unlikely to be able to distinguish between them." If the species is not removed from the Appendices, it may be that any impact on the handicraft industry might be mitigated by the proposed change to annotation #15.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Range

Native: Afghanistan, Bangladesh, Bhutan, India, Islamic Republic of Iran, Iraq, Myanmar, Nepal, Pakistan *(although according to Javaid* et al., *(2014) it was introduced to Pakistan in the mid-1800s).*

Introduced: Antigua and Barbuda, Australia, Cameroon, Chad, China, Cyprus, Dominican Republic, Ethiopia, French Polynesia, Ghana, Guinea Bissau, Indonesia, Israel, Kenya, Mauritius, Malaysia, Mozambique, New Caledonia, Niger, Nigeria, Oman, Pakistan, Paraguay, Philippines, Puerto Rico, Senegal, Sierra Leone, South Africa, Sri Lanka, Sudan, Thailand, Togo, Uganda, United Republic of Tanzania, United States of America, Virgin Islands of the USA, Zambia, Zimbabwe.

IUCN Global Category

Not assessed.

Biological and trade criteria for retention in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

Dalbergia sissoo is primarily harvested for its timber, which is used to produce a wide range of products including handicraft items, boats, carts, carriages, gun handles, rail-sleepers, cabinets, furniture, decorative veneer, ornamental turnery, plywood, musical instruments, skis, carvings, tool handles, floorings, etc. Within India, *D. sissoo* is said to be the second most important cultivated timber tree.

The species is native to nine range States and has also been introduced to many others. In some cases, it is considered to be an invasive species (CABI, 2019). While there is a lack of data regarding the status of natural populations (Dhayani, in litt., *2019),* Dalbergia sissoo'*s natural range primarily occurs throughout the sub-Himalayan tract and outer Himalayan valley, ranging from Bangladesh to Afghanistan (Khan, 2000).*

It is also reported to be widespread in plantations within Bangladesh, India, *Nepal and Pakistan (Hossain and Martin, 2012; Javaid* et al., *2014). While their current extent is unclear, in 1979, Pakistan was said to have 100,000 ha of irrigated plantations (National Research Council, 1979).*

Within India the Extent of Occurrence (EOO) is at least 198,974 km^2 considering only the sub-Himalayan tracts from where wild subpopulations of the species are reported. In parts of India, following afforestation programmes, this adaptable species has also become naturalised, further increasing its range.

The density of wild populations in different parts of India is reported to be 8–38 mature individuals per ha, compared with 3–39 per ha for cultivated stocks and up to 1,600 per ha for pure and mono specific plantations. Though disease has caused population declines in some parts of India during the last few decades, *based on a recent non-detriment finding (NDF) study submitted by the Botanical Survey of India, the species is not considered to be under threat (Dhyani* in litt., *2019).* Harvest or trade primarily utilises cultivated trees, *although wild exports have been reported (see below).*

The Supporting Statement reports that between February 2013 and November 2016, a total of 4,739 shipments of *Dalbergia sissoo* were exported from India, worth USD1,079,870, (with an average price per unit of USD4.15 and average value per shipment of USD228), destined for a number of countries around the globe.

According to the CITES Trade Database, the predominant commodities in trade in 2017 were wood products and carvings (see Table 1). Trade data are not yet complete for 2018. There were significant discrepancies reported by exporting countries and importing countries, with importers reporting far more than exporters (see Table 1). India and the main importers (Germany and the USA) have all submitted annual reports for 2017. India has taken a reservation on Dalbergia *spp. (since January 2017) and a Notification (2018/031) states that it has banned the export for commercial purposes of all wild-taken specimens of species in the Appendices apart from certain products of* Dalbergia sissoo *and* D. latifolia.

Table 1. *Global trade in* Dalbergia sissoo *in 2017 reported by importing (Imp.) and exporting (Exp.) countries (according to the CITES Trade Database).*

	kg		Number of items (no unit specified)	
Trade Term	**Imp.**	**Exp.**	**Imp.**	**Exp.**
carvings	5,753,236	-	6,897	-
wood product	79,763	-	735,549	34,324
timber	33,152	-	-	-
derivatives	7,958	-	-	-

Nearly all exports were reported as pre-Convention ("O") (86% carvings reported in kg) with some reports of artificially propagated ("A") (14% of carvings reported in kg) and wild ("W") (66% wood products reported in kg). All trade was for commercial purposes.

Of the products reported as imported, the vast majority were from India, with Pakistan reporting the export of just over 34,000 wood products. The main importers were European countries (predominantly Germany but also others including France and Portugal) and the USA. Most of the trade has been reported as pre-Convention and therefore it is not possible to tell what proportion of trade is wild-sourced or from plantations.

In the years 2017 and 2018, 2,206 import permits for 12,243 t of furniture made of Dalbergia sissoo *have been issued by the German Management Authority (*in litt., *2019). In comparison, only approx. 5.1 t of other small wood products (most of which have been chess boards/men) have been imported with 28 import permits over that period.*

Seizures of Dalbergia sissoo *have been reported by the UK and the Netherlands because they were shipped without CITES permits. The UK seized four shipments from India and three from Pakistan, while the Netherlands seized one shipment from India and one from Suriname (EU-TWIX, 2019). Further* Dalbergia *seizures recorded to genus level may also include* D. sissoo.

Retention in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

In 2017, the entire Dalbergia genus *was listed in CITES Appendix II except for the species listed in Appendix I. It was argued at the time of the proposed listing that some species met the criteria in Annex 2a but that enforcement and customs officers who encountered specimens of* Dalbergia *products would be unlikely to be able to distinguish between the various species of* Dalbergia *reliably so that that the whole genus should be listed. It was also noted that many species of* Dalbergia *have the same wood anatomy, and the process of identification of different species is very difficult, due to the hardness of the wood which hampers the preparation of thin sections for microscopic analysis (McLure* et al., *2015).*
The SS notes that *Dalbergia sissoo* is easy to identify in living condition, unlikely to be confused with other species. *However, live specimens are not the main product in trade.*

"*Topical legumes: Resources for the future*" *published in 1979, states that "Although closely related to the rosewoods Dalbergia sissoo wood is light coloured and lacks the rosewoods' striking grain" (National Research Council, 1979). However, many experts acknowledge that, without the use of technology or high levels of expertise, it is difficult for non-experts readily to identify* Dalbergia sissoo *once made into finished products (Hartvig* et al., *2015; Dhyani,* in litt., *2019; Koch,* in litt,, *2019; Sivadas,* in litt., *2019; Vlam and Zuidema,* in litt., *2019).*

*Koch (*in litt., *2019), also notes that, in particular,* Dalbergia oliveri *(range: Myanmar, Thailand, Lao People's Democratic Republic (PDR), Viet Nam and Cambodia) bears a similar colour and texture to* D. sissoo *and requires expertise for a differentiation, particularly if the origin of the wood is unknown.*

A range of scientific techniques to enable the identification of Dalbergia sissoo *in trade are available (Hartvig* et al., *2015; Espinoza* in litt., *2019; Koch* in litt., *2019; Vlam and Zuidema,* in litt., *2019). However, the application of these is currently severely restricted, due to the financial resources and expertise required to implement them.*

For example: Macroscopic visual identification methods using identification guides can be utilised by non-experts to identify Dalbergia sissoo *to genus level (Koch,* in litt., *2019). However, to identify* D. sissoo *to species level, microscopic inspection of a range of additional structural features is required, which demands a high level of expertise and laboratory equipment to perform (Koch* et al., *2011; Koch* in litt., *2019). The only exception is for distinguishing between* D. sissoo *and* D. latifolia*, when macroscopic inspection would suffice (Koch* in litt., *2019). The level of expertise and experience required to perform microscopic inspection (Dormontt* et al., *2015; Koch,* in litt.*, 2019), means that, at present, this technique is not widely available to global enforcement efforts. In addition, others consider that visual techniques cannot always be used to identify wood within composite materials, or that have been stained/dyed a different colour.*

Technological methods include DART TOFMS (Direct Analysis in Real Time, Time of Flight Mass Spectrometry), which has proven ability to identify Dalbergia sissoo *in trade (Espinoza,* pers. comm.*, 2019). This system works by combusting a small sample of wood, which enables its chemical profile to be analysed. It is capable of identifying samples to species level, with 2,000 species (including 90% of those listed within CITES) catalogued within its database, including* D. sissoo*. The system is also accurate regardless of the age or part of the tree that is tested, and with the exception of very thin plywood (which is contaminated with glue), it is capable of identifying all forms of wooden products in trade. The cost of this system (USD 250,000 to install and, in the US, USD 250 to process each sample) may currently be a barrier to its implementation, and to date, uptake of the system by global enforcement agencies has been low (Espinoza,* pers. comm., *2019).*
DNA barcoding has also been demonstrated as capable of identifying Dalbergia sissoo *to species level (Hartvig* et al., *2015). To be used practically however, this technique first requires the creation of reference data sets, or species-specific assays. DNA extracted from timber may also be of poor quality, which can hamper the process (Hartvig* et al., *2015).*

At the present time, therefore, a considerable gap remains between the potential and realised application of such methodologies (Dormontt et al., *2015).*

While prior knowledge of the wood's origin is likely to help in the identification of Dalbergia sissoo*, (Koch* in litt., *2019), as it is not endemic to any one country, this may be of little assistance to global enforcement efforts.*

Additional Information

Threats

The main threats to both wild and cultivated populations of *Dalbergia sissoo* are fungal and bacterial diseases (wild and dieback being the diseases that have the largest impact) and insect infestations. Wilt disease has been reported from some plantations within India, where *D. sissoo* has been raised in unsuitable conditions. *Plantations of D. sissoo have suffered from significant dieback in Bangladesh, where mortalities in excess of >50% have been reported (Winfield* et al., *2016).* The species' high regeneration and growth rate, however, reduces their impact upon the species as a whole. The frequency of mortality due to diseases is also lower in wild subpopulations than in cultivated plantations.

Conservation, management and legislation

The Government of India has banned the export for commercial purposes of all wild-taken specimens of species included in Appendices I, II and III. It has, however, taken a general reservation to Dalbergia *spp. (except species in Appendix I) and permits the export of cultivated varieties of plant species included in Appendices I and II and of products from wild sourced* D. sissoo *and* D. latifolia *that are authorised for export by a CITES Comparable Certificate,* ***except*** *logs, timber, stumps, roots, bark, chips, powder, flakes, dust and charcoal. CITES Comparable Certificates will be issued with a footnote, stating that the wild (W) source specimens are covered under Legal Procurement Certificate as per regional and national laws in India (Notification No. 2018/031).*

The SS also notes that within India, wild populations of *Dalbergia sissoo* are found within several protected areas. Harvest outside of protected areas is regulated but this varies geographically. Its fast growth rate and use within a number of industries have made *D. sissoo* a preferred choice for forest departments and other agencies undertaking afforestation programmes, and also for farmers who grow this species for commercial use.

Artificial propagation/captive breeding

The species can be found in plantations and/or agroforestry systems in almost every part of India. It can be found growing under controlled conditions within farms, gardens and plantations. Artificial propagation is possible from almost all common practices such as: sowing seeds; planting stumps, root sections and stem cuttings; cloning cuttings; and entire transplanting.
Commercial plantations exist in both the area of natural distribution (Indian subcontinent), as well as in China and some African countries (Koch*, in litt.,* 2019).

Potential risk(s) of a deletion
A lack of enforcement capability would leave open the possibility of other rosewood species being mis-declared as Dalbergia sissoo*, with the detection and prosecution of these crimes hampered by the practical difficulties outlined above. As the range of* D. sissoo *overlaps with that of other* Dalbergia *species, (Winfield* et al., *2016), it is conceivable that such opportunities may arise. Prior to the* Dalbergia *genus listing, traffickers were said to have taken advantage of gaps in the CITES listings for rosewood, for example, by mis-declaring* D. retusa*, as the then unlisted and similar-looking,* D. bariensis *(EIA, 2016).*

Potential benefit(s) of deletion for trade regulation
It is likely that deleting *Dalbergia sissoo* from Appendix II would mitigate the negative impacts that this listing has reportedly had had on some areas of international trade, particularly the negative impacts on the trade in wooden handicrafts from India.

Other comments
Many species of Dalbergia *are under a range of threats, including deforestation, forest conversion for agriculture/human development, and legal and illegal logging to supply domestic and international markets (Winfield* et al., *2016). Trade in some species of* Dalbergia *considered to be "precious woods" with high market values, has resulted in their over-exploitation (Jenkins* et al., *2012). The IUCN currently lists 57 species within the Vulnerable (26 species), Endangered (29 species) and Critically Endangered (2 species) categories.*

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Amendment of Annotation #15

Proponents: Canada and the European Union

Summary: The scope of the listing for species of *Dalbergia* included in Appendix II as well as *Guibourtia demeusei, G. pellegriniana* and *G. tessmannii* is defined by annotation #15 to the listing, which currently reads:

All parts and derivatives are included, except:
a) Leaves, flowers, pollen, fruits, and seeds;
b) Non-commercial exports of a maximum total weight of 10 kg per shipment;
c) Parts and derivatives of *Dalbergia cochinchinensis,* which are covered by Annotation # 4;
d) Parts and derivatives of *Dalbergia* spp. originating and exported from Mexico, which are covered by Annotation # 6.

The changes proposed to this annotation are:
- to remove the current part b) and add a new b) "Finished products to a maximum weight of wood of the listed species of 500g per item"
- add a new c) "Finished musical instruments, finished musical instrument parts and finished musical instrument accessories"
- relabel the current c) and d) as d) and e) respectively.

There have been challenges in interpretation and implementation of this annotation. These include concerns that some of the commodities currently covered by the listing (including finished products such as musical instruments and furniture) are not those that first appear in international trade as exports from range States and therefore their inclusion under the annotation was inconsistent with the guidance on annotations provided in *Res. Conf. 11.21 (Rev. CoP17)*. These issues have led to considerations on this matter by the Standing Committee and its Working Group on Annotations.

Analysis: The proposed amendment to annotation #15 is the result of the extensive discussions and consensus reached by the Standing Committee Working Group on Annotations (see SC70 Com.17). The Standing Committee has supported the proposed amendment, which is intended to reduce the challenges with interpretation and implementation of the current annotation #15 experienced by Parties and ensure the annotation is in line with guidance on use of annotations in *Res. Conf. 11.21 (Rev. CoP17)*. Given the extensive debate on these changes and the consensus reached by the Standing Committee, the proposed changes should address the issues raised by (the majority of) stakeholders. Finished pieces of furniture made from the species to which the annotation applies are unlikely to contain wood of those species weighing less than 500g, so if the proposal is accepted these would continue to be covered by the listing, regardless of whether they were being exported by a range State or a processing country.

Summary of information and discussion

Before CoP17 the following taxa in the *Dalbergia* genus were already included in Appendix I and II.

Appendix I
Dalbergia nigra
Appendix II
Dalbergia granadillo (Logs, sawn wood, veneer sheets and plywood)
Dalbergia retusa (Logs, sawn wood, veneer sheets and plywood)
Dalbergia stevensonii (Logs, sawn wood, veneer sheets and plywood)
Dalbergia spp. (populations of Madagascar) (Logs, sawn wood and veneer sheets)

At CoP17 (2016) a proposal to list all *Dalbergia* species (except for species already listed in Appendix I, namely *D. nigra*) with no annotation was submitted (CoP17 Prop. 55). A separate proposal (CoP17 Prop. 54) was submitted to list 13 species of the genus *Dalbergia* native to Mexico and Central America in Appendix II with annotation #6 (Logs, sawn wood, veneer sheets and plywood) (*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).

A third proposal (CoP17 Prop. 56) was submitted to include three species of *Guibourtia (Guibourtia demeusei, G. pellegriniana* and *G. tessmannii*) in Appendix II with annotation #4:
All parts and derivatives, except:

a) seeds (including seedpods of Orchidaceae), spores and pollen (including pollinia). The exemption does not apply to seeds from Cactaceae spp. exported from Mexico, and to seeds from *Beccariophoenix madagascariensis* and *Dypsis decaryi* exported from Madagascar;
b) seedling or tissue cultures obtained in vitro, in solid or liquid media, transported in sterile containers;
c) cut flowers of artificially propagated plants;
d) fruits, and parts and derivatives thereof, of naturalized or artificially propagated plants of the genus *Vanilla* (Orchidaceae) and of the family Cactaceae;
e) stems, flowers, and parts and derivatives thereof, of naturalized or artificially propagated plants of the genera *Opuntia* subgenus *Opuntia* and *Selenicereus* (Cactaceae); and
f) finished products of *Euphorbia antisyphilitica* packaged and ready for retail trade).

The final decisions at CoP17 resulted in inclusion of all species of *Dalbergia* (except for those in Appendix I) and the three species of *Guibourtia* in Appendix II with annotation #15, which combined different annotations that had been agreed for *D. cochinchinensis* (CoP17 Prop. 53) and the 13 species from Mexico and Central America.

The proponents of the current proposal (CoP18 Prop. 52) wish to amend annotation #15, which currently reads:

"All parts and derivatives are included, except:
a) Leaves, flowers, pollen, fruits, and seeds;
b) Non-commercial exports of a maximum total weight of 10 kg per shipment;
c) Parts and derivatives of *Dalbergia cochinchinensis*, which are covered by Annotation #4;
d) Parts and derivatives of *Dalbergia* spp. originating and exported from Mexico, which are covered by Annotation # 6."

To
"All parts and derivatives, except:

a) leaves, flowers, pollen, fruits, and seeds;
b) finished products to a maximum weight of wood of the listed species of 500g per item;
c) finished musical instruments, finished musical instrument parts and finished musical instrument accessories;
d) parts and derivatives of *Dalbergia cochinchinensis*, which are covered by annotation #4;
e) parts and derivatives of *Dalbergia* spp. originating and exported from Mexico, which are covered by annotation #6.

The substantive changes being the removal from the annotation of "non-commercial exports with a maximum weight of 10 kg per shipment", instead specifying that finished products with a maximum weight of wood of the listed species of 500g per item, whether commercial or non-commercial, and finished musical instruments, finished musical instrument parts and finished musical instrument accessories, are all excluded from the controls of the Convention.

There have been challenges in interpretation and implementation of this annotation. For example, there are concerns that some of the commodities currently covered by the annotation (including finished products such as musical instruments and furniture) are not those that first appear in international trade as exports from range States, and therefore that their inclusion under the annotation was inconsistent with the guidance on annotations provided in *Res. Conf. 11.21 (Rev. CoP17) Use of annotations in Appendices I and II.* These issues have led to considerations by the Standing Committee and its Working Group on Annotation.

The Standing Committee considered various options for amending the annotation presented by the Working Group and the proposed revision is a result of the consensus reached (see SC70 Com.17). No definition for "musical instruments" has been put forward for inclusion in *Res. Conf. 10.13 (Rev. CoP15) Implementation of the Convention for timber species*, but the Standing Committee has recommended the re-establishment of an annotation working group after CoP18, which would consider this, and other terms used in definitions.
If the proposed amendments to #15 are accepted, this would result in finished pieces of furniture comprising the species subject to the annotation remaining covered by the listing unless they contain wood of those species weighing less than 500 g, which may be unlikely, whether being exported by a range State or a processing country.

The Standing Committee Working Group on Annotations did note that “the inclusions in annotations of external considerations such as the status in trade of a specimen or shipment, or of thresholds defined by weight or volume that affect the application of an annotation to a part or derivative, require subjective evaluation and are likely to create uncertainty with regard to their interpretation and application” (See SC70 Doc 67.1 for further information).

Amendment of the annotation to the listing of *Pericopsis elata* in Appendix II: expand the scope of the annotation (currently #5) to include plywood and transformed wood

Proponents: Côte d'Ivoire and the European Union

Summary: *Pericopsis elata*, commonly known as Afrormosia or African Teak, is a highly valued tropical timber native to Central and West Africa. *Pericopsis elata* was listed in Appendix II in 1992 with annotation #5 (amended in 2007), which restricts the listing to "logs, sawn wood and veneer sheets". At the time the annotation was intended to cover the major products in trade.

The European Union (EU), one of the main importers of timber of this species, has observed instances where traders from range States have been exporting sawn wood with minor, superficial transformation in order to circumvent CITES controls. The Standing Committee Annotations Working Group considered that the extent and scale of cases where the listing was being circumvented warranted a change to the annotation to ensure that CITES controls cover those commodities that dominate the trade, and supports the amendment proposed by Côte d'Ivoire and the EU. Although the full extent of trade in this transformed wood is unknown, it is likely to be only superficially different to sawn wood, which currently dominates the reported international trade.

The proposed amendment would expand the current annotation for *P. elata* to include plywood and transformed wood to read:

"Logs, sawn wood, veneer sheets, plywood, and transformed wood[1]."

In addition, a footnote is included to "transformed wood" that would read:

"[1] Whereby transformed wood is defined by HS code 44.09: Wood (including strips, friezes for parquet flooring, not assembled), continuously shaped (tongued, grooved, v-jointed, beaded or the like) along any edges, ends or faces, whether or not planed, sanded or end-jointed."

The proposed amendment is intended to expand the scope of the listing of *P. elata* to close the observed loophole and include commodities that first appear in international trade as exports from range States, and commodities that dominate the trade and demand for the wild resource, as advised in *Res. Conf. 11.21* (*Rev. CoP17*) *Use of Annotations in Appendices I and II*.

A similar proposal was submitted for *Dalbergia cochinchinensis* at CoP17 (2016) where the same loophole under annotation #5 was identified as being exploited. That proposal was accepted and the species is now listed with annotation #4.

Other species are currently listed in Appendix II and III with annotation #5, including some species of *Cedrela*. A separate proposal has been submitted to list the *Cedrela* genus in Appendix II with no annotation (CoP18 Prop. 57). It does not appear that the present proposal (CoP18 Prop. 53) is intended to apply to all taxa listed with annotation #5. Therefore, amending the annotation for *P. elata* only would require a new annotation solely for *P. elata*.

Analysis: International trade in *Pericopsis elata* appears to involve products not included in the current listing under annotation #5, based on the observation of shipments into the EU of superficially transformed sawn wood. The intention to include transformed wood (and plywood) to close the observed loophole seems an appropriate amendment and has been supported by the Standing Committee Working Group on Annotations.

As other species are also listed with annotation #5, if the proposed amendment is accepted a new annotation would be required specifically to cover *P. elata*.

The proposed amended annotation includes the term plywood, which is already defined in *Res. Conf. 10.13 (Rev. CoP15) Implementation of the Convention for Timber Species*. No other existing annotations include the term "transformed wood". The proposed footnote to the annotation provides a definition of "transformed wood" (as HS44.09) in line with guidance in *Res. Conf. 10.13 (Rev. CoP15*). However, it may be more appropriate to include the proposed definition in *Res. Conf. 10.13 (Rev. CoP15)* rather than to have this as a footnote to the annotation. Thus, any changes to the definition could be amended through an amendment to the Resolution rather than through another proposal to amend the Appendices.

Other Considerations: There is a proposal (CoP18 Prop. 49) to include the genera *Handroanthus, Tabebuia* and *Roseodendron* in Appendix II with annotation #6 to cover "logs, sawn wood, veneer sheets and plywood", however transformed wood also appears to be one of products in trade and if the amended annotation for *Pericopsis elata* is accepted then the same annotation may also be appropriate for those genera.

The trade term "transformed wood" is not included in the Annex to the *Guidelines for the preparation and submission of CITES annual reports* (Notification No. 2017/006), which may need addressing.

Summary of additional information

Range
Cameroon, Central African Republic, Côte d'Ivoire, Democratic Republic of the Congo, Ghana, Nigeria and Republic of Congo.

IUCN Global Category
Endangered A1cd ver. 2.3 (1998) - needs updating.

Discussion
Pericopsis elata, commonly known as Afromosia or African Rosewood, is a highly valued tropical timber species native to Central and West Africa. *Pericopsis elata* was included in Appendix II in 1992. It is currently listed with annotation #5 to restrict the listing to "logs, sawn wood and veneer sheets".

The EU, one of the main importers of timber of this species, has observed instances where traders from range States have been exporting sawn wood with minor, superficial transformation in order to circumvent CITES controls. The Standing Committee Annotations Working Group considered that the extent and scale of cases where the listing was circumvented warranted a change to the annotation to ensure that CITES controls cover those commodities that dominate the trade, and supports the amendment proposed by Côte d'Ivoire and the EU.

Sawn wood currently dominates the reported trade in *P. elata* (see Table 1). It is unclear whether superficially transformed wood is being reported (if the assumption is that it is not covered by the annotation), and therefore it is not possible to determine the full extent of the trade. However it is likely to be only superficially different to sawn wood, which currently dominates the reported international trade.

The proposed amendment would expand the current annotation for *P. elata* to include plywood and transformed wood to read:

"Logs, sawn wood, veneer sheets, plywood, and transformed wood[1]."

With a footnote to "transformed wood" that would read:

"[1] Whereby transformed wood is defined by HS code 44.09: Wood (including strips, friezes for parquet flooring, not assembled), continuously shaped (tongued, grooved, v-jointed, beaded or the like) along any edges, ends or faces, whether or not planed, sanded or end-jointed."

Res. Conf. 10.13 (Rev CoP15) Implementation of the Convention for timber species recommends that "for the purpose of annotations in the Appendices for parts and derivatives of species traded as timber, definitions to be used should, to the extent possible, be based on the tariff classifications of the Harmonized System of the World Customs Organization"; the following terms used in timber-related annotations are already defined in *Res. Conf. 10.13 (Rev CoP15*):

i) Logs

All wood in the rough, whether or not stripped of bark or sapwood, or roughly squared, for processing, notably into sawn wood, pulpwood or veneer sheets (HS code 44.03);

ii) Sawn wood

Wood simply sawn lengthwise or produced by a profile-chipping process. Sawn wood normally exceeds 6 mm in thickness (HS code 44.06, HS code 44.07);

iii) Veneer sheets

Thin layers or sheets of wood of uniform thickness, usually 6 mm or less, usually peeled or sliced, for use in making plywood, for veneering furniture, veneer containers, etc. (HS code 44.08); and

iv) Plywood

Consisting of three or more sheets of wood glued and pressed one on the other and generally disposed so that the grains of successive layers are at an angle (HS code 44.12.13, HS code 44.12.14, and HS code 44.12.22); and

The definitions also include footnotes for the different HS codes used as follows;
44.03: Wood in the rough, whether or not stripped of bark or sapwood, or roughly squared
44.06: Railway or tramway sleepers of wood
44.07: Wood sawn or chipped lengthwise, sliced or peeled, whether or not planed, sanded or finger-jointed, of a thickness exceeding 6 mm
44.08: Veneer sheets and sheets for plywood (whether or not spliced) and other wood sawn lengthwise, sliced or peeled, whether or not planed, sanded or finger-jointed, of a thickness not exceeding 6 mm

The amended annotation includes plywood, which is already defined in *Res. Conf. 10.13 (Rev. CoP15)*. Although proposed annotation includes plywood it is not clear whether this is a significant product in trade. No other existing annotations include the term “transformed wood”. The proposed footnote provides a definition of “transformed wood” in line with guidance in *Res. Conf. 10.13 (Rev. CoP15*). Currently there is no trade term for “transformed wood” included in the *Guidelines for the preparation and submission of CITES annual reports* (Notification No. 2017/006).

The proposed amendment is intended to expand the listing of *P. elata* to close the observed loophole and include commodities that first appear in international trade as exports from range States, and commodities that dominate the trade and demand for the wild resource, as advised in *Res. Conf. 11.21 (Rev. CoP17) Use of Annotations in Appendices I and II*.

A similar proposal was submitted for *Dalbergia cochinchinensis* at CoP17 where CITES controls under annotation #5 were being circumvented through superficial transformation. The proposal by Thailand (CoP17 Prop. 53) was to change the annotation to #4 thus including all parts except,

a) seeds, spores and pollen.
b) seedling or tissue cultures obtained in vitro,
c) cut flowers of artificially propagated plants;
d) fruits, and parts and derivatives thereof,
e) stems, flowers, and parts and derivatives thereof.

Thus, any timber-related, semi-processed or finished product for *Dalbergia cochinchinensis* is now included in the listing.

Other species in the Appendices are currently listed with annotation #5 including:

- Appendix II: *Diospyros* spp. (Populations of Madagascar) and *Swietenia mahagoni*
- Appendix III: *Quercus mongolica* (Russian Federation), *Cedrela fissilis* (Plurinational State of Bolivia, Brazil), *Cedrela lilloi* (Plurinational State of Bolivia, Brazil), *Cedrela odorata* (Plurinational State of Bolivia, Brazil, Colombia, Guatemala and Peru*), Fraxinus mandshurica* (Russian Federation), *Pinus koraiensis* (Russian Federation).

Therefore, amending the current annotation #5 for *P. elata* only would require a new annotation solely for *P. elata* as it does not appear from the proposal that the proposed amendment is intended to apply to all taxa currently listed with #5. There is a proposal (CoP18 Prop. 49) to include the genera *Handroanthus, Tabebuia* and *Roseodendron* in Appendix II with annotation #6 to cover “logs, sawn wood, veneer sheets and plywood”, however transformed wood also appears to be one of products in trade and if the amended annotation for *P. elata* is accepted then the same annotation may also be appropriate for those genera. There is also a proposal to list the *Cedrela* genus in Appendix II at CoP18 with no annotation (CoP18 Prop. 57).

Table 1: Summary of trade reported by importers in the CITES Trade Database for *Pericopsis elata* 2008-2017

Term/ unit	Quantity reported by importers
chips	
# specimens	41
derivatives	
m^3	54
live	
# specimens	28
logs	
m^3	66,345
# specimens	140
sawn wood	
kg	
m^3	160,783
# specimens	158
timber	
kg	383
m^3	1,129
veneer	
m^2	6
m^3	51
# specimens	24
wood products	
m^3	64

References

African Regional Workshop (Conservation & Sustainable Management of Trees, Zimbabwe, July 1996) (1998). *Pericopsis elata.* The IUCN Red List of Threatened Species 1998.

Inclusion of African Padauk *Pterocarpus tinctorius* in Appendix II

Proponent: Malawi

Summary: *Pterocarpus tinctorius* is a tree species native to nine countries across Africa's belt of miombo woodland vegetation. It is a slow-growing tree, estimated to take up to 90 years to reach maturity. The species is in international trade, mainly for its timber which is used for furniture and flooring. It is commonly traded under the general name "mukula" or sometimes "African Padauk", names that are also applied to similar species, such as *P. angolensis, P. soyauxii* and *P. castelsii*. Domestic demand is also said to be high for timber, firewood and a variety of other uses.

There is very little information about population size, structure and rates of decline for *P. tinctorius*, although it is thought to be locally common, but declining across its range, and some national populations are known to be decreasing (e.g. Zambia). Taking into account the risk of over-harvesting, *P. tinctorius* was assessed as Least Concern in 2017. The assessment recommended the species' harvest and trade be monitored to identify any major increase in its use, particularly as other *Pterocarpus* species in trade become rare or protected.

The main international market is considered to be China, and to a lesser extent Viet Nam. Although *Pterocarpus tinctorius* is not officially recognised as a "hongmu" species (other species of *Pterocarpus* are) or included on China's list of precious furniture woods, reports suggest that the species has seen an increase in exploitation due to a growth in consumption of "hongmu" and other "rosewoods" in China since 2010.

As multiple species are commonly traded under the same names, it is difficult to determine specific trade levels of *P. tinctorius.* There is some confusion surrounding legislation in certain range States, so it is not clear how much of the trade is illegal, although a number of seizures have taken place. Examples of trade volumes include from the Democratic Republic of the Congo (DRC) where it was estimated in 2015 that almost 45,000 m^3 of "mukula" was transported across the border annually to Zambia, and onwards to China. Trade data from Tanzania show that exports of *P. tinctorius* increased seven-fold between 2012 and 2014 from around 800 m^3 to 5,600 m^3.

Pterocarpus erinaceus, listed in Appendix II in 2017, is a "hongmu" species native to west and central Africa, including in countries that border range States of *P. tinctorius*. There is conflicting information regarding the ease of identification of this species and others in the genus. Some consider *P. erinaceus* wood to be distinguishable from other *Pterocarpus* species due to the light base colour of the heartwood, although others say that identification is only reliable at the genus level.

The proposal is to list *P. tinctorius* in Appendix II without annotation, in order to include all readily recognisable parts and derivatives.

Analysis: *Pterocarpus tinctorius* is harvested for timber and has a number of other local uses. There is evidence of a recent increase in export of timber from some range States, largely to meet demand in China for furniture-making. A proportion of this export appears to be unauthorised or illegal. The species is widespread and locally common, and although it is thought to be declining it was assessed by IUCN as Least Concern in 2017. It is a slow-growing, late-maturing species. 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. However, very little species-specific trade data are available, and it is unknown how much harvest is for domestic versus international markets. While there is insufficient evidence to determine clearly whether the species meets the criteria in Annex 2a of *Res. Conf. 9.24 (Rev. CoP17),* given the uncertainty and the apparent serial exploitation of similar precious wood-producing trees, it may be precautionary to list the species in Appendix II.

There seem to be some difficulties in distinguishing between *P. tinctorius and P. erinaceus* (already listed in Appendix II), and therefore it seems likely that *P. tinctorius* meets the look-alike criteria for listing in Appendix II provided in Annex 2b of *Res. Conf. 9.24 (Rev. CoP17)*.

The proposal without annotation is intended to avoid the potential for regulations to be circumvented as has been seen with other rosewood listings and seems a sensible approach.

Other Considerations: Some trade is likely to be illegal as certain range States have export bans in place. Any additional benefits of an Appendix II listing are not clear unless enforcement efforts are increased. If this proposal is accepted, those range States with export bans could request that the CITES Secretariat posts zero quotas on the CITES website if they wished to reflect national legislation.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS

Taxonomy

Synonyms: Pterocarpus chrysothrix, Pterocarpus megalocarpus, Pterocarpus stolzii:

There is ongoing debate regarding the taxonomy of Pterocarpus tinctorius*, and the wider genus. Of the two species in the genus that are currently listed in the Appendices, only one has a CITES Standard Reference—Pterocarpus erinaceus (Mabberley, 1997), although other taxonomic references are provided* (UNEP-WCMC, 2019)*. CoP18 Doc. 99 notes that the Plants Committee at its 24th meeting recommended that Mabberley (1997) be deleted from the list of Standard References, and thereafter that relevant decisions will be made on a case by case basis. For* P. tinctorius *it is unclear which Standard Reference will be used.*

In a wider taxonomic debate, the circumscription of Pterocarpus tinctorius *varies considerably. In Flora Zambesiaca upholds* P. chrysothrix, P. megalocarpus *and* P. stolzii *as separate species, whereas the African Plants Database maintains them as synonyms of* P. tinctorius *(Conservatoire et Jardin botaniques de la Ville de Genève and South African National Biodiversity Institute, 2019; Timberlake, 2012). Furthermore, a recent publication on the trees and shrubs of Mozambique considers* P. chrysothrix *and* P. megalocarpus *as separate species to* P. tinctorius *(Burrows* et al., *2019).*

For the purpose of the analyses provided below, P. chrysothrix *and* P. stolzii *have been considered as synonyms of* P. tinctorius *as per the IUCN Red List assessment (Barstow, 2018).*

Range

Angola, Burundi, Democratic Republic of Congo, Malawi, Mozambique, *and Rwanda* (Winfield, Scott, & Grayson, 2016)*, Republic of Congo (Barstow, 2018),* Tanzania and Zambia,

IUCN Global Category

Least Concern (Assessed 2017, criteria version 3.1)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

A) Trade regulation needed to prevent future inclusion in Appendix I

Habitat trends: *Pterocarpus tinctorius* is found across Africa's broad belt of miombo woodland, a 2.7 million km^2 area of tropical seasonal forest and dry forests. Forest disturbance and loss are occurring throughout the miombo woodlands and associated ecosystems that form *P. tinctorius*'s range. Agriculture and fuel wood collection play a role in the degradation of existing forest areas. While a number of protected areas do exist across the range, these are not always well-protected in practice and can be subject to encroachment and illegal logging.

Population size: The total population of *Pterocarpus tinctorius* is not known, nor are quantitative data available on the total area of relevant habitat or average density of stems per hectare. The SS notes that the species was apparently locally common in areas including east and south Tanzania and north Malawi, at least before the rosewood demand increase; it is now in decline throughout Zambia and likely other countries as well.

A recent study in Zambia estimated a stems per hectare figure of 0.274 for Pterocarpus tinctorius *(Ng'andwe et al., 2017), using this figure, it would require a forested area of less than 200 km^2 for the population to be greater*

than 5,000 individuals. The study did not report if saplings were included in the final analysis, or if the densities only referred to mature trees.

In Tanzania, a survey of the Katavi-Rukwa ecosystem classified Pterocarpus tinctorius *as uncommon (Banda* et *al., 2008), whilst at one study site in Ugalla area of Tanzania,* P. tinctorius *was found to make up 9.3% of the forest (Ogawa* et al., *2007).*

Population structure: Very little information is available. However, given that the largest specimens are disproportionately targeted for timber production, it can be expected that the recent increase in illegal and unsustainable harvesting will be leading to a skewing of the population structure towards immature specimens. In Zambia, the minimum cutting diameter of *Pterocarpus tinctorius* was reduced to 30 cm in 2015 in response to commercial pressures. Rosewoods as a group exhibit poor recruitment, even in protected areas where large numbers of mature trees exist. While some local loggers in Zambia claimed to observe large numbers of seedlings in areas where they worked, this may mean little in terms of survivorship. Other *Pterocarpus* species in this region demonstrate low regeneration patterns.

Where species specific information is lacking, comparisons are made with a similar species within the genus, Pterocarpus angolensis*, although the IUCN Red List assessment for* P. tinctorius *does state that it is a slow-growing tree, estimated to take up to 90 years to reach maturity (Barstow, 2018). Whereas other sources for* P. angolensis *state that maturity is reached at around 20 years (Winfield* et al., *2016)*

Data on a similar species Pterocarpus angolensis *suggests that this species is slow-growing and it takes a minimum of 85 years and up to 100 years to reach a harvestable size* (Therrell, Stahle, Mukelabai, & Shugart, 2007)*.*

*Recent research on a similar species (*Pterocarpus angolensis*) in the Miombo woodland in Zambia, shows that selective logging can hinder regeneration.* Pterocarpus *spp. are light-demanding and therefore activities such as slash and burn or felling for charcoal production can promote faster regrowth as more light penetrates through the canopy, whereas in areas of selective logging trees exhibited static stem diameters (Syampungani & Geldenhuys, 2015).*

Population trends: At a genus level, 90% of the *Pterocarpus* and *Dalbergia* populations, where studies exist, show declining or unstable population dynamics. According to the 2018 IUCN Red List assessment, the *P. tinctorius* population "is considered to be in decline as a result of the harvesting of the species for its timber…currently in high demand in local markets and it is predicted that in the future its international demand could increase as other *Pterocarpus* timber species become rare or protected.". A study of the mukula value chain in Zambia found that 84% of community cutters had entered the business since 2012. 68% of these cutters observed depletion of population stocks in the field and thought that they would not be able to continue harvesting this tree at the same rate in five years' time, 95% of respondents in the same study agreed with this assessment and anticipated the species "going extinct".

In Kiwengoma Forest, Tanzania, there was a high proportion of high value timber species (notably Pterocarpus tinctorius*) in 1991, but in 2005 logging had reduced the number of high value and large timber trees (Ahrends* et al., *2010).*

B) Regulation of trade required to ensure that harvest from the wild is not reducing population to level where survival might be threatened by continued harvest or other influences

While *Pterocarpus tinctorius* has non-timber uses, its current unsustainable extraction in various range States is thought to be almost entirely linked to international trade.

It has been reported that the main two threats to Pterocarpus tinctorius *are selective logging for domestic use and harvest for export (Winfield* et al., *2016).*

Prices of Pterocarpus tinctorius *have increased from USD 2.40/plank (equivalent to USD 43.60/m^3) in 1990, to USD 4.00/plank (equivalent to USD 72.00/m^3) in 2000 (Winfield* et al., *2016).*

National utilisation

Pterocarpus tinctorius has various uses within range States. As well as timber use, these include, use of sap for fabric dyes and body colouring; medicinal treatments for respiratory congestion, anaemia, diarrhoea, snakebites, stomach aches, and eye pain; to prevent miscarriage, and to prevent wound infection; firewood, charcoal and honey production.

Legal international trade
It should be noted that much of the trade in this species is reported under the general trade name "mukula", that includes other species within the genus such as Pterocarpus angolensis *(Cerutti* et al., *2018).*

It is difficult to separate out legal and illegal trade from the available data, given the patchy data, irregular enforcement and lack of clarity around national regulations in some countries. Official Chinese data show rapidly increasing imports of rosewood species from African nations—up 700% since 2010. While *Pterocarpus tinctorius* is not on the official hongmu list *or the list of precious furniture woods*, it has achieved market demand due to its look-alike characteristics. Chinese buyers in Zambia reported to CIFOR interviewers that an early increase in *Pterocarpus tinctorius* (beginning in 2010) was actually due to its being used as a false rosewood: shipments were sent through intermediary traders and nations to Viet Nam and the Philippines, where it was mixed with *P. santalinus* (red sandalwood) and sold onto the Chinese furniture market.

It is not possible to analyse specific-species data, but general trends are showing an increasing proportion of China's rosewood imports are coming from Africa and that Asian rosewood imports are decreasing in quantity and log diameter reflecting the decline in Asian rosewood species due to decades of over-harvesting (Treanor, 2015).

Although species-specific trade data are difficult to compile for timber, imports of Pterocarpus *spp. into Viet Nam between 2013 and 2016 have been assessed and show* P. erinaceus *dominate imports of* Pterocarpus *spp. into Viet Nam. Imports of* P. tinctorius *logs are almost negligible in comparison and examinations of re-exports show that no* P. tinctorius *sawn wood was re-exported by Viet Nam to China between 2013 and 2015 (Winfield* et al., *2016).*

However, over time the species has become recognised in its own right and direct shipments to China are more common. Chinese customs data show log imports from Zambia going from 35,000 m^3 in 2015, to 65,000 m^3 in just the first half of 2017; CIFOR research indicates that "the vast majority" of these logs are *Pterocarpus tinctorius (SS attributes this to FAOSTAT and Chinese Customs and Cerutti 2017, Fig. 14 and 15). However, the increase in shipments to China referred to are not guaranteed to be* P. tinctorius*, the Zambian and Chinese customs data presented by Cerutti* et al., (2018) *is for sawn wood and logs, which are not reported to species level using data from FAOSTAT. It is inferred that the majority of these shipments consist of "mukula", but this is based on stakeholder opinions. This data are then extrapolated to suggest an annual estimation of production of mukula of 110,000 m^3, which is the equivalent of between 450,000 and 800,000 logs.*

In 2017 the going price for mukula in Zhang Jiagang was between RMB17,000 and 22,000 per tonne (USD 2,500 and USD 3,200). Greenpeace estimates that as much as 15,000 t of mukula are sold each month from just the four biggest mukula markets. In Tanzania, one of the few countries where species-specific official data are available, export permits for *Pterocarpus tinctorius* increased almost seven times between 2012 and 2014 alone (830 to 5,600 m^3), according to Tanzania Forest Service data. *To clarify, it is not the number of permits that increased seven-fold between 2012 and 2014 in Tanzania, it was the volume of* P. tinctorius *that increased from 831 m^3 to 5,578 m^3 (Lukumbuzya & Sianga, 2017).*

Range State governments have struggled to improve governance over this resource. For example, Zambia has imposed and lifted moratoriums on harvest and/or export of mukula three times since 2014, and an export ban on all logs is currently in place, although the Minister and Director of Forests may still issue export permits for timber if "deemed necessary in the interest of the Republic". Malawi banned export of all roundwood in 2008 but has faced many legal battles to enforce the ban for *Pterocarpus tinctorius* which is apparently transported to China originating from neighbouring countries.

Parts and derivatives in trade
The products in international trade are primarily round and rough squared logs (HS code 44.03) and rough sawn timber (HS code 44.07). The majority of the trade is destined for China, although Viet Nam also imports significant volumes. In importing markets, the main usage is for decorative furniture consumed in China. There is no species-specific information available on re-exports of furniture or secondary processed products from China.

Illegal trade
Illegal mukula export trade is fundamentally a regional issue, in part because the most significant extraction appears to be occurring in the landlocked forests of south-eastern Congo (Katanga Plateau), Zambia and north-east Angola. Trucking routes are documented towards ports on both the Atlantic (Angola, Namibia, even South Africa) and Pacific (Tanzania, Mozambique), though the key routes change as range States have attempted to control trade in this and other precious wood species through log export bans. When Mozambique and Angola banned log exports in 2017, exports surged in Namibia. While not a range State itself, Namibia exports an

estimated 250 to 300 containers of mukula (*species not specified*) logs monthly to China from Walvis Bay port—a trade worth some USD 9 to 16 million monthly.

As with other timber species, the mukula trade is connected to illegal trade in other species. In 2016, Chinese customs officials seized a 2.9 t shipment of pangolin scales hidden in a container of mukula timber. In Namibia, the Chinese national identified as owner of the key exports logistics company for Angolan and Zambian clients has also been repeatedly linked to trafficking in rhino horn and animal skins.

Angola: In response to growing concern over harvest and export of mukula, in January 2018 the Ministry of Agriculture suspended "all activities related to the exploitation of forest resources such as felling, movement and transportation of logs" and created a multisectoral commission to inventory seized timber. Immediately thereafter officials seized 540 m^3 (1,880 logs) of wood that police investigators stated had been harvested illegally in Cuando Cubango province and was being prepared for export without proper documentation. Sources describe *Pterocarpus tinctorius* logs being harvested in South-eastern Angola, Zambia and DRC, then transported to Namibia to avoid the export ban.

Burundi: Almost all natural forest and woodland areas in Burundi where *Pterocarpus tinctorius* might be present have been set aside as protected areas where logging is not permitted. Illegal timber from DRC forests and, to a lesser extent, Tanzania's miombo woodlands, is known to be brought across the border and sold locally, but specific reports of mukula logging or trade are not currently available.

Democratic Republic of the Congo: Lubumbashi, the provincial capital of Haut-Katanga, is the hub for trade in *Pterocarpus tinctorius*, with at least 10 Chinese-owned mukula trading companies. A field study by local civil society showed that the volume of trees cut down in five months of 2016 totalled almost 3,300 m^3, approximately five times the number given by the Ministry of Environment, suggesting widespread corruption and illegality. In 2016 the new head of the Ministry's Nature Conservation and Sustainable Development division, addressing the surge in *P. tinctorius* smuggling, determined that logging activities "far exceeded" the limits allowed by government issued permits for artisanal logging. In April 2017, officials in Haut-Katanga arrested 14 Chinese people with tourist visas "suspected of illegally exporting red wood [mukula]". The acting governor stated that 17,000 t of mukula had been illegally exported to China through Zambia over four months, with Zambian officials seizing hundreds of vehicles. An ITTO-sponsored study of regional timber flows conducted surveys of cross-border timber movements at three of the main checkpoints in 2015 between DRC and Zambia. They found 90% of the wood moving across the border from DRC was mukula, 90% of it in debarked logs or cants; the primary crossing was Kasumbalesa. The study extrapolates that 49,804 m^3 of timber were being traded annually through these three checkpoints, of which 44,824 m^3 was mukula. All this timber was recorded as destined for China, via ports in Tanzania, Namibia, Zimbabwe and Botswana.

The extrapolation of 49,804 m^3 of annual cross border trade mukula between DRC and Zambia should be treated with some caution as it is based on nine days of data collection over a three month period (CIFOR, 2016).

Malawi: Reports state that there has been "recent large-scale expansion of mukula harvesting and trade into Zambia's neighbouring countries such as Malawi, Mozambique and the DRC." Increases in illegal harvest of mukula and two other species, as well as trafficking of mukula smuggled from Zambia, caused the government to ban exports of native hardwood logs in 2008.

Mozambique: Mozambique is China's biggest supplier of African logs. Species-specific export data are not available for *Pterocarpus tinctorius* on either end of this trade flow. However, reports suggest a "recent large-scale expansion of mukula harvesting and trade into Zambia's neighbouring countries such as Malawi, Mozambique and the DRC." Field investigations have found evidence of largescale illegal harvest and trafficking of Precious and Class 1 timber species from Mozambique into Tanzania. Chinese buyers often go directly to individuals in the countryside to avoid the costs of obtaining logging licences, obligations to replant etc. Trade data discrepancy analysis between Chinese import data and Mozambique export data shows significant export underreporting. Likewise, almost none of the logs and sawn wood reported as imports by Tanzania (in the order of tens of thousands m^3 yearly) are reported by Mozambique, suggesting the bulk this trade is illegal. Nearly 90% of logs are exported to China, the majority consisting of only five species: *Afzelia quanzensis* (chanfuta), *Millettia stuhlmannii* (jambirre or panga-panga), *Combretum imberbe*, *Swartzia madagascariensis*, and *Pterocarpus angolensis* (umbila). Mozambique implemented regulations to ban log exports of Class 1 timber species in 2007, although roughly sawn timber is considered to be processed. A new log ban was put in place in 2017.

Tanzania: While Tanzanian Forest Service (TFS) data show a sharp increase in export permits for *Pterocarpus tinctorius* (almost 7 times between 2012 and 2014 alone (831.4 to 5,578.4 m^3), TFS reports that this is primarily

logs from Zambia that are transited through Tanzania to the Dar es Salaam port for export to China. They typically enter already in containers—as many as 60 containers a month pass through—and have in some cases been subject to confiscations by Tanzanian authorities due to insufficient documentation.

Zambia: Zambia is "currently facing immense pressure due to widespread illegal harvesting accelerated by its high international demand". Pressure on mukula seems to have begun since at least 2010 in Zambia, with increased presence of Chinese traders. A series of harvest moratoriums and export bans have been imposed and lifted in attempts to control a trade that nevertheless continues to grow. As of June 2017, export of sawn logs not only of mukula but any species is banned, although the Minister of Lands, Natural Resources and Environmental Protection, in consultation with the Director of Forests, "may issue export permits for any timber if that is deemed necessary in the interest of the Republic". Chinese syndicates are reported to be financing harvest in Zambia, south-east Angola and DRC, exporting 250–300 containers of logs monthly via Namibia. Other reports, meanwhile, estimated that national annual production of mukula in Zambia alone was 110,000 m^3—between 1,500 and 2,000 containers of logs, affecting a forest area between 90,000–150,000 ha (assuming 7 stems/ha in high stocked forests and 3–4 in low-stocked). Most of this is technically illegal and transported with bribes corresponding to USD 16–27/log. Chinese customs data indicate a rapid increase in import of logs from Zambia, research indicates that the vast majority of those are mukula logs. Imports went from just over 35,000 m^3 in 2015 to 65,000 m^3 in the first half of 2017 alone. A large portion of this is falsely declared as sawn wood in export documents: in 2016, Zambia declared only 3000 m^3 of log exports (approximate value USD 900,000) while China declared log imports of about 61,000 m^3 (approximate value USD 87 million).

Inclusion in Appendix II to improve control of other listed species

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2 a or listed in Appendix I

Pterocarpus erinaceus *was listed in Appendix II following CoP17. There are no overlaps in range States between this species and* P. tinctorius *(native to West and Central Africa (UNEP-WCMC, 2019)). The structural features of* P. erinaceus *are largely identical with those of other* Pterocarpus *species and also with some of the genus* Dalbergia*; the wood, however, can be distinguished from those with a similar structure due to the light base colour of the heartwood with regular brown colour streaks (Richter* et al., *2014). While undertaking a non-detriment finding for* Pterocarpus tinctorius *(Mukula), Belgium noted that identification is only reliable at genus level (PC24 Doc. 30.4). It was previously recommended that genus level regulation is needed as accurate/affordable species recognition systems are not yet available and if species level regulation was implemented, traders would simply relabel timber under a non-regulated species name (Winfield* et al., *2016).*

Additional Information

Threats

The primary threat to *Pterocarpus tinctorius* is over-harvesting, including both legal and widespread illegal extraction, for the international trade. The threat represented by this trade is compounded by deforestation from forest conversion and aridification due to climate change and more severe fires.

Conservation, management and legislation

National legislation: See table below. Note that this table is focused on national laws and regulatory measures. However, in many of the range States, traditional chiefs have significant authority over harvest rights in their villages or territories. Their decisions may not always be in line with the national government's policies.

Country	Special measures for protection and management of the species	Export-related regulation
Angola	None	Partial log export ban since 2017. Wood can only be exported upon presentation of "proof of deposit of the corresponding value in one of the country's banks or a credit note".
Burundi	None	None
DRC	One news article reports that the government in Kinshasa has restricted harvesting of mukula, but unconfirmed.	DRC has a Voluntary Partnership Agreement with the European Union
Malawi	The Forest Regulations under the Malawi Forest Act lists indigenous fine hardwood species, including closely related *Pterocarpus angolensis* as protected tree species.	Native hardwood log exports banned since 2008.
Mozambique	None	Ban on export of unprocessed logs of precious and Class 1 species since 2007. Additional ban since 2017.

Tanzania	None	2002 Forest Act regulations require that sawn wood be issued a Grading Certificate and Export Licence from TFS. Log export ban since 2004.
Zambia	Forest Act No. 4 of 2015. A series of bans on commercial harvesting and trade since 2013. Moratorium on harvest and movement in place since 2017. The legal limit to harvest mukula is set (since 2013) at 30 cm top diameter over-bark (it was 40 cm previously).	Log export ban (since 2017). However, Ministry of Lands, Natural Resources, and Environmental Protection, in consultation with Director of Forestry, can issue export permits for timber "deemed necessary in the interest of the Republic".

International legislation: There are no international controls specifically related to *Pterocarpus tinctorius* in place. Imports to the USA, European Union and Australia are subject to national legislation in those jurisdictions prohibiting the import and/or sale of wood which was illegally sourced in the country of origin. However, little or no African rosewood is traded to these countries. Chinese companies may choose to operate under Voluntary Guidelines called the Guide on Sustainable Overseas Forest Management and Utilization by Chinese Enterprises.

Management measures are defined by each range State's forest legislation, which define aspects including minimum cutting diameters and areas off-limits from harvest activity such as parks and other protected areas, riparian corridors, steep slopes etc. In practice these measures are unevenly enforced and monitoring of timber harvesting is not routinely undertaken.

Artificial propagation/captive breeding
Pterocarpus tinctorius can be propagated by either seed or cuttings, and wild seedlings can also be collected for planting. However, there is little to no information available regarding artificial propagation for commercial purposes. Genetic exploration using tissue culture has not been done. At present, almost all harvest of this slow-growing species appears to be from wild sources.

There has also been very little success with artificial propagation in similar species such as Pterocarpus angolensis*. Reasons for these failures are listed as low germination rates, slow tree growth and high levels of competition for sunlight (Mojeremane & Lumbile, 2016).*

Implementation challenges (including similar species)
There exists some confusion as to whether the common name used by traders in Zambia, mukula, refers specifically to *Pterocarpus tinctorius* or to a rosewood species complex. The same may be the case in other range States. These look-alike issues are a complexity of the rosewood trade in general and were the key factor in listing of the entire *Dalbergia* genus in Appendix II in 2016. *Pterocarpus angolensis* (common names: muka, kiaat, African teak) is a species of the miombio woodland savannas of Eastern and Southern Africa with similar morphological and timber characteristics to *P. tinctorius*. This species is a keystone of domestic timber markets. IUCN Red List has assessed *P. angolensis* as Near Threatened. *Pterocarpus soyauxii* (common names: Padauk d'Afrique) is another highly sought-after timber with rosewood properties.

It has been reported that Pterocarpus angolensis *and* P. tinctorius *are regularly mixed and traded together as one species in areas where their ranges overlap in Zambia and Tanzania (Cunningham, 2016).*

Identification of timber species is known to be very difficult for genera such as Pterocarpus *and* Dalbergia*. A lot of more technologically advanced methods such as DNA analysis, chemical analysis and isotope analysis are still in their infancy and are thought to be some years off being available at the local level for timber identification (Winfield* et al., *2016).*

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Amendment of the annotation to the listing of Bitter Aloe *Aloe ferox*

Proponent: South Africa

Summary: *Aloe ferox* is a medicinal plant native to South Africa and Lesotho that has been included in CITES Appendix II since 1975 as part of the genus level listing of *Aloe* spp.
The current annotation to the listing #4 includes the following:

All parts and derivatives, except:
a) seeds (including seedpods of Orchidaceae), spores and pollen (including pollinia). The exemption does not apply to seeds from Cactaceae species exported from Mexico, and to seeds from *Beccariophoenix madagascariensis* and *Dypsis decaryi* exported from Madagascar;
b) seedling or tissue cultures obtained in vitro, in solid or liquid media, transported in sterile containers;
c) cut flowers of artificially propagated plants;
d) fruits, and parts and derivatives thereof, of naturalized or artificially propagated plants of the genus *Vanilla* (Orchidaceae) and of the family Cactaceae;
e) stems, flowers, and parts and derivatives thereof, of naturalized or artificially propagated plants of the genera *Opuntia* subgenus *Opuntia* and *Selenicereus* (Cactaceae); and
f) finished products of *Euphorbia antisyphilitica* packaged and ready for retail trade.

The Proponents seek to amend the annotation so that part f) reads:
finished products[1] of *Aloe ferox* and *Euphorbia antisyphilitica* packaged and ready for retail trade.

The proposed amendment provides a footnote defining "finished product" as follows:
[1]This term, as used in the CITES Appendices refers to products, shipped singly or in bulk, requiring no further processing, packaged, labelled for final use or the retail trade in a state fit for being sold to or used by the general public.

Almost all trade in wild *Aloe ferox* originates from South Africa. Bitter sap ("bitters") is extracted from harvested leaves, crystalised and sometimes ground to powder then exported. Recently, further processing of secondary extracts has been taking place including the inner leaf jelly as a juice, gel or powder, and these products too have been increasingly going into international trade. End uses of *Aloe ferox* products include health drinks, medicines and a range of healthcare and cosmetics products. South Africa's exports have been dominated by extracts (bitters) however, there has been increasing trade reported as derivatives and in the years 2013-2015 exports of derivatives exceeded those of extract by gross weight. South Africa states that its exports of derivatives refer to finished products and the quantities of *Aloe ferox* in these are minimal. However, studies provided on concentrations of *Aloe ferox* extracts contained in finished products do not help to clarify the significance of export quantities reported as derivatives.

Other species in the Appendices are also annotated with #4 but would not be affected by this amendment.

Analysis: According to *Res. Conf. 11.21 (Rev. CoP17)* annotations should concentrate on those commodities that first appear in international trade as exports from the range State and include only those commodities that dominate the trade and the demand for the wild resource. South Africa exports large quantities of wild sourced *Aloe ferox* extract and derivatives, the latter having increased proportionately in recent years apparently due to increased processing of finished products in South Africa. South Africa has said that most of the derivatives they have reported are finished products packaged and ready for retail trade and propose that they be excluded from CITES controls by the proposed amendment to the annotation. Exports of derivatives have been increasing over the

last 10 years and in some years the total weight reported (which might include significant amounts of other ingredients) has exceeded the export by weight of the primary extract. These derivatives, or finished products are commodities that first appear in international trade. If they are becoming dominant in the volumes of exports their exemption would not be in line with the guidance in *Res. Conf. 11.21 (Rev. CoP17).* However, this is not possible to ascertain without more detailed insight into the concentration of primary and secondary *Aloe ferox* extracts in the products exported.

If the amendment to exclude finished products of *Aloe ferox* is adopted, it would not be necessary to include a footnote defining "finished products" as this definition is the same as that provided in the Interpretation text of the Appendices and therefore does not require a footnote defining it specifically in this annotation.

Aloe ferox would only be differentiated from other *Aloe* species on the basis of ingredient lists.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS

Range
South Africa and Lesotho.

IUCN Global Category
Not assessed.

A) Background
Aloe ferox was listed in the Appendices in 1975 as part of the genus level listing for *Aloe*. Its distribution is within South Africa's Western and Eastern Cape Province, the south-east of Free State and in southern Lesotho. Its distribution is around 168,000 km^2, occurring intermittently in dense stands across this range. Population numbers or trends are not available. The Red Data list of southern African plants considered it as least concern in South Africa and Lesotho. Current levels of offtake are considered to be sustainable; although localised damage and even extinctions have resulted from over-exploitation and destructive harvesting by untrained harvesters, particularly in the Eastern Cape Province. Longer-term impacts of high levels of harvesting on populations remains unknown.

Harvesting is mainly of leaves from which the aloe bitters are extracted. Generally between 6-30 leaves can be harvested from each plant every 1.8 years depending on the plant's health. Harvested leaves are "tapped" to remove the bitter sap ("bitters") which are further processed with this extract historically being the dominant product in trade. Recently secondary extracts have also been increasingly going into international trade including the inner leaf jelly as a juice, gel or powder. It is a key ingredient in laxatives; other end uses include health drinks, medicines and a range of healthcare and cosmetics products. In recent years there has been further extraction of secondary products from the leaves post removal of the bitters; aloe gel is derived from the white spongy mesophyll layer of the leaf. Both primary and secondary extract are further processed to make finished products.

South Africa is the only range State exporting commercial levels of products. *Lesotho's reported exports of wild harvested Aloe ferox have been minimal; 10 live plants imported to South Africa (according to the CITES Trade Database).*

Assessment against the guidance in Res. Conf. 11.21 (Rev. CoP17) Use of annotations in Appendices I and II
This Resolution recommends the following guidance and principles for annotations:

b) two main principles be followed as standard guidance when drafting annotations for plants:
i) controls should concentrate on those commodities that first appear in international trade as exports from range States; these may range from crude to processed material; and
ii) controls should include only those commodities that dominate the trade and the demand for the wild resource

A rapid assessment was carried out of exports reported by South Africa (in the CITES Trade Database) in weight (the majority of reported products) to determine which commodities dominate the exports from range States. Figure 1 shows that extract has dominated the trade over the period between 2000 and 2016, however in the

years 2013-2015 exports of derivatives outweighed those of extract and powder. It is likely that if derivatives are finished products that a significant part of the quantity is of other ingredients and packaging. In addition to products reported in weight over this period, items with no unit (presumed individual items) exported from South Africa accounted for ca 720,000 derivatives and ca 61,000 items of extract.

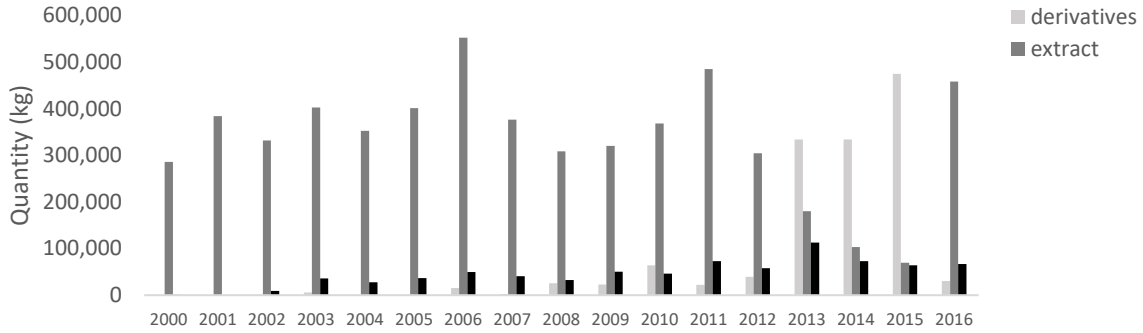

***Figure 1**: Quantity of main commodities of* Aloe ferox *in trade reported as weight by South Africa (source: CITES Trade Database).*

According to South Africa, exports of *Aloe ferox* reported as "derivatives" refer to "finished products". *The CITES glossary describes derivatives as "Any processed part of an animal or plant (e.g. medicine, perfume, watch strap)".* The increases in exports of derivatives (reported in weight) in recent years reflects a development of the in-country processing. Ten companies in the country are involved in the creation, distribution and sale of ready packaged consumer goods containing. *It remains unclear whether all exports reported as derivatives are indeed of finished products packaged and ready for retail trade.*

According to the SS, exports of derivatives may contain minimal amounts of *Aloe ferox* material and that much of the *Aloe ferox* material in these finished products is derived from the secondary extracts from the leaves of the *Aloe ferox* from which bitters have already been removed. The SS reasons that it is primary extracts that are driving the harvest of leaves and therefore the exemption of finished products (which are mainly derived from secondary extracts) would not have any impact on the total amount of *Aloe ferox* harvested, *assuming that the amount of leaves harvested for bitters extract is enough fulfil the demand for secondary extract*. A limited study included in the SS of finished products found for sale online in South Africa and internationally found that:

- Primary extract
  - 15% of the items contained more than 80% of primary extract
  - only 6% contained less than 50% of primary extract.
- Secondary extract:
  - most (66%) products had less than 50% of secondary extract
  - 13% contained more than 80% of secondary extract

However, it is not possible to extrapolate the figures in this study to the exports of finished products from South Africa as there is no indication of whether the products in the study have been manufactured in South Africa or elsewhere. The vast majority of those finished products could contain anything below 50%. No study is known to have been made on the contents of exports of finished products from South Africa and the actual quantity of Aloe ferox product being exported in this form is unknown and nor the proportion of it that is likely to be secondary versus primary extract.

Other species in the Appendices are also annotated with #4, but would not be affected by this amendment.

Additional Information

Potential benefit(s) of listing for trade regulation

Significant quantities of *Aloe ferox* products are re-exported subsequently by non-range States. The Proponent argues that the exemption of finished products would significantly reduce the burden on Parties (both importing and exporting) from this trade by eliminating the need to inspect consignments of finished products which contain minimal amount. *Germany has stated that their customs seized more than 3000 pieces of such products in 2018, in most cases cosmetics but also food supplements containing* Aloe ferox*. Seizures are primarily finished products for personal consumption, ordered by internet and shipped by post to the recipients (M. Stertz, in litt. 2019).*

References

Stertz, M. (2019) *in litt.* to IUCN/ TRAFFIC Analyses Team, Cambridge UK.

Amend annotation #16 to the listing of Grandidier's Baobab *Adansonia grandidieri* in Appendix II by deleting reference to live plants

Proponent: Switzerland

Summary: Grandidier's Baobab *Adansonia grandidieri*, a species of baobab tree endemic to Madagascar, was included in Appendix II at CoP17 with annotation #16 “seeds, fruits, oils and live plants” to indicate the parts and derivatives that were covered by the listing. Switzerland, as the Depositary Government for the Convention, draws attention to the fact that the inclusion of the term “live plants” is redundant, inconsistent with other listings and potentially misleading. This is because, according to Article I of the Convention and *Res. Conf. 11.21 (Rev. CoP17),* live plants (and whole dead plants) are automatically covered by listings in the Appendices. By including reference to live plants in #16, and not in any other annotations, it could be mistakenly interpreted that live plants were not covered by those other annotations. The original intent of listing *Adansonia grandidieri* with #16 was to ensure that enforcement officers would be aware of the full extent of the listing.

Switzerland suggests that the interpretation section of the Appendices be changed to emphasize the fact that all live and whole dead plants (and animals) are always included in listings. The Standing Committee Working Group on Annotations proposed an amendment to paragraph 7 that serves this purpose (see SC70 Doc. 67.1 Annex 2). This amendment will be considered at CoP18 (see CoP18 Doc. 101).

Analysis: The proposal is sound and in full accord with the provisions of the Convention.

Inclusion of all species of the genus *Cedrela* in Appendix II

Proponent: Ecuador

Summary: *Cedrela* is a genus of tree with 17 species occurring in Mexico and the Caribbean islands south to Argentina. *Cedrela odorata* is the most widespread species and appears to be the most highly traded species internationally, although other species are also used for their valuable timber.

Cedrela odorata has been listed in Appendix III by Colombia and Peru since 2001, by Guatemala since 2008, by Bolivia since 2010, and by Brazil since 2011. Two other species in the genus, *C. fissilis* and *C. lilloi,* have been listed in Appendix III by Bolivia and Brazil since 2010 and 2016 respectively. All listed populations are covered by annotation #5 (logs, sawn wood and veneer sheets).

Cedrela odorata was assessed as globally Vulnerable with a decreasing population trend on the IUCN Red List in 2017, with the unsustainable harvest of timber cited as the main threat. Many populations appear to have been severely depleted by targeted over-exploitation, are categorised as nationally endangered or vulnerable, and are subject to laws and other measures to regulate harvest. Illegal trade has been reported. Extensive loss of habitat also threatens the species; deforestation data indicate that the range has decreased by 29% in the last 100 years, and is estimated to decline by 40% in the next 100 years.

The wood of *C. odorata* is used extensively for furniture making and other purposes. According to the CITES Trade Database, large quantities of sawn wood have been exported by Peru, Bolivia and Brazil (noting that data reported to CITES primarily reflect exports from range States with Appendix III-listed populations), as well as non-range States where plantations have been established. The principal importers were the USA and Mexico (43% and 33% of total reported imports from 2007-2016, respectively). Available data for the principal range State exporters indicate that domestic trade exceeds international trade (annual average of 72,000 m^3 relative to 46,000 m^3 for Bolivia, Brazil and Peru combined over the period 2004-2008).

There was a substantial increase in exports and prices of *C. odorata* timber following the 2003 listing of Big-leaf Mahogany *Swietenia macrophylla* in Appendix II. Reported exports of *C. odorata* timber peaked at over 60,000 m^3 in 2007 but subsequently declined to under 10,000 m^3 in 2010. Exports then increased slightly with the listing of the Bolivian and Brazilian populations in 2010/2011, and remained relatively stable at around 14,000 m^3 per year from 2014 to 2016.

Cedrela odorata has been planted widely in parts of the region and introduced to many countries elsewhere. Although monospecific plantations have not generally been successful in the tropical Americas due to vulnerability to the Shoot Borer *Hypsipyla grandella*, in other regions monospecific plantations are well established. The vast majority of reported exports from plantations ("artificially propagated") were from non-range States (Côte d'Ivoire and Ghana). Although exports from plantations exceeded exports from the wild in every year since 2013, there was an overall decline in exports from plantations from 2013 (over 12,000 m^3) to 2016 (ca. 8,000 m^3).

Other species
Both *C. fissilis* and *C. lilloi* are also widely distributed and categorised as globally threatened (Vulnerable and Endangered respectively), with certain national populations also categorised as threatened. Over-exploitation for timber has been reported to be a threat, in addition to habitat loss.

While *C. fissilis* timber is considered inferior to that of *C. odorata*, timber of the two species is reportedly marketed interchangeably. In Ecuador, it was reported in 2018 that most wild populations of *C. fissilis* had been destroyed and the remaining large trees were being felled for export to Colombia. Total exports of *C. fissilis* reported in the CITES Trade Database primarily comprised 1,650 m^3 wild-sourced sawn wood and 6,400 m^2 source "I" veneer (the majority exported from

Brazil); no trade was reported from 2014 onwards. ITTO reports include exports of *C. fissilis* totaling ca. 83,000 m^3 sawn wood (60% from Bolivia and the remainder from Brazil) in the period 2002-2016; exports showed a marked overall decrease from 17,000 m^3 in 2002 to 2,000 m^3 in 2015 (no exports were reported in 2016).

No exports of *C. lilloi* have been reported. Many of the remaining species in the genus are reported to be threatened in all or part of their range due to a combination of deforestation and targeted over-exploitation, although demand for timber of these species appears to be primarily domestic.

The USA, which appears to be one of the principal importers of *Cedrela,* reported imports of sawn/chipped wood (HS code 4407) of unspecified *Cedrela* species totaling 144,663 m^3 from 2007-2018. The principal exporters were Peru (21%), Côte d'Ivoire (18%), Ghana (15%), Bolivia (15%) and China (10%). It is not clear if exports from non-range States are re-exports or originate from plantations in those countries.

Although identification manuals have been developed to differentiate the woods of certain *Cedrela* species, several range States have reported identification difficulties and according to one expert it is not possible to distinguish between species in the genus based on either macroscopic or microscopic characters of the wood.

Analysis: *Cedrela* is a genus of New World trees of which *C. odorata* is the most widespread species. *Cedrela odorata* has been intensively exploited for its timber, for both domestic and international trade. Based on available data, the principal exporters of *C. odorata* appear to be Bolivia, Brazil, Peru, Côte d'Ivoire and Ghana. Although the timber of certain other species is also reported to be valuable and can be marketed interchangeably with *C. odorata*, it is not clear whether there is significant international demand for other species. Some populations of *C. odorata* and several other species are known to have been substantially reduced by the combined effects of deforestation and targeted over-exploitation. Given the estimated historic and future declines for *C. odorata,* and significant historic impact of international trade, the species may meet the criteria for inclusion in Appendix II set out in Annex 2a of *Res. Conf. 9.24 (Rev. CoP17)*. Given the reported identification difficulties, the remaining species in the genus would appear to meet the criteria for inclusion in Annex 2b.

Other Considerations: The proposal does not include an annotation. However, the scope could be restricted using an annotation that covers the main products in trade (all populations currently included in Appendix III are covered by annotation #5). Sawn wood has been the most common product in reported international trade.

Summary of Available Information

Text in non-italics is based on information in the Proposal and Supporting Statement (SS); text in italics is based on additional information and/or assessment of information in the SS.

Taxonomy

The proposal recognises 17 species in the genus. *C. odorata* has a large number of synonyms included in Annex 2 of the proposal.

Range

C. balansae: Argentina, Bolivia, Paraguay
C. discolor: Mexico
C. dugesii: Mexico
C. fissilis: Argentina, Bolivia, Brazil, Colombia, Ecuador, El Salvador, Guyana, Paraguay, Peru, Venezuela; *Costa Rica, Panama (Barstow, 2018)*
C. kuelapensis: Peru, Ecuador
C. lilloi: Argentina, Bolivia, Ecuador, Peru; *Brazil, Paraguay (Llamozas, 1998)*
C. longipetiolulata: Peru
C. molinensis: Peru
C. monroensis: El Salvador
C. montana: Colombia, Ecuador, Peru, Venezuela

C. nebulosa: Colombia, Ecuador, Peru
C. oaxacensis: Mexico
C. odorata: *Antigua and Barbuda, Argentina, Barbados, Belize, Bolivia, Plurinational States of, Brazil, Cayman Islands, Colombia, Costa Rica, Cuba, Dominica, Dominican Republic, Ecuador, El Salvador, French Guiana, Grenada, Guadeloupe, Guatemala, Guyana, Haiti, Honduras, Jamaica, Mexico, Montserrat, Nicaragua, Panama, Peru, Saint Kitts and Nevis, Saint Lucia, Suriname, Venezuela (Mark & Rivers, 2017)*
C. saltensis: Argentina, Bolivia, Peru
C. salvadorensis: Costa Rica, El Salvador, Guatemala, Honduras, Mexico
C. tonduzii: Costa Rica, El Salvador, Guatemala, Honduras, Mexico, Nicaragua, Panama
C. weberbaueri: Peru

IUCN Global Category
C. odorata: Vulnerable A3bcd+4bcd (Assessed 2017, Criteria version 3.1)
C. fissilis: Vulnerable A2cd+3cd (Assessed 2017, Criteria version 3.1)
C. lilloi: Endangered A1a+2cd (Assessed 1998, Criteria version 2.3)
The remaining species in the genus have not yet been assessed by IUCN.

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP17) Annex 2a)

Cedrela odorata
Cedrela odorata *is one of the most widely used tropical hardwoods in Central and South America, used for furniture and cabinet-making, panelling and joinery in general (Pennington & Muellner, 2010). In 2011 it was reported to be the second most valuable species in Latin America and the Caribbean after Big-leaf Mahogany* Swietenia macrophylla *(Pérez Contreras, 2011).*

Although estimates of the current total population of *C. odorata* are not available, populations of this species have been reduced by selective logging for at least 250 years, especially of the largest individuals, and are now in decline (Mark & Rivers, 2017). *Cintron (1990) reported that although widespread,* C. odorata *was not common in moist tropical American forests and its numbers continued to be reduced by exploitation without successful regeneration. In Central America, although the greatest threat to natural forests is conversion of land to other uses, selective harvesting of timber was reported to particularly threaten native species including* C. odorata *(UNEP, 2003). Cavers* et al. *(2004) reported that genetic erosion of this species had already occurred throughout its natural distribution due to significant over-exploitation, and trees of good form were rarely found except in isolated areas. Pennington & Muellner (2010) report that across the species' range it was difficult to find large trees (1m DBH or more) in natural forest, except in protected areas, because of the species' long history of logging.*

The species' habitat is highly fragmented by deforestation (Mark & Rivers, 2017). The total distribution range of *C. odorata* was reported to have decreased by 29% in the last 100 years (approximately three generations), and it was estimated that it will be reduced by a further 40% in the next 100 years *(Mark, 2017)*.

Available data on population sizes/trends for specific areas is summarised below:

- Amazon: Reported to be scarce in sample units across the Amazon.
- *Argentina: Range reported to have declined in 2004 (Zapater* et al., *2004).*
- *Barbados: Formerly common and widespread, but had become rare by 1965 (Gooding* et al., *1965).*
- Bolivia: In 2011 it was reported that no distribution/density studies had been conducted in Bolivia.
- Brazil: In 2011 it was reported that no distribution/density studies had been conducted in Brazil. Nationally assessed as vulnerable.
- Cayman Islands: Nationally assessed as critically endangered.
- Colombia: Average density reported to be 0.39 individuals / ha in 2015. The selective extraction that this species had suffered for years was reported to have affected the presence of large trees and the availability of seed; the density of trees with DBH> 80 cm (proposed minimum cutting diameter) was reportedly zero or close to zero, suggesting a clear depletion of harvestable trees in natural forests nationwide. Nationally assessed as endangered.
- Costa Rica: The species was reported in 1999 to be threatened due to heavy exploitation, and in 2006 was reported to be in a vulnerable condition due to timber extraction and a 57% reduction in its habitat*, but a study by Rivera* et al*. (2010) concluded that the population in Costa Rica was increasing*. The number of individuals of *C. odorata* was estimated at 12,110 in its natural distribution area in 2010, equivalent to a population density of 0.96 individuals / km.
- Dominican Republic: Nationally assessed as critically endangered.
- Ecuador: A 2013 study reported that small, fragmented populations of *C. odorata* made up mostly of young individuals were observed in areas where forest extraction had been more intensive in the

country, while in areas where there had been no extraction, a greater density of individuals and predominance of the major diameter classes was found. *C. odorata* has lost 34% of its habitat in the country and is assessed nationally as vulnerable.

- *El Salvador: Included in an official list of species threatened with extinction in the country in 1997 (Anon., 1997).*
- *Guatemala: Numbers reported to have been greatly reduced by intensive logging by 1946 (Standley & Steyermark, 1946).* Nationally assessed as vulnerable.
- Mexico: A 2017 study estimated 1.40 ± 0.93 million trees of *C. odorata* in the Pacific region, 4.52 ± 1.75 million trees in the South-Southeast region and 9.01 ± 2.85 million trees in the Gulf of Mexico, totalling ca. 14.98 million trees *(the SS indicates that these figures do not differentiate between wild trees or plantations, but the study specifies natural forests).* It was estimated that the population was stable.
- *Panama: Reported to be rare but previously common in certain areas, and most trees less than 50cm in diameter; large individuals likely to have been harvested (Condit & Pérez, 2002).*
- Peru: A 2011 study estimated the total population at 1.1 million trees with a commercial population between 261,159 and 300,743 trees and densities of up to 1.15 individuals / ha. Nationally assessed as vulnerable.
- *Puerto Rico: Native trees reported to have been reduced to scattered remote areas by 1964 (Little & Wadsworth, 1964).* Nationally assessed as critically endangered.

There was a substantial increase in exports and prices of C. odorata *sawn wood coinciding with the listing of* Swietenia macrophylla *in CITES Appendix II in 2003 (which led to a marked decrease in exports of mahogany, and increase in price of the latter, from 2000-2008).*

Cedrela odorata has been listed in Appendix III by Colombia and Peru since 2001, by Guatemala since 2008, by Bolivia since 2010, and by Brazil since 2011. *The CITES Trade Database primarily reflects exports from the latter countries since the reporting of trade from non-listed populations is not required.*

According to data included in the CITES Trade Database, exporting countries reported total direct exports of 236,779 m^3 and 409,180 m^2 of C. odorata wood and wood products over the 10-year period 2007-2016. Of the trade reported in m^3:
- The vast majority of trade was reported as "sawn wood" (95%) and "timber" (4.5%), with much smaller quantities reported as logs, veneer and plywood;
- 75% was reported as source "W" (176,555 m^3) and 25% reported as source "A";
- Of the source "A" exports reported by exporters, 99% were from non-range States (Cote d'Ivoire and Ghana);
- The top source "W" exporters were Peru (50%), Bolivia (37%) and Brazil (7%).
Importing countries reported total imports (in m^3) around 30% lower than reported exports; according to importer-reported data, the top importers over the period 2007-2016 overall were the USA (43%) and Mexico (33%), with a similar pattern for wild-sourced imports only (USA 41%, Mexico 38%).

Figure 1 below shows exporter-reported trade in source "A" and "W" Cedrela odorata *recorded in the CITES Trade Database from 2001-2016:*
- Total source "W" exports reported by exporters peaked in 2007 at 61,378 m^3 and subsequently declined to 4,731 m^3 in 2014, with small increases in both 2015 and 2016 (Figure 1), mainly reflecting increases in exports from Bolivia and Brazil in these years;
- Total source "A" exports reported by exporters peaked in 2013 at 12,774 m^3 and subsequently declined to 8,086 m^3 in 2016.

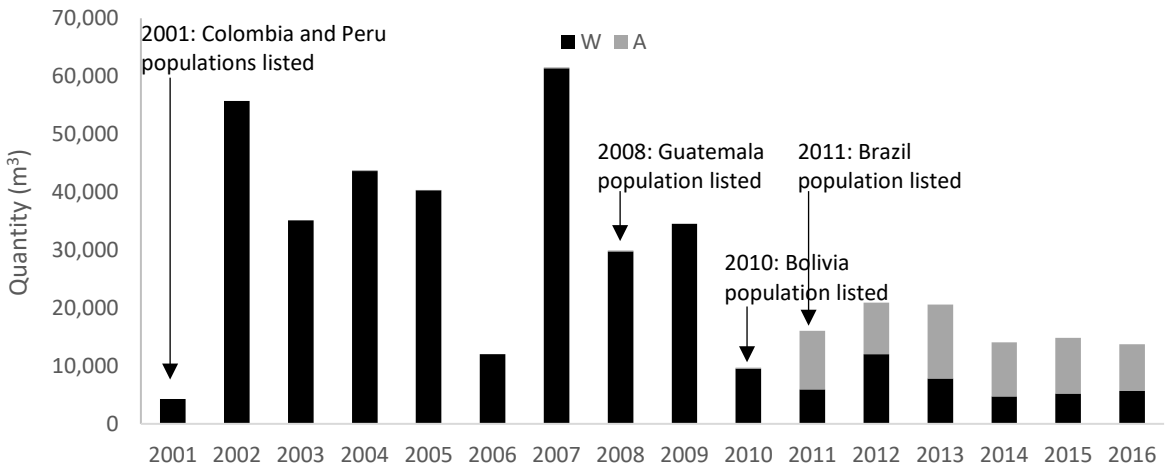

Figure 2 *Exporter-reported trade in source "A" and "W"* Cedrela odorata *recorded in the CITES Trade Database.*

ITTO reports (ITTO, 2005-2017) include the following trade in C. odorata *involving ITTO member countries over the period 2002-2016 (trading partner countries were not specified):*
- *Total exports of ca. 14,000 m³ sawn wood (15,000 m³ from Ghana (non-range State) in the period 2011-2015 only, and the remainder from Guyana in 2006 only) and ca. 2,000 m³ logs (all from Mexico in 2002 only);*
- *Total imports of ca. 6,000 m³ logs (all to Mexico in the period 2002-2005 only).*

*Trade was also reported as "*Cedrela spp.*" (species unspecified) in both the CITES Trade Database and ITTO reports; see next section for details.*

Data on domestic consumption of C. odorata *versus exports for top producing countries is detailed below:*
- *Bolivia: Exports between 2004-2008 averaged 10 thousand m³ / year (a maximum of 16 thousand m³ in 2004 decreasing to 7 thousand m³ in 2008). Domestic consumption was estimated at 4 thousand m³ / year from 2004-2008 (Pérez Contreras, 2011).*
- *Brazil: Exports between 2007-2009 averaged 14 thousand m³ / year. Domestic consumption data not available for the country as a whole but was at least 24.8 thousand m³ / year on average from 2007-2009 (Pérez Contreras, 2011).*
- *Peru: Exports between 2000-2008 averaged 22 thousand m³ / year (1.5 thousand m³ in 2000, increasing progressively - and parallel to the decrease in exports of mahogany to the USA - up to a maximum of 55 thousand m³ in 2007). Domestic consumption 2000-2008 averaged 43 thousand m³ / year (maximum of 62 thousand m³ in 2004, decreasing to 10 thousand m³ at the end of the period) (Pérez Contreras, 2011).*

Rivera et al. *(2010) report that exports of* C. odorata *from Costa Rica were primarily of manufactured products (e.g. furniture) rather than raw timber, and that the raw timber primarily originated in Nicaragua and Panama, the majority of which appeared to be moving into Costa Rica illegally. CoP14 Prop. 33 includes multiple reports of illegal logging of* C. odorata *in several range States.*

Cedrela fissilis
The species is very widespread but the population is experiencing significant decline as a result of large scale forest clearance across its range (Barstow, 2018). The species is also exploited for its valuable and desirable timber; the timber is considered inferior to that of C. odorata *(Pennington & Muellner, 2010; Barstow, 2018) but can be sold interchangeably (Barstow, 2018). Although the species has been reported to be widely protected and cultivated for its timber (Pennington & Muellner, 2010), logging has led to many subpopulations of the species becoming extinct (CNCFlora, 2012) and has caused a decline in genetic diversity (Nunes* et al.*, 2007). It is estimated that across the species range there has been at least a 30% population decline over the last three generations (150–300 years), and is predicted to decline at least 30% further over the coming century as a result of continued illegal logging and forest clearance (Barstow, 2018). Available data on population sizes/trends for specific areas are summarised below:*
- *Argentina: Subpopulations are restricted to the north, partly within sub-Andean piedmont forest, a habitat which is under severe threat (Barstow, 2018).*
- *Bolivia: The species has become rare and has been lost from some sectors, though it is mostly harvested opportunistically and is considered to have high germination and good success in populating*

new areas. Assessed nationally as vulnerable due to historical and projected population decline (Arrázola et al., *2018).*
- *Brazil: Although reported as a common and widespread species in central and eastern Brazil, much of its former range within the southeast of the country had been cleared for agro-industry (Pennington & Muellner, 2010). Subpopulations were estimated to have declined by 30% as a result of logging and habitat loss, which have led to the extinction of some subpopulations over time; the species no longer forms dense stands although individual trees may be frequent across the fragmented landscape (CNCFlora 2012). Assessed nationally as vulnerable (CNCFlora 2012).*
- *Colombia: Overexploitation has resulted in the species becoming threatened (Barstow, 2018).*
- *Costa Rica: There are very few individuals in Costa Rica, if any at all (Barstow, 2018).*
- *Ecuador: Most natural subpopulations have been destroyed; some large trees remain but they are reportedly being felled for export to Colombia (Barstow, 2018).*
- *El Salvador: Threatened with extinction (Anon., 1997).*
- *Panama: Few individuals (Barstow, 2018).*
- *Paraguay: Apparently still abundant in the Región Oriental, especially along the Paraná valley (Barstow, 2018).*
- *Peru: Overexploitation has resulted in the species becoming threatened in Amazonian Peru (Barstow, 2018).* Assessed nationally as vulnerable.
- *Suriname: The species is still reasonably common (Barstow, 2018).*

The Bolivian and Brazilian populations were included in Appendix III in 2010 and 2016 respectively, and were included in Annex D of the EU Wildlife Trade Regulations in 2008 (the Bolivian and Brazilian populations were transferred to Annex C in 2012 and 2016, respectively). According to data included in the CITES Trade Database, the only trade in C. fissilis *reported by exporting countries was 267 m^3 wild-sourced sawn wood exported by Bolivia for commercial purposes over the period 2012-2014. Importers reported 1,381 m^3 sawn wood and 4 m^3 veneer (all wild-sourced and for commercial purposes) from Brazil (98%), Bolivia (2%) and Peru (<1%) over the period 2008-2014; the main importer was Germany. The USA also reported the import of 6,428 m^2 source "I" veneer from Brazil in 2013.*

ITTO reports (ITTO, 2005-2017) include the following trade in C. fissilis *involving ITTO member countries over the period 2002-2016 (trading partner countries were not specified):*
- *Total exports of ca. 83,000 m^3 sawn wood (50,000 m^3 from Bolivia in the period 2002-2005 only, and the remainder from Brazil in the period 2008-2015 only) and ca. 2,000 m^3 veneer (all from Brazil in 2008-2009 only). Exports decreased over this period overall from 17,000 m^3 in 2002 to 2,000 m^3 in 2015; no trade was reported in 2006, 2007 or 2016 (see Figure 2 below).*

Total imports of ca. 1,000 m^3 veneer (all to Brazil in 2008 only).

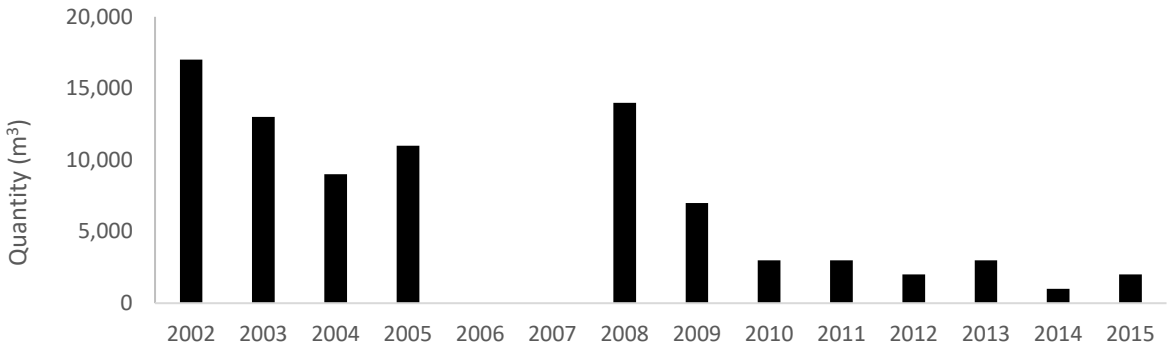

Figure 3 *Exporter-reported trade in* Cedrela fissilis *recorded in ITTO reports (ITTO, 2005-2017)*

Cedrela spp.
Trade was reported in the CITES Trade Database as Cedrela spp. *(species unspecified) in the period 2009-2013 from Guatemala (32 m^3 timber reported without a source or purpose), Nicaragua (20 m^3 wild-sourced timber for commercial purposes) and Suriname (21,140 wild-sourced sawn wood, reported without units, for commercial purposes). ITTO reports (ITTO, 2015 & 2017) include the following trade reported as* Cedrela spp. *involving ITTO member countries over the period 2003-2016: ca. 6,000 m^3 sawn wood imported by Japan in the period 2003-2011, and ca. 1,000 m^3 logs imported by Egypt in 2003 (further imports were also reported but lumped with other non-*Cedrela *species so the quantities of* Cedrela *species are not known).*

The USA reported trade in Cedrela spp. *(species unspecified) under four HS codes between 2000–2018 (USTIC DataWeb, 2019):*

- *HS4407.29.01.60 (Wood sawn or chipped lengthwise, sliced or peeled, whether or not planed, sanded or end-jointed, of a thickness exceeding 6 mm)*
- *HS4412.31.26.10 (Plywood, veneered panels and similar laminated wood)*
- *HS4412.34.26.00 (Plywood, veneered panels and similar laminated wood: Other plywood consisting solely of sheets of wood (other than bamboo), each ply not exceeding 6 mm in thickness: Other, with at least one outer ply of non-coniferous wood not specified under subheading 4412.33: 4412.34 Not surface covered, or surface covered with a clear or transparent material which does not obscure the grain texture or markings of the face ply)*
- *HS4412.94.31.11 (Blockboard, laminboard and battenboard: With at least one outer ply of non-coniferous wood: Plywood: Not surface covered, or surface covered with a clear or transparent material which does not obscure the grain, texture or markings of the face ply).*

The main Cedrela spp. *commodity in trade was sawn/chipped wood (HS4407.29.01.60); a total of 144,663 m^3 was imported into the USA between 2007–2018 (no trade was reported using this HS code prior to 2007). Imports were at their highest in 2007 (ca. 35,000 m^3) and subsequently declined in 2008 and 2009, but has since remained at a fairly constant level (between ca. 6,000 – 10,800 m^3) (Figure 3). The main exporters of sawn/chipped wood (HS4407.29.01.60) into the USA are listed in Table 1 below; top exporters were Peru (21%), Côte d'Ivoire (18%), Ghana (15%), Bolivia (15%) and China (10%).*

Other commodities were not traded in such high volumes or as regularly, but trade reported for each of the other HS codes was:

- *HS4412.31.26.10 – Trade was only reported in 2017 and 2018; a total of ca. 7,000 m^3 was imported into the USA from Brazil, mainland China, Paraguay, Viet Nam and Taiwan POC.*
- *HS4412.34.26.00 – A total of 4 m^3 was imported into the USA from Canada in 2018.*
- *HS4412.94.31.11 – Trade was only reported in 2008 (ca. 14,000 m^3 imported from China) and 2012 (ca. 8,000 m^3 imported from Paraguay).*

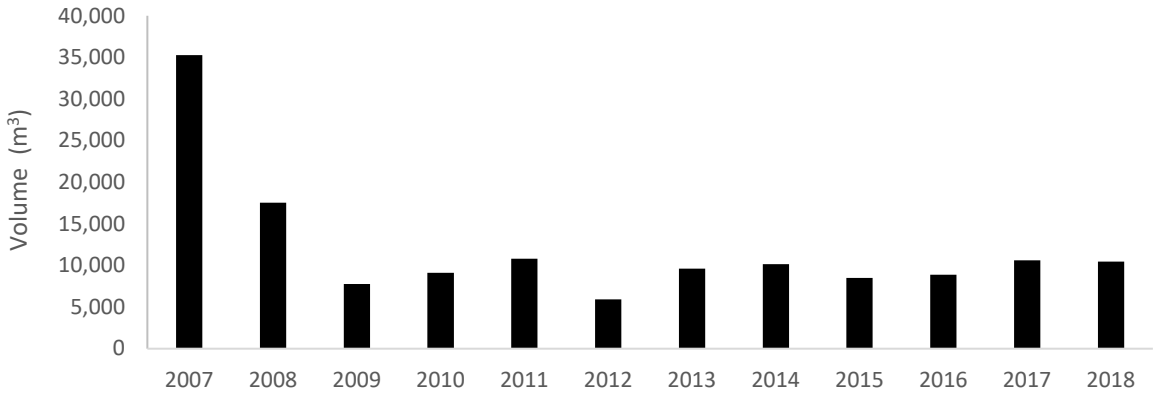

Figure 4 *USA imports of* Cedrela spp. *sawn/chipped wood (HS4407: Wood sawn or chipped lengthwise, sliced or peeled, whether or not planed, sanded or end-jointed, of a thickness exceeding 6 mm) between 2007 – 2018 (Data recorded to November 2018) (USTIC DataWeb, 2019).*

Table 1 *Top exporters of Cedrela spp. sawn/chipped wood (HS4407: Wood sawn or chipped lengthwise, sliced or peeled, whether or not planed, sanded or end-jointed, of a thickness exceeding 6 mm) to the USA between 2007 – 2018 (Data recorded to November 2018) (UST IC DataWeb, 2019).*

Importer	*Volume (m^3)*
Peru	*29,841*
Côte d'Ivoire	*25,795*
Ghana	*22,396*
Bolivia	*21,916*
China	*14,773*
Congo (ROC)	*9,400*
Brazil	*6,800*
Guatemala	*4,238*
Cameroon	*3,250*
Others combined	*6,254*

A total of 236.8 m^3 of *Cedrela* spp. wood was seized in Ecuador from 2014-2017 (domestically rather than at border points).

Other species

C. balansae*: Much of its habitat is heavily disturbed. In Paraguay, the timber is valued for furniture making and the species is planted in urban areas (Pennington & Muellner, 2010).*

C. discolor: *In 2010 the species was reported to be known from only a single type specimen collected in 1906; the type locality was not reported to be within a protected area and there were no protected areas in the vicinity (Pennington & Muellner, 2010).*

C. dugesii: *Habitat threatened by urban expansion in central Mexico, where the species is endemic. Used locally for firewood (Pennington & Muellner, 2010).*

C. kuelapensis: *Reported in 2010 that the entire extent of its range in Peru had been subject to deforestation by subsistence farmers and very little pristine forest remained; the population in Peru was estimated to have declined by at least 80% and the total area of occupancy was estimated at 48 km^2, and the species was not reported to be present in any protected area (Pennington & Muellner, 2010).* Assessed nationally as endangered in Ecuador (where it has lost 54% of its habitat).

C. lilloi: Assessed nationally as endangered in Ecuador (where it has lost 56% of its habitat) and Peru. *The species was reported to be widely cultivated including outside its natural range due to the value of its timber, and also frequently grown as an ornamental tree in urban areas. The timber was formerly widely used for construction of doors, windows and furniture, but large trees are now scarce and only found in reserved and inaccessible areas (Pennington & Muellner, 2010). Logging has been reported as a threat (Llamozas, 1998). The Bolivian and Brazilian populations were included in Appendix III in 2010 and 2016 respectively, and were included in Annex D of the EU Wildlife Trade Regulations in 2008 (the Bolivian and Brazilian populations were transferred to Annex C in 2012 and 2016, respectively). Although listed in Appendix III by Bolivia and Brazil since 2010 and 2016 no trade in* C. lilloi *has been recorded in the CITES Trade Database.*

C. molinensis*: Mature trees were reported to be steadily being cut down for fuelwood and timber, which is used locally for making furniture, and much of the species range was unprotected and threatened by grazing which prevents regeneration (Pennington & Muellner, 2010).*

C. monroensis: *Reported in 2010 that only a small percentage of the original population was likely to remain and the range continued to reduce due to habitat loss (Pennington & Muellner, 2010).*

C. montana: Assessed nationally as endangered in Ecuador (where it has lost 52% of its habitat) and vulnerable in Peru. *Reported in 2010 that the timber was widely used for furniture, and the species was often protected in pasture for its timber and as a shade tree. Some populations on Andean slopes were protected due to the inaccessible nature of the habitat (Pennington & Muellner, 2010). Listed in Annex D of the EU Wildlife Trade Regulations since 2008, but no trade is reported in the CITES Trade Database.*

C. nebulosa: *The timber is used locally in construction, and there were reported to be small plantations in northern Peru (Pennington & Muellner, 2010)*. Assessed nationally as endangered in Ecuador (where it has lost

31% of its habitat). *However, Pennington and Muellner (2010) reported that there were vast areas of undisturbed habitat within its range.*

C. oaxacensis: *Habitat reported to be threatened by urban expansion in southern Mexico, where the species is endemic (Pennington & Muellner, 2010). Listed in Annex D of the EU Wildlife Trade Regulations since 2008, but no trade is reported in the CITES Trade Database.*

C. saltensis: *Valued locally for its timber (Pennington & Muellner, 2010).*

C. salvadorensis: *Slash and burn reported to cause a continuous decline in suitable habitat. The timber is used locally in Mexico (Pennington & Muellner, 2010). Threatened with extinction in El Salvador (Anon., 1997). Listed in Annex D of the EU Wildlife Trade Regulations since 2008, but no trade is reported in the CITES Trade Database.*

C. tonduzii: *The timber is valued for construction, panelling and furniture, though it is considered to be inferior to C. odorata. Its habitat was reported to be heavily disturbed for coffee cultivation and cattle ranching. However, because of the value of its timber the species was often found protected in pasture after the forest had been felled (Pennington & Muellner, 2010). Threatened with extinction in El Salvador (Anon., 1997). Listed in Annex D of the EU Wildlife Trade Regulations since 2008, but no trade is reported in the CITES Trade Database.*

C. weberbaueri: *Reported in 2010 to be seriously threatened due to its limited range (estimated extent of occurrence of 1700 km^2 in the Mantaro Valley in Central Peru) and forest clearance; not reported to be present in any protected area (Pennington & Muellner, 2010).*

Inclusion in Appendix II to improve control of other listed species (Res. Conf. 9.24 (Rev. CoP17) Annex 2b)

A) Specimens in trade resemble those of species listed in Appendix II under Res. Conf. 9.24 (Rev. CoP17) Annex 2a or listed in Appendix I

Brazil reported that mahogany wood (three species of Swietenia are listed in Appendix II) is similar to that of Cedrela *but that they have technical material for the identification of mahogany and have the capacity to apply an identification methodology "using near infrared"; no further detail provided (PC24 Doc. 22 (Rev. 1)). It has been reported that a differentiation between* Cedrela *and* Swietenia *timbers should be possible in practice based on distinctive macroscopic and microscopic characters recorded in the CITESwoodID database (Richter* et al.*, 2014 and subsequent updates), and from experience of training courses for enforcement personnel (G. Koch,* in litt.*, 2019).*

Additional Information

Threats

The main threat to C. odorata *is unsustainable harvest of timber; deforestation and the associated habitat loss also threaten the species (Mark & Rivers 2017). See above for details.*

Conservation, management and legislation

- Brazil, Colombia, Guatemala and Peru have carried out population studies of the *Cedrela* species, in order to understand the current status of these species and adopt measures to promote their forest management. *Peru also has a very detailed national forest inventory where management units have been defined, and carried out a population study of the genus* Cedrela *in its natural range from April 2008 to March 2009 in order to evaluate the commercial stocks and consider a sustainable management strategy (PC24 Doc. 22 (Rev. 1)).*
- Mexico, Guyana and Cuba have estimated coverage and density of populations.
- Argentina, Brazil, Colombia, Jamaica, Mexico and Peru require verified management plans to regulate harvest of *Cedrela* species.
- In Colombia, some Regional Autonomous Corporations have prohibited use of *C. odorata*.
- *There are broad bans/restrictions on timber logging/export in several range States (only the three top exporters of* C. odorata *mentioned here): In Brazil, the export of unfinished timber of native species (i.e. destined to be processed abroad) is prohibited according to Normative Instruction 15/2011 (IBAMA, 2011), amended by Normative Instruction 13/ 2018 (IBAMA, 2018). In Bolivia, export of unprocessed forestry products is subject to restrictions and highly regulated, mainly through forest certification (from 1996 onwards, last updated 2016; Forest Legality Initiative, 2016). In Peru, there is an export ban on logs and forest products "in their natural state" except when they originate from nurseries or forest plantations, and if they do not require processing for final consumption (from 1972 onwards, last updated 2016; Forest Legality Initiative, 2016).*

Occurrence of *C. odorata* in protected areas: Tikal National Park in Guatemala (57,400 ha), Corcovado NP (54,568 ha) and La Amistad Biosphere Reserve (584,592 ha) in Costa Rica; elsewhere in South America it is present in many reserved areas throughout its range (Pennington & Muellner, 2010).

Artificial Propagation

Cedrela odorata plantations in tropical America have not been successful due mainly to attack by the Shoot Borer *Hypsipyla grandella. However, planting at low densities in a matrix of other species appears to confer some protection (Pennington & Muellner, 2010).* For this reason, it is sown mainly within agroforestry plantations in its natural distribution area. *In areas without Shoot Borer (Africa, Asia and the Pacific) it establishes well in plantations and even has the potential to become an invasive weed, forming dense monospecific stands which can shade out less aggressive native species (Pennington & Muellner, 2010).* Poor selection of plantation sites is also cited as a factor in the lack of success of plantations, by not meeting with the specific requirements of the species, such as having well-drained, deep and humid soils.

As detailed above, around 25% of global exports reported over the period 2007-2016 were recorded as artificially propagated, the proportion of artificially propagated exports increased over this period from <1% in 2007 to 59% in 2016.

In the period 2010–2018, Ecuador harvested a volume of standing timber of 3,911.94 m^3, of which 86% came from cultivated forests (plantations) and the remaining 14% from relict tree cutting programs. In Mexico there has been an increase in commercial forest plantations. *Mexico reported 37,176 ha of plantations with a yield of 160 m^3/ha, with its commercial purpose being roundwood, while Colombia reported that it has registered 8,021 ha of plantations of this species (PC24 Doc. 22 (Rev. 1)). A 2011 report noted that plantations in Bolivia, Brazil and Peru were not yet of harvestable age (Pérez Contreras, 2011). It is not clear whether there is an ongoing reliance on wild seeds/saplings to establish or supplement plantations.*

Implementation challenges (including similar species)

Taxonomic, genetic, physical, mechanical and anatomical knowledge to differentiate the woods of *Cedrela* species is limited, and the woods of different species may be marketed interchangeably. Several countries have expressed difficulties in differentiating between *Cedrela* species (Brazil, Ecuador, El Salvador and Peru). Guatemala, Jamaica and Peru reported that they have developed identification tools to differentiate *Cedrela* species; the only further details provided were that Peru has developed manuals for the dendrological and anatomical identification of the main commercial *Cedrela* species which will be published online in the near future. *However, based on the CITESwoodID database (Richter* et al., *2014 and subsequent updates), it is not reported to be possible to distinguish the individual species within the genus* Cedrela *using macroscopic or microscopic wood anatomical methods (G. Koch,* in litt., *2019). Furthermore, the timbers of* Cedrela *and Toona (the latter naturally distributed in Asia and not listed in the CITES Appendices) are virtually indistinguishable in practice given their similiar colour, structure and odour, hence an annotation to restrict the listing to the native range of* Cedrela *(Americas) could be considered (G. Koch,* in litt., *2019).*

Potential risk(s) of a listing

In the case of Brazil, the prohibition of trade in mahogany coincided with substantial export increases of "other tropical species", including cedar, with signs of illegal trade (Pérez Contreras, 2011).

Potential benefit(s) of listing for trade regulation

The commitment assumed by Peru from 2001 onwards to transparently provide up-to-date information on export permits for cedar and mahogany to current and potential consumers has contributed to improve the image and value of Peruvian cedar sawn wood, of which the FOB price is 45% higher than that of Brazil and 40% higher than that of Bolivia (and a similar difference for mahogany) (Pérez Contreras, 2011).

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Front cover photo:
Rhino-horned Lizard *Ceratophora stoddartii*
endemic to Sri Lanka by Shanaka Aravinda.

AUGUST 2019

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