

IUCN AND TRAFFIC **ANALYSES**

of the proposals to amend the CITES Appendices at the
19TH MEETING OF THE CONFERENCE OF THE PARTIES

14 - 25 NOVEMBER 2022



TRAFFIC

IUCN/TRAFFIC analyses of the proposals to amend the CITES Appendices at the

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Panama City, Panama

14th - 25th November 2022

Prepared by IUCN Biodiversity Assessment and Knowledge Team and Species Survival Commission and TRAFFIC



TRAFFIC



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The IUCN Species Survival Commission (SSC), the largest of IUCN's six commissions, has over 10,500 species experts recruited through its network of 168 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 over 170 staff working on five continents 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, towards the shared goal of reducing the pressure of unsustainable trade on wild species.

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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 IUCN and TRAFFIC now produce the Analyses of the Proposals for CITES CoP19, which will take place in Panama City, Panama, in 2022. We would like to thank the team in TRAFFIC and IUCN for producing such a complex and helpful document in a very short time.

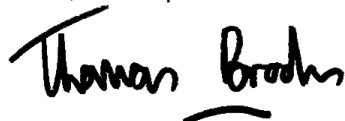
Information on the status and biology of species was collected from IUCN's Species Survival Commission Specialist Group network and the broader scientific community, and used to evaluate the proposals and the information provided by proponents against the CITES listing criteria. TRAFFIC has drawn on 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.

The Analyses for CoP19 will not only provide an assessment of whether or not each proposal meets the criteria specified by CITES, but also summarises any additional considerations that maybe relevant to the decision on whether or not to adopt the proposal. These include, for example, any potential implementation challenges, benefits or risks that maybe 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 18,373 species assessed as threatened on The IUCN Red List of Threatened Species) underscored by the recent release of the IPBES Assessment of Sustainable Use of Wild 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 184 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 Nineteenth Meeting of the Conference of the Parties to CITES (CoP19).

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.iucnredlist.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, an additional paragraph is included for some 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 or benefits for the conservation of the species concerned). Information used to compile these sections 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. See the Methods section for more information.

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. They are produced with a limited budget under severe time constraints and are not exhaustive. Where proposals include many different taxa it is not possible to address them all in detail and there will invariably be omissions and differences of interpretation. 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 proposals.

The Analyses were completed and made available online on 9th September 2022 to allow CITES Parties and other stakeholders sufficient time to consider the information in advance of the Conference of the Parties, which convenes for the nineteenth time on 14 November 2022 in Panama. 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 CoP19.

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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.

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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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METHODS

Criteria

The taxa included in the proposals were assessed according to the criteria in Res. Conf. 9.24 (Rev. CoP17). All relevant criteria were addressed regardless of those explicitly cited in the proposal. Where multi-taxon proposals included many taxa with a large number of these proposed as lookalikes we focused on those identified as traded in significant volumes were assessed against all relevant criteria.

For proposals regarding commercially exploited aquatic species, the general guidelines listed in the footnote to Annex 5 of Res. Conf. 9.24 (Rev. CoP17) with respect to the application of the definition of “decline” were taken into account when species were assessed against criterion 2aA for inclusion of species in Appendix II (refer to doc. AC25 Inf.10 for a detailed explanation of TRAFFIC’s application of listing criteria to commercially exploited aquatic species).

Information sources

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, peer reviewed and grey literature, news reports and experts in the broader scientific community. TRAFFIC has drawn on its own expert network and information sources to determine the nature and scale of any trade. Where possible, currencies were converted to USD with the average exchange rates of August 2022 sourced from xe.com unless otherwise specified.

Various databases were consulted to assess reported legal and illegal trade for all taxa proposed for listing. These include the CITES Trade Database, the EU-TWIX database, the TRAFFIC Wildlife Trade Information System (WiTIS), the United States Fish and Wildlife Service’s (USFWS) Law Enforcement Management Information System (LEMIS), ITTO Biennial Reports, and UN Comtrade.

CITES Trade Database

CITES trade data were downloaded on the 27th of July 2022, covering at least the last decade of reported trade for the taxa of interest. The last most complete year for which data exist for the majority Parties is 2020, however, some Parties have reported trade for 2021 prior to the 31st of October 2022 deadline (Res. Conf. 11.17 [Rev. CoP18]). Data were downloaded in Comparative Tabulation format. In some cases, the terms “whole organism equivalents” and “parts and derivatives” are referred to in the *Analyses*. Whole organism equivalents are an aggregation of term codes that can be reasonably equated to a whole individual. These include the term codes for bodies, live, skeletons, skins, and skulls (codes BOD, LIV, SKE, SKI, and SKU) where they are reported by number (see CITES Notif. 2021-044-A1 for more information on term codes). Parts and derivatives are the remaining term codes reported by number as well as other units. Sources are sometimes combined for brevity as “wild sources” and “captive sources”. Wild sources include source codes for wild, ranched, and unknown or unreported (codes “W”, “R”, “U” and unreported), and captive sources include captive-born and captive-bred or artificially propagated (codes “C”, “F”, “A” and “D”). We reported only direct imports and exports when summarising total volumes of commodities in trade to avoid duplication of records, unless otherwise specified.

LEMIS Database

LEMIS data used in the *Analyses* included trade reports from 2008 to 2020, reported by the USA. Data are based on USFWS Form 3-177 (Rev. 10/2017) that all wildlife importers and exporters must declare with the US Fish and Wildlife Service; this includes information on trade in CITES and non-CITES listed species. These data include records for imports, exports, and transits, as well as information regarding the outcome of the reported trade. This can be cleared, i.e. permitted trade, or refused, i.e. not permitted for trade. Records referred to as “cleared” in the *Analyses* refer to records that have been cleared under the Action as well as the final Disposition. Some records that are refused may subsequently be seized, abandoned, re-exported, or cleared. The reason for refusal cannot be inferred from the data. Sources are sometimes combined for brevity as “wild sources” and “captive sources”.

Wild sources include source codes for wild, ranched, and unknown or unreported (codes “W”, “R”, “U” and unreported), and captive sources include captive-born and captive-bred or artificially propagated (codes “C”, “F”, “A” and “D”).

LEMIS data was obtained through a Freedom of Information Act request to the US Government for data in the period 2008–2020, which included taxonomic data, description of the commodity, dates, action and disposition, purpose, source, quantity and unit, the type of trade (import/export/transit), country of origin, and the trading partner.

EU-TWIX

EU-TWIX is the Europe Trade in Wildlife Information Exchange database, managed by TRAFFIC on behalf of the participating countries. The EU-TWIX system provides users with a mailing list which facilitates the rapid exchange of information, expertise, and experience on wildlife trade enforcement. EU-TWIX also includes access to a dedicated website containing useful resources to support enforcement efforts as well as a database of wildlife seizures. The EU-TWIX database contains centralised data on seizures and offences reported by all 27 EU Member States, Bosnia and Herzegovina, Iceland, Switzerland, Ukraine, and the UK. Access to this database is restricted to wildlife law enforcement and management authorities in Europe. Data for use in the Analyses were downloaded with prior authorisation from the relevant countries on 27th April 2022, and included seizures made in the years 2011-2020.

TRAFFIC Databases

The Wildlife Trade Information System (WiTIS) is a database managed by TRAFFIC containing information on incidents involving wildlife seizures, poaching, and law enforcement actions, in addition to market monitoring and actionable information. This information is sourced predominantly from open-source media reports as well as some CITES Management Authorities, government agencies, customs organisations, social media platforms and NGOs. The whole database was consulted for the *Analyses*, and data are available from records created between 2010 to 2022. Non-confidential records held in this database are publicly available via the TRAFFIC Wildlife Trade Portal. Due to the nature of this database and illegal trade, it should not be inferred that there is a direct correlation between WiTIS incident data and the overall illegal wildlife trade, or that information across locations, species or time is consistent in the database.

In addition to the databases described above, a rapid search of online advertisements was conducted by TRAFFIC in August 2022 for some taxa through an online investigation software and intelligence platform. Common names, trade names, and scientific names as well as commonly used synonyms were used to search for online advertisements on publicly accessible fora. Results were catalogued in a database.

The references for these data sources are as follows:

CITES (2022). CITES Trade Database, UNEP World Conservation Monitoring Centre, Cambridge, UK.

EU-TWIX (2022). *Trade in Wildlife Information Exchange*.

LEMIS (2022). United States Fish and Wildlife Service Law Enforcement Management Information System.

TRAFFIC (2022a). Survey of online sales [electronic dataset], viewed 18 August 2022.

TRAFFIC (2022b). *Wildlife Trade Information System* [database].

Transfer of Common Hippopotamus *Hippopotamus amphibius* from Appendix II to Appendix I

Proponents: Benin, Burkina Faso, Central African Republic, Gabon, Guinea, Liberia, Mali, Niger, Senegal, Togo

Summary: The Common Hippopotamus or hippo *Hippopotamus amphibius* is a large semi-aquatic African mammal. It is widespread throughout sub-Saharan Africa, predominantly across southern and eastern Africa, with smaller, isolated populations in central and western Africa. In Africa it is known to be extant in 38 range States and is extinct in five.

Hippos require fresh water with areas shallow enough for them to stand and be completely submerged and large enough to contain the territories of several males. They live near rivers, lakes and wetlands in forest, savanna and shrubland areas where there are suitable open grasslands for grazing. Habitat selection is highly dependent on season and water availability. Hippos are gregarious and social in water, gathering in large herds comprising hundreds of individuals. They have a generation length of 10 years and produce only one offspring every two years under optimal conditions. Hippo densities are highly variable and dependent on local environmental factors, so accurately estimating the size of populations can be difficult.

The 2016 IUCN Red List assessment estimated a population of approximately 115,000–130,000 hippos. The greatest numbers were reported in southern Africa (60,000) with a stronghold in Zambia, and eastern Africa (50,000) with a stronghold in Tanzania. The western African population totals an estimated 7,500. According to the 2016 IUCN Red List assessment, country-level trends for *H. amphibius* populations indicated that they were decreasing in 16 range States (42%), stable in nine (24%), unknown in nine (24%) and increasing in four (11%). Based on evidence of declines of ≥30% over three generations (30 years, 1986–2016), the IUCN Red List assessed *H. amphibius* as Vulnerable in 2016. The Red List assessment reported that past overestimations in population data have made accurately tracking long-term trends difficult, and also have important implications for management decisions.

While recent population census data indicate trends continue to vary across range States, increases have been reported within range States comprising some of the largest hippo populations, such as Botswana, and particularly within protected areas. Botswana's total hippo population, estimated at 2,000–4,000 in 2016, was estimated at 11,231–15,233 in 2018 so that the country now contains one of the largest hippo populations in Africa. Recent population census estimates have also been reported in Tanzania (20,000 in 2016 to 26,152–36,020 in 2018) and South Africa (7,000 in 2016 to 11,061 in 2018).

Major threats to hippo populations identified are habitat loss resulting from water diversion for agricultural, energy, mining, and residential development, as well as habitat degradation from water pollution. Competition for habitat and water, and activities such as fishing also lead to the threat of human–hippo conflict. Climate change such as extended periods of drought also poses a significant threat to hippos. The 2016 IUCN Red List assessment identified illegal and unregulated hunting for meat and ivory as a primary threat; however, the IUCN/SSC Hippo Specialist Group (IUCN SSC HSG) has not identified ivory poaching as a current driver of hippo declines.

Conservation management and State protection strategies for hippos vary across Africa, with regulation and enforcement remaining weak in many countries. Hippos are completely protected from hunting for commercial or other purposes in 14 range States (Angola, Burkina Faso, Cameroon, Central African Republic, Congo, Gabon, Ghana, Guinea Bissau, Kenya, Niger, Nigeria, Rwanda, Senegal, and Somalia). In most other range States, hippos are partially protected, with hunting for commercial or other purposes allowed with a permit. Culling has been used as a management strategy in some range States as a response to drought (South Africa) or for problem animal control

(Malawi).

Hippopotamus amphibius has been included in CITES Appendix II since 1995. *H. amphibius* products in trade predominantly comprise ivory (teeth and tusks), trophies and skins from wild-sourced specimens. The largest exporters include Zimbabwe, Uganda, South Africa, and Zambia. Hippos have been the subject of a CITES Review of Significant Trade (RST) process twice, firstly in 1999 and again in 2008. The RST resulted in Tanzania establishing an export quota of 10,598 teeth from 1,200 animals and hunting trophies. Illegal trade has been reported but is not considered to be a serious threat.

Analysis: The Common Hippopotamus does not have a restricted range, nor does it have a small population. Estimates of population trends vary across African range States, with some *H. amphibius* populations stable or increasing, and others decreasing. At the species level, the global hippo population was reported to have declined by $\geq 30\%$ (but less than 50%) over three generations (30 years, 1986–2016). This is less than the guideline figure given in Res. Conf. 9.24 (Rev. CoP17) for a marked recent rate of decline of 50% or more over 10 years or three generations, whichever is the longer. Furthermore, the rate of decrease is likely to be slowing because stable or increasing *H. amphibius* populations, mainly in southern and eastern Africa, make up a large proportion of the overall population. *H. amphibius* would not therefore appear to meet the biological criteria for inclusion in Appendix I.

Hippo products, mainly teeth and tusks, are in legal and illegal trade, predominantly from Uganda and Tanzania; this trade is not considered a significant threat to the species as this trade has remained stable or declined over the last ten years. Previous concerns regarding implementation of the Appendix II listing have been addressed through the CITES Review of Significant Trade (RST) process, resulting in three range States establishing export quotas. Any future concerns regarding export levels of hippo products could also be addressed through the RST process.

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 *Hippopotamus amphibius* as a single species. There are two extant members of the Hippopotamidae family, *H. amphibius* commonly referred to as Common Hippopotamus or hippo, and the Pygmy Hippopotamus recognised as *Hexaprotodon liberiensis* in Wilson and Reeder (2005) and *Choeropsis liberiensis* in the IUCN Red List (Ransom et al., 2015). Morphological research indicated that five subspecies may exist (Lydekker, 1915). However, more recent genetic studies only support the existence of putative *H. amphibius* subpopulations (Stoffel et al., 2015) and further research is required to validate this.

Range

Native: Angola, Benin, Botswana, Burkina Faso, Burundi, Cameroon, Central African Republic, Chad, Congo, Côte d'Ivoire, Democratic Republic of the Congo, Equatorial Guinea, Ethiopia, Gabon, Gambia, Ghana, Guinea, Guinea Bissau, Kenya, Malawi, Mali, Mozambique, Namibia, Niger, Nigeria, Rwanda, Senegal, Sierra Leone, Somalia, South Africa, South Sudan, Sudan, Swaziland, Tanzania, Togo, Uganda, Zambia, and Zimbabwe.

Regionally extinct: Algeria, Egypt, Eritrea, Liberia, and Mauritania.

Introduced: Colombia (Castelblanco-Martínez et al., 2021).

IUCN Global Category

Vulnerable (assessed 2016, ver. 3.1)

Biological criteria for inclusion in Appendix I

A) Small wild population

Hippopotamus amphibius was assessed in 2016 as Vulnerable on the IUCN Red List under criterion A4acd due to a population decline $\geq 30\%$ over three generations (30 years, 1986–2016, Lewison and Pluháček, 2017).

The best available estimates indicate a total hippo population of between 115,000 and 130,000 individuals in 2016 (Lewison and Pluháček, 2017). There is regional variation in the size of *H. amphibius* populations, with the largest numbers of hippos occurring in southern Africa (60,000) and eastern Africa (50,000). In western Africa,

hippos occur in fragmented, isolated populations at lower densities with the total population estimated as 7,500 individuals.

A lack of recent national hippo census data and a general lack of population data throughout their range make it hard to assess current population numbers and trends. This is illustrated by the hippo population estimate of 20,000 individuals in Tanzania, which is based on a 2001 census. According to the 2016 census data, over half of the hippo population occurred in three range States: Zambia (35%; 40,000–45,000), Tanzania (15–17%; 20,000) and Uganda (6–8%; 7,000–10,000). However, updated census data indicate Botswana may contain one of the largest populations in Africa (13,232).

Zambia: Hippos were reported to be widespread, locally abundant with a stable population estimated to total 40,000–45,000 individuals in 2016 (Table 1). *A long-term hippo population census study along a 165 km section of the Luangwa valley, 1976–2015, reported a carrying capacity of 6,000 individuals, with the highest population 7,862 individuals in 2015, and ranged between 5,000 and 8,000 individuals (Chomba et al., 2021). Chomba (2013) reported that the population size of hippos on the Luangwa River was regulated by the amount of food (e.g., grass biomass) and that culling, trophy hunting, problem animal control, poaching and disease did not significantly affect either population size or density. A large-scale programme to cull more than 2,000 hippos over a five-year period in Eastern Province's Luangwa Valley, reportedly planned by the Zambesian government in 2016, and again in 2018, was cancelled on both occasions (Mwenda, 2019).*

Tanzania: *H. amphibius* were reported to be widespread and locally abundant with a stable population estimated to total 20,000 individuals in 2016. However, this assessment was based on a 2001 census. In 2018, an aerial survey of the Selous-Mikumi ecosystem in southern Tanzania estimated a total hippo population of 31,086 (estimated to range between 26,152 and 36,020 individuals; Tanzania Wildlife Research Institute (TAWIRI, 2019). This represented a significant increase from figures estimated during a 2014 survey (23,243 individuals \pm 5483; TAWIRI, 2019).

Botswana: An estimated total of between 2,000 and 4,000 hippos was reported in 2016, comprising 2–3% of the estimated global hippo population. However, these estimates were based on census studies first made in 1993 (Chase et al., 2018). Updated census data in 2018 estimated a total of 13,232 hippos in northern Botswana, comprising 10–12% of an updated total global population and meaning the country contains one of the largest hippo populations in Africa (Chase et al., 2018). The declining population trend reported in 2016 likely resulted from a period of extended drought and low flooding resulting in scarce food and water availability (Chase et al., 2018). The hippo population in northern Botswana between 2014 and 2018 increased significantly, particularly within the Okavango Delta (Chase et al., 2018).

Uganda: The hippo population was estimated at between 7000 and 10,000 individuals in 2016. The Red List assessment reported historical decreases in the hippo population across the country. The population of Queen Elizabeth National Park had once numbered 21,000 individuals and had reduced to 2,172 in 1989 due to poaching and numbered 5,000–6,000 individuals in 2016. Lake Edward and Lake George were reported to have 30,000 hippos in the 1960s but this has decreased to only around 6,000 in 2021. The hippo population of Murchison Falls National Park was reported to have decreased from 1,683 in 2016 to 590 in 2021.

Zimbabwe: The hippo population was estimated at 7,000 individuals in 2016. Utete (2020) reported that the 2016 assessment estimates of hippo populations were based on studies carried out in protected areas which may have led to bias in the national inferences. *Available hippo census data from national parks indicate hippo populations are now recovering after experiencing severe historical decreases, with gradual increases reported in the Hwange National Park from 1972 to 2019 and in the Gonarezhou National Park from 1992 to 2012 (Utete, 2020).*

South Africa: Hippos were reported to be locally abundant but with a restricted distribution, estimated to total 7,000 individuals and have a stable population trend in 2016. According to 2013–2015 surveys hippo populations were estimated to total a minimum of 11,061 individuals, equating to around 6,378–7,743 mature individuals (Eksteen et al., 2016). The Kruger National Park hippo population was estimated to have increased from 2,510 in 1986 to 7,270 in 2015, a 190% increase over three generations (Eksteen et al., 2016). Populations in Mpumalanga increased by 78% and by 20–30% in Limpopo between 2003 and 2013 (Eksteen et al., 2016). In the KwaZulu Natal Province an annual 4% population increase between 2004 and 2011 was followed by a population decrease between 2011 and 2013, due in part to increased poaching (Eksteen et al., 2016).

Hippo populations in northern central Africa (i.e., Cameroon, the Central African Republic, and Chad) were reported to total around 2,500 individuals, mostly found in the Faro and Bénoué National Parks in northern Cameroon (Scholte et al., 2017). However, updated hippo population census data in Cameroon indicate populations are higher than previously estimated (see Table 1).

B) Restricted area of distribution

Hippos inhabit a wide range of countries throughout sub-Saharan Africa in suitable wetland habitats. Hippos still occupied a large proportion of their historical range in 1959 (see Table 1). However, hippo populations in western Africa are increasingly fragmented and isolated, and according to the 2016 IUCN Red List assessment hippos have restricted distributions in 33 range States (87%).

C) Decline in number of wild individuals

Population dynamics vary across range States with evidence of both recent decreases and increases, with increases particularly noted within protected areas (IUCN SSC HSG, in litt., 2022). Each of these populations is subject to threats, conservation management strategies and levels of utilisation that are specific to their range State or region. Hippo population estimates have also been difficult to assess accurately in the past due to a paucity and geographical bias in census data resulting in overestimates and repeated use of outdated data. On the basis of the 2016 IUCN Red List assessment, hippos were classified by IUCN as Vulnerable based on a projected population reduction of greater than 30% within three generations (30 years) as a result of the growing and unabated threats of habitat loss and unregulated hunting. According to the 2016 IUCN Red List assessment, country-level trends for *H. amphibius* populations indicated that they were decreasing in 16 range States (42%), stable in nine (24%), unknown in nine (24%) and increasing in four (11%).

Since the 2016 assessment, updated census data indicate Botswana's hippo population is estimated to total 11,231–15,233 individuals (Chase et al., 2018). Population estimates have also been updated in Tanzania (20,000 in 2016 to 26,152–36,020 in 2018; TAWIRI, 2019) and South Africa (7,000 in 2016 to 11,061 in 2018; Eksteen et al., 2016). Despite the large degree of uncertainty in hippo populations estimations, the increase in Kruger National Park, South Africa was reported to be genuine, as opposed to a previous underestimate (Eksteen et al., 2016). Recent hippo population increases within national parks in Zimbabwe highlight the species' resilience and adaptability, indicating that under favourable environmental conditions hippo populations are able to recover rapidly (Utete, 2020).

Trade criteria for inclusion in Appendix I**The species is or may be affected by trade****National trade**

Hippos are hunted for food, with hippo meat consumed in several African countries.

Legal trade

*Analysis of the CITES trade data shows that over the most recent ten years of complete *H. amphibius* trade, 2011–2020, a total of just under 9,000 whole organism equivalent (WOE, including bodies, live, skins, skulls, skeletons and trophies) wild-sourced individuals were directly exported for trophy hunting, personal and commercial purposes, as reported by exporters (7,220 as reported by importers; Table 1). The countries exporting the highest quantities of WOE trade were Zimbabwe (37%), South Africa (32%), and Zambia (12%), as reported by exporters. The countries importing the highest quantities of WOE trade were the United States of America (45%), Mexico (13%), and South Africa (13%).*

*Trade in *H. amphibius* commodities, other than WOE, 2011–2020, predominantly comprised teeth and tusks (combined to represent trade in hippo ivory) and skin pieces. *H. amphibius* ivory trade by number comprised 25,845 items reported by exporters (11,887 reported by importers), largely exported by Tanzania (43%), Zimbabwe (22%), and South Africa (14%). Countries/territories importing the highest quantities of *H. amphibius* ivory by number, 2011–2020, were Hong Kong, SAR (33%) and the United States of America (28%).*

**H. amphibius* ivory trade by weight, 2011–2020, totalled over 14,000 kg, and was largely exported by Uganda (77%) and to a lesser degree Malawi (11%), as reported by exporters. Importers reported equivalent trade totalling over 23,000 kg. Hong Kong, SAR, imported the highest quantity of *H. amphibius* ivory trade by weight (86%) with mainland China importing (13%).*

**H. amphibius* range States that permit trophy hunting and the export of ivory derived from government stockpiles (e.g., natural deaths, problem animal controls, and/or confiscations) include Ethiopia, Mozambique, South Africa, Tanzania, Zambia, and Zimbabwe (Moneron and Drinkwater, 2021).*

A 2021 report assessing international trade in hippo ivory noted that quantities between 2009 and 2018 equated to an annual offtake of approximately 1,349 individuals (Moneron and Drinkwater, 2021). These offtake estimations equated to around 1% of the total population and ivory exports were from countries with relatively high hippo populations (Moneron and Drinkwater, 2021). Hippo population models have revealed that offtake levels of 1% can lead to population declines over 30–40 years (Lewison, 2007; Lewison and Pluháček, 2017). However, it has been suggested that offtake levels of less than 4% can be sustainable as long as specific sex or age categories are not targeted (IUCN SSC HSG, in litt., 2022).

Illegal trade

Illegal trade in hippo parts and products, particularly teeth, is evident and was *recently reported across 48 countries/territories (Moneron and Drinkwater, 2020)*. The legal international trade in hippo parts and derivatives was reported to have a detrimental impact on hippos by providing a way for illicit commodities acquired from poached hippos, primarily ivory, to enter the legal market. Numerous hippo teeth seizures and associated arrests have been reported since 2016. Tanzania and Uganda are origin countries for the majority of hippo ivory trade between 2011 and 2020 and are also countries with the largest quantities of illegally traded ivory. *Illegal trade of hippo ivory is likely to be placing additional pressures on hippo populations with the inadequate reporting of trade undermining conservation efforts (Moneron and Drinkwater, 2020)*.

WiTIS contained 174 incidents of seizure events with 129 (74%) in WOE and ivory commodities. Law enforcement authorities seized around 2,500 kg and just under 2,000 items of hippo ivory, and 25 WOE specimens between 2011 and 2022 (Table 2). Hippo teeth were the most commonly seized trade commodity, with worked ivory pieces and tusks also seized, but in smaller quantities. Two incidents related to a total of 10 wild-sourced hippos illegally hunted in Tanzania in 2017.

Additional information

Threats

The main threats to hippos are habitat loss and degradation (e.g., water pollution), hippo–human conflict (e.g., agriculture, energy, fishing, mining and residential development) and climate change (e.g., droughts; IUCN SSC HSG, in litt., 2022). The 2016 IUCN Red List assessment identified illegal and unregulated hunting for meat and ivory to be a primary threat to hippos (Lewison and Pluháček, 2017). *However, the IUCN SSC HSG (in litt., 2022) stated that currently ivory poaching does not seem to be the main driver of hippo population reductions. Furthermore, international trade in hippo ivory and products has either declined or remained stable over the last ten years (IUCN SSC HSG in litt., 2022)*.

Hippo habitat loss is caused by a range of factors including competition with humans for freshwater resources whereby freshwater is being diverted for agricultural and human development. In western and central African range States habitat loss has increased hippo population fragmentation resulting in small, isolated populations restricted to protected areas.

H. amphibius is targeted by illegal hunters in many range States. Illegal hunting is most prevalent during periods of civil unrest and can cause dramatic population decreases. Civil unrest has impacted hippo populations in south-eastern Central African Republic, Cote d'Ivoire, Virunga National Park, Garamba National Park and wildlife reserves in north-eastern Democratic Republic of the Congo and the Dinder National Park in Sudan. During civil unrest, militia target hippos to provide meat for human consumption and ivory as a trade commodity for financial resources.

Hippos are threatened by climate change and particularly susceptible to periods of drought. The hippo population in the Gonarezhou National Park in Zimbabwe fell by 95% due to a severe drought occurring between 1984 and 1992.

Disease (including anthrax), is considered to be an additional threat to some hippo populations (Driciru et al., 2018). Mass outbreaks of anthrax occur periodically within hippo populations in Zambia and Uganda (Dudley et al., 2016). Across several range States hippo threats are greatest outside protected areas, highlighting the importance of these areas for hippo conservation (IUCN SSC HSG, in litt., 2022).

Conservation, management and legislation

Conservation management strategies and the level of State protection for hippos varies across Africa. Hippos are officially protected in many range States, however the enforcement of regulations remains weak in many countries. Hippos are completely protected from commercial hunting or other purposes in 14 range States including Angola, Burkina Faso, Cameroon, Central African Republic, Congo, Gabon, Ghana, Guinea Bissau, Kenya, Niger, Nigeria, Rwanda, Senegal, and Somalia. In most other range States the hippo is partially protected with hunting for commercial or other purposes only allowed with a permit, except Equatorial Guinea where, apparently, there are no specific regulations concerning the removal of hippos from the wild.

H. amphibius was listed in CITES Appendix III by Ghana in 1976 and subsequently transferred to Appendix II in 1995. *H. amphibius* has been included in the Review of Significant Trade process in 1999 and 2008. This led to Tanzania, Cameroon, and Mozambique establishing export quotas. CITES quotas for 2022 are in place for three countries (Cameroon 10 trophies, Ethiopia six hunting trophies from wild specimens, 20 kg of raw ivory and 20 kg of worked ivory and Tanzania 10,598 teeth from 1,200 trophy animals recommended by the Animals Committee or Standing Committee). Mozambique has a 2021 CITES quota of 49 (no further details provided). CITES suspensions are currently in place for four countries, with all commercial trade in specimens of CITES-listed

species suspended from Guinea (exceptions not related to *H. amphibius*), Liberia, and Somalia, and all CITES-listed species trade suspended from Djibouti.

Hippo culls have been proposed and carried out in several range States. In Zambia, the government planned a large-scale programme to cull more than 2,000 hippos over a five-year period in Eastern Province's Luangwa Valley in 2016 (Mwenda, 2019). The 2016 cull was cancelled, reinstated in 2018, and cancelled again (Mwenda, 2019). In 2016, hippo culling was reinstated in Kruger National Park (KNP), South Africa, where a population of over 7,000 individuals had been recorded (Eksteen et al., 2016). Culling was managed adaptively due to drought conditions across South Africa between late 2015 and early 2016 and attributed to a lack of food availability in KNP (Eksteen et al., 2016). In Malawi, a number of hippos are reportedly culled each year by the department of national parks and wildlife due to crop damage (20–30 cases each year).

Captive breeding

Hippos breed very well in captivity in zoological institutions. Breeding restrictions for hippos have been applied on all continents as zoos have no additional space for potential offspring. No known commercial breeding operations exist.

Implementation challenges (including similar species)

The Pygmy Hippopotamus *Choeropsis liberiensis* is similar in overall appearance to the hippo but much smaller in body size. Pygmy Hippos are also reclusive animals that share only a small area of overlapping range with the hippo in West Africa. While teeth of *H. amphibius* are typically larger and found more frequently in trade than those of Pygmy Hippos, the teeth of both appear very similar. *However, according to the IUCN SSC HSG (in litt., 2022), Pygmy Hippos are not hunted for ivory.* Hippo teeth and ivory can be easily distinguished from the ivory of other species, such as African Elephant (*Loxodonta africana*), Asian Elephant (*Elephas maximus*), Mammoth (*Mammuthus* spp.), Walrus (*Odobenus* spp.), Killer Whale (*Orcinus orca*), Sperm Whale (*Physeter macrocephalus*), Narwhal (*Monodon monoceros*), and Warthog (*Phacochoerus* spp.).

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

*Legal hippo trophy hunting in certain range States provides financial incentives for sustainably harvesting hippos and provides conservation benefits. The transfer of *H. amphibius* to CITES Appendix I may impact hippo trophy hunting and, in turn, may negatively affect hippo conservation in areas where trophy hunting provides conservation benefits (IUCN SSC HSG, in litt., 2022).*

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Table 1. Overview of the Hippopotamus amphibius country-level population status and trends and direct wild-sourced Hippopotamus amphibius trade from exporter countries, 2011–2020, for commercial, hunting and personal purposes.

| Range State | Status | Trend | Estimated population size (2016) | Updated population size estimate (date) | Whole organism equivalent trade (WOE; i.e., bodies, live, skins, skulls and trophies) by number as reported by exporters | Hippo ivory (teeth and tusks) as reported by exporters. Equivalent number of animals given in parentheses. * | Hippo ivory (teeth and tusks) by weight in kilogrammes as reported by exporters. Equivalent number of animals given in parentheses. ** | Total number of hippos based on ivory trade (teeth and tusk) by number and weight converted to number as reported by exporters |
|----------------------------------|--------|-------|----------------------------------|---|--|--|--|--|
| Angola | RD-LD | D | 500 | | | | | |
| Benin | RD-LA | D | 500 | | 19 | 47 (4) | | 4 |
| Botswana | RD-LD | D | 2,000–4,000 | 11,231–15,233 ¹ (2018) | | | | |
| Burkina Faso | RD-LD | I | 1,500–2,000 | | | | | |
| Burundi | RD-LD | U | 500–1,000 | | | | | |
| Cameroon | RD-LD | D | 1,500–2,000 | 3827–4424 ² | 7 | 40 (3) | | 3 |
| Central African Republic | RD-LD | D | 200–500 | | | | | |
| Chad | RD-LD | S | 500 | | | | | |
| Congo | RD-LD | D | 50 | | | | | |
| Côte d'Ivoire | RD-LD | D | 500–600 | | | | | |
| Democratic Republic of the Congo | RD-HD | I | 5,000 | | | | | |
| Equatorial Guinea | RD-LD | U | 50–100 | | | | | |
| Eswatini (formerly Swaziland) | RD-LD | S | 150 | | | | | |
| Ethiopia | W-LD | D | 2,500 | | 1 | 12 (1) | | 1 |
| Gabon | RD-LD | D | 200–300 | | | | | |
| Gambia | RD-LD | U | 40 | | | | | |
| Ghana | RD-LD | U | 150–200 | | | | | |
| Guinea | RD-LA | D | 500 | | | | | |
| Guinea Bissau | RD-LD | D | 200–500 | | | | | |

¹ Chase, M., Schlossberg, S., Sutcliffe, R., and Seonyatseng, E. (2018). Dry season aerial survey of elephants and wildlife in northern Botswana, July–October 2018. Kasane, Botswana: *Elephants Without Borders*. 167.

² [E-AC28-09-03-A1.pdf \(cites.org\)](#)

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|--------------|-------|---|---------------|--------------------------------------|-------|--------------|-------------------|-------|
| Kenya | W-LA | S | 5,000–7,000 | | | 10 (1) | | 1 |
| Malawi | RD-LD | S | 3,000 | | 3 | 80 (7) | 1,793 (342) | 349 |
| Mali | RD-LD | U | 100 | | | | | |
| Mozambique | RD-LD | D | 3,000 | | 889 | 1,444 (120) | | 120 |
| Namibia | RD-LA | I | 3,500 | | 359 | 326 (27) | | 27 |
| Niger | RD-LA | U | 150–200 | | | | | |
| Nigeria | RD-LD | D | 100–200 | | | | | |
| Rwanda | RD-LA | S | 1,000 | | | | | |
| Senegal | RD-LA | U | 500 | | | | | |
| Sierra Leon | RD-LD | U | 100–200 | | | | | |
| Somalia | RD-LD | D | 50 | | | | | |
| South Africa | RD-LA | S | 7,000 | 11,061 ³ (2016) | 2,834 | 3,656 (305) | 147 (27) | 332 |
| South Sudan | RD-LD | D | 2,000–3,000 | | | | | |
| Sudan | RD-LD | D | unknown | | | | | |
| Tanzania | W-LA | S | 20,000 | 26,152–36,020 ⁴ (2018) | 485 | 11,215 (935) | | 935 |
| Togo | RD-LD | U | 250–500 | | | | | |
| Uganda | W-LA | I | 7,000–10,000 | | 8 | 243 (20) | 12,313 (2,345) | 2,365 |
| Zambia | W-LA | S | 40,000–45,000 | | 1067 | 3,048 (254) | | 254 |
| Zimbabwe | RD-LA | S | 5,000 | | 3301 | 5,724 (477) | 4 (1) | 478 |

Table key: Status of population (Status): W = widespread, RD = restricted distribution, LD = low density, HD = high density, LA = locally abundant (Lewison and Pluháček, 2017).

Trends in population (Trend): U = unknown, S = stable, D = decreasing, I = increasing (Lewison and Pluháček, 2017).

*Conversion factor 12 teeth = 1 animal, rounded to nearest whole number (Moneron and Drinkwater, 2021).

** Conversion factor 1 animal = 5.25 kg rounded to nearest whole number (Moneron and Drinkwater, 2021).

Table 2. Estimated quantities of hippo whole organism equivalent (WOE) and ivory (teeth and tusks) commodities seized globally, 2011–2022 (WiTIS)

| Whole organism equivalent (WOE) trade (dead individual, head, skin and skull) | | |
|---|--------------------|--------------------------------|
| Commodity | Number of seizures | Quantity (Number of specimens) |
| Dead individual | 2 | 10 |
| Head | 1 | 1 |
| Skin | 3 | 4 |
| Skull | 4 | 10 |
| Ivory (teeth and tusks) trade | | |
| Commodity | Number of seizures | Quantity (Number of specimens) |
| Teeth | 86 | 1941 |
| Tusk | 4 | 20 |
| Worked ivory pieces | 4 | 32 |

³ Eksteen J., Goodman P., Whyte I., Downs C., and Taylor R. (2016). A conservation assessment of Hippopotamus amphibius. In Child, M.F., Roxburgh, L., Do Linh San, E., Raimondo, D., and Davies-Mostert, H.T., editors. The Red List of Mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa. 9.

⁴ Tanzania Wildlife Research Institute (TAWIRI). 2019. Aerial Wildlife Survey of Large Animals and Human Activities in the Selous-Mikumi Ecosystem, Dry Season 2018. TAWIRI Aerial Survey Report. 65.

| Whole organism equivalent (WOE) trade (dead individual, head, skin and skull) | | |
|---|--------------------|--------------------------------|
| Commodity | Number of seizures | Quantity (Number of specimens) |
| Dead individual | 2 | 10 |
| Head | 1 | 1 |
| Commodity | Number of seizures | Quantity (Weight kg) |
| Teeth | 18 | 2447 |
| Tusk | 1 | 1 |
| Worked ivory pieces | 1 | 150 |

Table 3. Legal protection in hippo range States.

| Range State | Legal Protections per 2016 IUCN Assessment | Legal Protections in proposal |
|----------------------------------|--|---|
| Angola | Unknown | Hunting prohibited (since 2016). |
| Benin | Unknown | Males are partially protected (since 2011); hunting and capture of males allowed by permit; females and young totally protected. |
| Botswana | Total protection | Partially protected (since 2008); hunting, capture, utilization allowed by permit; no age or sex restrictions. |
| Burkina Faso | Total protection | Totally protected (since 1996); hunting for recreational or commercial purposes prohibited. |
| Burundi | Partial protection | Partially protected (since 1937); hunting allowed by permit; no age or sex restrictions. |
| Cameroon | Total protection | Totally protected (since 2006); hunting for subsistence, recreational or commercial purposes prohibited. |
| Central African Republic | Unknown | Totally protected (since 1984); hunting or capture prohibited. |
| Chad | Unknown | Partially protected for adult males (since 2008); only adult males may be hunted by permit; females and young totally protected. |
| Congo | Unknown | Totally protected from hunting (since 2008). |
| Côte d'Ivoire | Unknown | Partially protected for adult males; can be hunted or captured under a licence or permit (since 1965); females and young totally protected |
| Democratic Republic of the Congo | Unknown | Totally protected from capture, hunting, harassing, and deliberate killing (since 2006); illegal to detain, give, sell, exchange, transport any products announcing to contain a product derived from hippos and illegal to exhibit these specimens publicly. However, reportedly, a decree ²⁸ issued in July 2020 established a permit system for hunting of totally protected species including hippo. |
| Equatorial Guinea | Partial protection | Not protected. |
| Eswatini (formerly Swaziland) | Total protection | Partially protected (since 1991); can be hunted and traded under permit; possession of trophies or raw products allowed under permit; no age or sex restrictions. |
| Ethiopia | Total protection | Adult males partially protected (since 2009); hunting and export allowed under permit; females and juveniles totally protected |
| Gabon | Total protection | Totally protected (since 2011); hunting, capture, possession, commercialisation or transport is prohibited. |
| Gambia | Total protection | Partially protected (since 2003); except in protected areas, male and female adult hippos can be hunted with a valid licence; immature animals and females with young are totally protected; export is allowed under permit; domestic sale is not allowed. |
| Ghana | Total protection | Totally protected from hunting, capturing or destruction (since 1971). |
| Guinea | Total protection | Partially protected (since 2018); can be hunted if authorised by the authority in charge of wildlife and protected areas. |

| | | |
|---------------|--------------------|---|
| Guinea Bissau | Total protection | Totally protected (since 2004). |
| Kenya | Total protection | Totally protected (since 2013); hunting, killing, capturing, wounding with intent to hurt a hippo is forbidden; import/export of hippo prohibited. |
| Malawi | Unknown | Adult males and females, partially protected (since 1994); hunting and taking are allowed under licence; export allowed under permit; dependent young and females with dependent young are totally protected from hunting. |
| Mali | Unknown | Partially protected from 1995 until 2019; now totally protected. |
| Mozambique | Total protection | Partially protected (since 1999); adults of either sex may be hunted, including for sport or commerce; as of 2017, young, pregnant females or females with their young are totally protected. |
| Namibia | Partial protection | Partially protected (since 1975); can be hunted under permit; no age or sex restrictions. |
| Niger | Unknown | Totally protected (since 1998); cannot be hunted for commercial purposes; law provides for sport hunting under Ministerial decree, but there is no such decree; Niger reportedly banned hunting, including hippos, in January 2001. |
| Nigeria | Partial protection | Totally protected (since 1991); cannot be killed, hunted or captured except under special licence issued for scientific or administrative purposes in exceptional circumstances; immature or female hippos accompanied by their young cannot be killed. |
| Rwanda | Total protection | Totally protected (since 2008); cannot be hunted, sold, injured, or killed. |
| Senegal | Total protection | Totally protected (since 1986); generally, cannot be hunted or captured. |
| Sierra Leon | Unknown | Partially protected (since 1972); classified as a "game animal" which allows hunting of adults under licence; no age or sex restrictions. |
| Somalia | Unknown | Totally protected (since 1969); generally, cannot be hunted, killed, or captured. |
| South Africa | Total protection | Partially protected (since: unknown); export requires permit issued by national authority; no other national-level management or protection (Kruger National Park culls hippos); some provincial and local management plans and policies exist especially for killing hippos as damage-causing animals, killed for hunting trophies, population management and as damage-causing animals. |
| South Sudan | Partial protection | Partially protected (since 2003); can be hunted or captured with licence, permit or written authorisation; no restrictions based on age or sex. |
| Sudan | Partial protection | Partially protected (since 1986); can be hunted under licence; purchase and sale of hippo parts is permissible. |
| Tanzania | Total protection | Partially protected (since 2009); hippos may be hunted, captured, and exported under permit; killing of young animals, pregnant females, and females accompanied by young is prohibited. |
| Togo | Total protection | Partially protected (since 1968); hunting of adult males for recreational purposes, and capture of any aged or sex, allowed under permit. |
| Uganda | Total protection | Partially protected (since 1996); hunting, farming, ranching, trading, import, export, re-export allowed under permit; no hunting restrictions based on age or sex; on 15th July 2013, hippo ivory trade and export reportedly was banned. |
| Zambia | Partial protection | Partially protected (since 2006); can be hunted, captured, purchased, sold, imported, exported under licence or permit; hunting of dependent young or females accompanied by dependent young prohibited. |
| Zimbabwe | Partial protection | Partially protected (since 1975); prohibited to hunt, take, sell, import or export except under permit; no hunting restrictions based on age or sex. |

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

Proponents: Botswana, 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*, considered functionally extinct with only two surviving individuals). The global wild population was estimated at around 15,940 in 2021, having increased from only a few hundred at most in the 1920s. From a peak of around 21,300 in 2012, numbers declined to around 18,000 in 2017, 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). From 2015–2018, the number of rhinos known to have been poached in Africa is estimated to have declined by a third from 1,349 to 930. This decline has continued, with a reduction by almost half in the number of African rhinos reported in poaching incidents from 2018 to 2021 when range States reported 501 poached rhinos. In 2021, 90% of poaching incidents for African rhinos were reported in South Africa, where around 81% of the population of the Southern White Rhinoceros is currently extant. *Ceratotherium simum simum* was categorised on the IUCN Red List as Near Threatened in 2020.

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 for *in-situ* conservation only, 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." A similar proposal was considered by the Parties at CoP18, but not adopted.

The populations of South Africa and Eswatini are already included in Appendix II (since 1995 and 2005 respectively) with an annotation allowing trade in live animals "to appropriate and acceptable destinations". The definition of this term was amended in 2019 in Res. Conf.11.20 (Rev CoP18), to require that management authorities should be satisfied that any such trade would promote *in-situ* conservation. The annotation proposed is therefore aligned with the restrictions now in place for the other Appendix II populations of this species.

Having become extinct in Namibia before the end of the 19th century, *C. s. simum* was 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 (2021) is between 1,123 to 1,237 individuals, around 900 of which are reported to be in private ownership across 85 subpopulations, 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: since 2008 South Africa recorded the export of 355 live *C. s. simum* to Namibia, 94% of these from 2012 onwards.

From 2008 to 2021 a total of 94 *C. s. simum* were legally hunted in Namibia, indicating an average annual offtake of 0.5–0.6% of the population. Virtually all resulting trophies appear to have been exported.

Reported poaching of *C. s. simum* 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 2015–2021) and is lower than the intrinsic population growth rate. Poaching of Southwestern Black Rhinoceros *Diceros bicornis bicornis* in Namibia has previously been reported as much higher: averaging approximately 50 animals per year for the period 2014–2018 (2.4% of the current population per year). Due to increasing security costs, which are reported not to be offset by available sources of income, 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 facilitate hunting.

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 for conservation through sustainable use.

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 85 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. In addition, the poaching rate is lower than the 2.3% threshold for sustained growth based on continental population analyses. 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 allowing international trade in live animals for *in-situ* conservation only, and hunting trophies, 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 license and track specimens in trade.

An annotation similar to the one proposed 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 functionally extinct in the wild).

Population size and Range

The species is extant in 12 range States. According to population size estimates from 2021, Namibia has the second largest population of C. s. simum (Table 1). Most (952) of the Namibian population are privately owned, with the remaining inhabiting three protected areas.

Table 1. *Estimated number of C. s. simum by country as of the end of 2021 (adapted from Ferreira et al., 2022).*

| Country | Population |
|-----------------------------------|---------------|
| South Africa | 12,968 |
| Namibia | 1,234-1,237 |
| Kenya* | 871 |
| Zimbabwe | 417 |
| Botswana | 242 |
| Eswatini | 98 |
| Uganda* | 35 |
| Rwanda | 30 |
| Democratic Republic of the Congo* | 20 |
| Mozambique | 14 |
| Zambia | 8 |
| Angola | 3 |
| End 2021 Total | 15,940 |

*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 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). Poaching events continued to decline between 2018–2021 on a continental scale, from 3.9% of the continental population (930 individuals) in 2018 to 2.3% (501 individuals) in 2021. Of the most recent 501 incidents of illegal killing of African rhinos reported in 2021, most (90%) were in South Africa (Ferreira et al., 2022). Namibia was one of three range States in which lower poaching rates were recorded during the COVID-19 pandemic (Ferreira et al., 2022).

IUCN Global Category

Near Threatened (assessed 2020, 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 restricted by the minimum level of rainfall per annum. White Rhinos occur in three national parks in Namibia comprising more than 1.5 million ha of suitable habitat. Additionally, 0.5 million to 1 million ha of suitable habitat in Namibian national parks is currently unoccupied by White Rhinos and reported to be able to support around 14,000 individuals.

The White Rhino distribution in Namibia is precisely known due to reintroductions to selected national parks and private land units. The reintroduction of this species in Namibia is taking place in new areas within its former range.

C) Decline in number of wild individuals

White Rhinos went extinct in Namibia at the end of the 19th century. The first reintroduction of *C. s. simum* to Namibia occurred in 1975 when 16 animals were imported from South Africa. The proponents state that the current estimated population size is 1,237 but another report published in 2022 estimates a population size of 1,234 at the end of 2021.

In Namibia, between 2002 and 2021 the White Rhino population grew at an average annual rate of 6.7%, including both intrinsic population increase and imports of live animals from South Africa. Additional estimates for population growth for the species between 2017 and 2021 state that the average annual rate of increase was 5.9% (Ferreira et al., 2022).

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 | CoP18 |
| Current | 1,234–1,237 | Ferreira et al., 2022, SS |

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

The proposal is for trade in live animals for *in-situ* conservation only and hunting trophies. 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 for conservation through sustainable use.

From 2008 to 2021 a total of 94 White Rhinos were legally hunted in Namibia, indicating an average annual offtake of 0.5–0.6% of the population, considerably below the rate of recruitment. Most recently, between 2018 and 2021, Namibia reported an increase in the hunting of White Rhinos, with 17 in 2019 and 22 in 2021 compared to the previous highest number of 11 in 2016 (Ferreira et al., 2022). Between 2008 and 2020, Namibia reported exporting 74 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, 2022).

The proponents state that between 2008 and 2018, under the current Appendix I listing, Namibia exported 54 White Rhinos to Angola, Cuba, the Democratic Republic of the Congo, South Africa, and Zambia. According to the CITES Trade Database, importers reported importing 52 live White Rhinos (reported as *Ceratotherium simum* and *Ceratotherium simum simum*) from Namibia between 2008 and 2020: the Democratic Republic of the Congo (32), China (8), Cuba (5), Saudi Arabia (1), and South Africa (6). In the same time period, Namibia reported exports of 42 live White Rhinos (CITES Trade Database, 2022). Since 2008, South Africa reported the export of 355 live White Rhinos to Namibia. Most (94%) of these were from 2012 onwards although reported exports recently decreased from an average of 52 per year between 2012 and 2017 to an average of eight per year between 2018–2020 (CITES Trade Database, 2019).

Since 2012, Namibia has experienced an increase in losses of rhinoceros from illegal killing. Analysis of data provided by the proponents show that this has increased from an average of three animals per year between 2008 and 2013, to an average of nine animals per year between 2015 and 2021 (see graph). Ferreira et al. (2022) reported total poaching of 471 African rhinos in Namibia between 2008 and 2021. Poaching of Black Rhinos in Namibia has previously been stated to have been much higher than Southern White Rhinos, averaging approximately 50 animals per year for the years 2014–2018, which represents around 2.4% of the current Black Rhino population per year (based on information in the SS CoP18 and Emslie, et al., 2019).

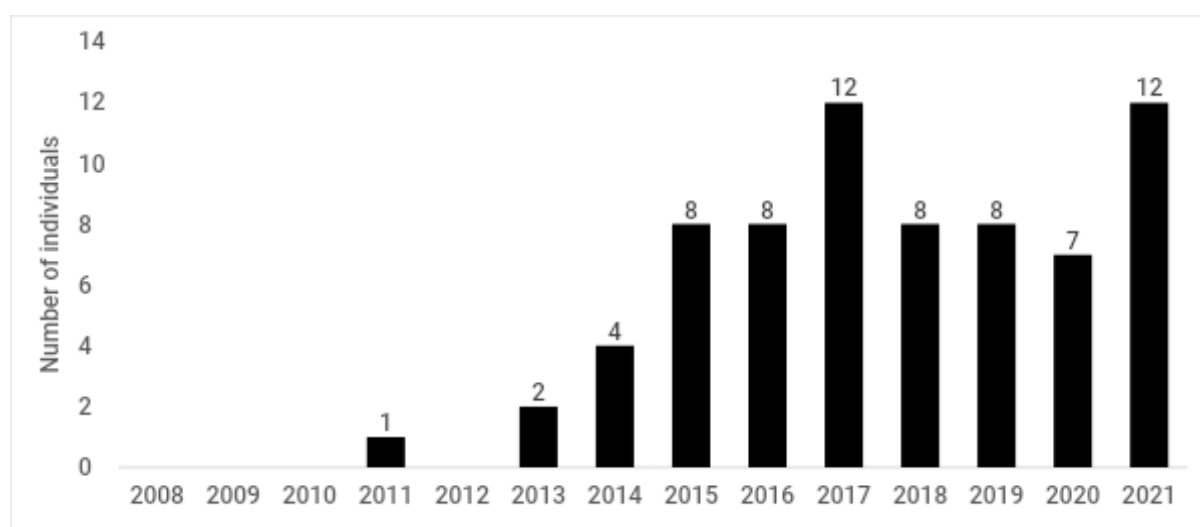


Figure 1. Numbers of *Ceratotherium simum simum* known to have been poached in Namibia (from Supporting Statement).

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 the 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 in 2018 (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)).

Since 2018, reported incidents of poaching of African rhinos in Namibia have declined from 93 (the second highest number recorded since 2006) to 56 in 2019 and 40 in both 2020 and 2021 (Ferreira et al., 2022). Although reported declines in poaching incidents in Namibia could be in part due to lower poaching pressure arising from government responses to COVID-19, it is thought that co-ordination responses amongst law enforcement in range States prior to COVID-19 is also a contributing factor (Ferreira et al., 2022).

Precautionary measures

Species is 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 Rhino 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 Rhinos 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 sent to the RhODIS database housed at Onderstepoort, South Africa. Private owners have their own monitoring systems and studbooks. 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 facilitate hunting. Domestic consumptive use of White Rhinos 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 for *in-situ* conservation only and hunting trophies.

Additional information

Threats

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 rhinos “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 6,195 including 2,156 in Namibia. This proposal is restricted to trade in *C. s. simum* only.

References

- CITES (2016) *Rhinoceroses (Rhinocerotidae spp.) a document submitted by the CITES Secretariat including recommendations from the Standing Committee*. CoP17 Doc. 68. [Working document for CITES CoP17](#)
- CITES (2018) *Questionnaire on the implementation of Resolution Conf. 9.14 (Rev. CoP17) and measures to prevent and combat rhinoceroses poaching and trafficking in rhinoceros horn*. SC70 Doc. 56. Annex 9. [*E-SC70-56-A9.pdf \(cites.org\)](#)
- CITES (2018) *Rhinoceroses (Rhinocerotidae spp.) report of the working group*. SC70 Doc. 56. [SC70 Doc. \(cites.org\)](#)
- CITES (2018) *Summary Friday 5 October afternoon*. SC70 Sum. 12 (Rev. 1). [SC70 Doc. \(cites.org\)](#)
- Emslie, R. Milliken, T., Talukdar, B., Burgess, G., Adcock, K., Balfour, D., and 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)*.
- Ferreira S.M., Ellis, S., Burgess, G., Baruch-Mordo, S., Talukdar, B., and Knight, M. (2022). *African and Asian Rhinoceroses – Status, Conservation and Trade. Report from the IUCN Species Survival Commission (IUCN SSC) African and Asian Rhino Specialist Groups and TRAFFIC to the CITES Secretariat, CoP19 Doc. 75*.

Remove the existing annotation for the Southern White Rhinoceros *Ceratotherium simum simum* population of Eswatini 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*, considered functionally extinct with only two surviving individuals). In 2021 the global wild population of *C. s. simum* was estimated at 15,940, having declined from a peak of around 21,300 in 2012 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). Reported poaching levels have declined since 2017 but remain a serious concern.

The family Rhinocerotidae was included in Appendix I in 1977 in response to major trade challenges driven by demand for rhino horn. In 1995 the South African population of *C. s. simum* was transferred to Appendix II, followed in 2005 by that of Eswatini. Both have 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”.

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. Improved protection, including through a change to national legislation, led to an increase to 90 animals in 2015, reduced to 66 in 2017 due to drought. The population has recovered and is currently (early 2022) estimated at 98, in two protected areas. Estimated total capacity is around 160 rhinos. No trophy hunting of *C. s. simum* takes place because all rhinos occur in reserves where sport and trophy hunting are not permitted. There is limited trade in live animals between Eswatini and South Africa. Current mortality from illegal killing is very low.

This proposal is to delete the existing annotation to the listing of Eswatini's population in Appendix II, with the intention of allowing limited and regulated trade in stocks of *C. s. simum* horn which have been legally collected in the past or recovered from poached Eswatini rhino (totalling 330 kg), as well as of horn to be harvested annually in a non-lethal way in the future, estimated to comprise up to 20 kg per year. A similar proposal was made by Eswatini at CoP18, but not adopted by the Parties.

The Supporting Statement proposes the establishment of a Rhino Horn Trade Protocol, based on Smart Trade principles, centred on a single broker or Central Selling Organisation, managed by professional traders that would be authorised to set prices. The CITES Management Authority of Eswatini would be the sole seller of horn. All horn offered for sale would be properly documented and recorded on a DNA database, a national register and with the CITES Secretariat to safeguard its integrity. All traded specimens would carry DNA certificates and the CITES Secretariat would be requested to monitor consignments closely. Traders would be licensed and required to make an undertaking not to trade horn from illegal sources.

According to the Supporting Statement, at a wholesale price of ca USD30,000 per kg, disposal of the existing stocks would be expected to raise ca USD9.9 million to be invested in an endowment fund. This, along with exports of horn from planned annual non-lethal harvest are predicted to yield an estimated potential annual income of USD1.2 million. The proponent states that its intention is to use proceeds from the horn sales to fund conservation, including security and improved park employee remuneration. The proponent notes that it would reserve the right to adjust prices and amounts adaptively once sales commence. If legal trade were ultimately proven to pose a renewed threat to the population, then further trade would be prohibited by Eswatini.

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) are addressed.

For a species in demand for international trade, which is clearly the case for this taxon, paragraph A 2 of Annex 4 of the Resolution provides the Parties with two options: to decide that existing trade management and enforcement measures are proportionate to anticipated risks; or to require an export quota or other special measure as an integral part of the amendment proposal subject to ongoing review by the Parties. Considering the historical and ongoing impacts of trade on the conservation of rhinoceros species, the risks associated with this proposal are sufficient to merit incorporation of special measures. Although Eswatini has provided some detail on precautions that they would implement, no such measures are integrated into the proposal.

With respect to the proposed trade in rhino horn, there is significant uncertainty about potential market impacts of the release of relatively small volumes of legal supply into the continuing global illegal trade in rhinoceros horn, estimated in the Supporting Statement at around 5000 kg annually.

Furthermore, few details are provided as to how the proposed legal trade would be carried out and controlled; for example, it is not specified which countries might permit legal imports and on what terms they might open currently closed markets, what market segments would be targeted, how and by whom the proposed licensing of traders would be implemented, or how trade would be monitored (including in end-user markets) to avoid laundering. While the CITES Secretariat is identified as playing a significant role, it is not clear how and with what resources it would undertake this work. It is also not clear if authorities in potential importing countries have been consulted.

The proponent states that if the trade were judged to be having a negative impact it would be stopped, but no mechanism is specified for how such an assessment would be undertaken, nor would there be any connection to the formal process provided in section B of Annex 4 of Res. Conf. 9.24 (Rev. CoP17) for such review when special measures apply.

Deletion of the current 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.

Overall, despite the strong case made by the proponent for innovation in efforts to address ongoing global rhinoceros conservation challenges, it is not possible to conclude that this proposal includes satisfactory Precautionary measures, as detailed in Annex 4 of the Resolution.

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*, which became effectively extinct in 2018 when the last male died).

Population size and range

Table 1. *Estimated number of C. s. simum by country as of the end of 2021, adapted from Ferreira et al. (2022).*

| Country | Population |
|-----------------------------------|------------|
| Angola | 3 |
| Botswana | 242 |
| Democratic Republic of the Congo* | 20 |
| Eswatini | 98 |
| Kenya* | 871 |
| Mozambique | 14 |

| | |
|-----------------------|---------------|
| Namibia | 1,234 |
| Rwanda* | 30 |
| South Africa | 12,968 |
| Uganda* | 35 |
| Zambia | 8 |
| Zimbabwe | 417 |
| End 2021 Total | 15,940 |

*Outside of natural range

The continental population, whose trends are defined by South Africa that holds 81% of the global population, is currently declining by 3% per annum (Ferreira et al., 2022).

IUCN Global Category

Near Threatened (assessed 2020, 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 km²; UNEP-WCMC and IUCN, 2022) and Mkhaya Game Reserve (est. 1980) (101 km²; UNEP-WCMC and IUCN, 2022). White Rhinoceros are also likely to be placed in the Mlilwane Wildlife Sanctuary (est. 1961) (46 km²; UNEP-WCMC and IUCN, 2022) in the future.

The White Rhinoceros became extinct in Eswatini due to hunting under colonial rule but was reintroduced in 1965. The population increased to approximately 120 animals by 1988, but poaching during the "rhino war" of 1988–1992 reduced the number to 24. New regulations and increased enforcement led to an increase to 90 in 2015, reduced to 66 in 2017 following several years of severe drought, necessitating major investment in keeping the remaining stock alive. *Following the end of the drought the population has increased to 98 animals as of early 2022 and is currently growing at an annual rate of 8% (IUCN/SSC RSG, in litt., 2022).*

Illegal killing is very rare: only three rhinos are known to have been poached in Eswatini in recent years (two in 2011 and one in 2014).

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 5.2 ± 2.0 kg for the front horn, and 1.9 ± 1.0 kg for the posterior horn. The horns of rhinos grow continuously throughout their life increasing in weight by approximately 1 kg per year.

Since 2004 Eswatini has sold or exchanged, and exported White Rhinoceros bulls to South Africa, and imported White Rhinoceros cows and bulls for genetic and sex ratio purposes.

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 5.2 kg.

The proposed export of 330 kg of rhino horn from stocks and 20 kg per year should be considered in the context of current global demand, which is estimated in the supporting statement at ca 5000 kg annually, and illegal trade flows, recently estimated to involve 575–923 horns per year during 2018–2020 (Ferreira et al., 2022).

Dehorning every 18 months is said to generate approximately 0.8 kg and 1.5 kg per adult female and male White Rhinoceros respectively (Emslie et al., 2019). Research of stress levels shows little impact of regular dehorning every 18 months in a South African White Rhinoceros population that is breeding well (Emslie, in litt., 2016). Other than indicating that it will be non-lethal, the Supporting Statement does not provide details as to how horn would be collected from living individuals. Routine temporary immobilisation for dehorning is a standardised procedure in some South African White Rhinoceros populations with no clear adverse effects on the population or its breeding performance (Emslie, 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 of horn. All horn offered for sale will be properly documented and recorded on a DNA database, a national register and with the CITES Secretariat to safeguard its integrity. DNA differentiates between individuals and species. All traded specimens will carry DNA certificates and the Secretariat will be requested to monitor consignments closely. Therefore, the chances of specimens of similar species, or illegal horn, being included in these transactions will be eliminated. The traders will be licensed and will qualify by undertaking not to trade horn from illegal sources. Any breach will disqualify such traders. Permitted trade will have the added advantage of providing transparent and legal documented information on formerly illegal trade (where there are no data) and will provide incentives to legal traders to protect their legal market.

The trade protocol may carry technical risks imposing on the proposal's intended goals and impact on global rhino populations as follows (IUCN/SSC African Rhino Specialist Group, 2022):

- i) *Lack of critical information on the market dynamics of the proposed legal Smart Trade. Key elements include no specifics of countries consulted (in particular market countries), the form of the consultations and outcomes.*
- ii) *The proposal indicated that retailers would be licensed, but did not specify a mechanism for undertaking this, or provide guidance on how licensing would be regulated.*
- iii) *There is no clear information on the market segment and form of rhino horn use that the Smart Trade would target.*
- iv) *The proposal advocates a highly disciplined, tightly controlled legal trade protocol modelled on a system that operated successfully for other high value products, but does not provide examples.*

According to the Supporting Statement, the proceeds from the sale of stocks should raise approximately USD9.9 million, and will be placed in a conservation endowment fund. It is further claimed that the proceeds of the annual sale of up to 20 kg of horn would raise a further USD600,000 per annum bringing the total recurrent annual income from horn to USD1.2 million. Eswatini would reserve the right to adjust prices and amounts adaptively once sales commence. The Supporting Statement states that the proceeds from the sale of horn will be entirely committed to nature conservation with priority expenditure focussed on the needs of rhinos, incorporating appropriate rural community enhancement.

The proposal does not provide supporting evidence that a legal rhino horn trade could expect USD30,000/kg. The IUCN/SSC African Rhino Specialist Group (2022) reports that it is unaware of any international legal horn prices. The South African Private Rhino Industry has reported high volatility in a domestic market with prices ranging from offers to buy at USD1,693 to USD5,016 per kg (Pelham Jones, Private Rhinos Association, South Africa in litt. to IUCN/SSC Rhino Specialist Group, 2022).

Additional information

Threats

The Supporting Statement draws attention to the severe economic problems created by the COVID pandemic and associated revenue declines, particularly in tourist revenues.

Conservation, management and legislation

The entire rhinoceros family Rhinocerotidae was included in Appendix I of CITES in 1977. The South African population of C. s. simum was transferred to Appendix II in 1995 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 2005, 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.

According to the Supporting Statement, Eswatini's national legislation is preventative and deterrent in nature, rather than remedial. It aims at stopping the killing of rhinos rather than jailing scores of poachers (see section 4.2 on page 3). It is arguably the strongest anti-poaching legislation on the continent and it is being implemented with commitment by law enforcers and the judiciary with the result that it has certainly served to curb rhino poaching in Eswatini, while there are perceived alternative soft targets. Levels of illegal trafficking through Eswatini remain surprisingly low. One trafficking syndicate was disrupted when two Asian nationals from Taiwan POC were arrested with 36 kg of White Rhinoceros horn at King Mswati III International airport. DNA identified the horn to have 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. In the same year two groups of South African rhino poachers were intercepted before they could reach a rhino. Four accused were all convicted and sentenced to 24 years imprisonment without the option of a fine.

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 18 years ago.

The Supporting Statement notes that Eswatini could, in the absence of progress in developing a legal trade in rhino horn, register its entire White Rhinoceros population as a captive breeding operation. Were it to do so, the provisions of Resolution Conf. 12.10 (Rev CoP15) *Registration of operations that breed Appendix-I listed species in captivity for commercial purposes* would apply. *The IUCN/SSC African Rhino Specialist Group notes (in litt., 2022) that the greatest diversity of outcomes across socio-economic-ecological values can be realised when land uses associated with rhinos favour free-ranging conditions in expansive areas.*

Implementation challenges (including similar species)

The only similar species is the Black Rhinoceros *Diceros bicornis* of which there are approximately 6,195 including 48 in Eswatini. This proposal is restricted to trade in *C. s. simum* only.

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

Proponent: 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 to the CITES Appendices. 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, the annotation currently 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 that occurred in 2008, during which time a decision-making mechanism for a process of trade in ivory would be developed. No agreed decision-making mechanism for allowing trade in ivory under the auspices of the Conference of the Parties has been developed.

The proponents argue that they have demonstrably been amongst the most successful in conserving elephants and seek to strengthen further their conservation programmes with finance derived from regulated trade in elephant products. Elephant conservation requires enormous resource inputs and the proponents claim that the costs of law enforcement alone are crippling conservation agencies, at the expense of many other important conservation activities. In Namibia and Zimbabwe rights over wildlife have been legally transferred to local communities. The participation of such communities through conservancy programmes have been pivotal in expanding wildlife numbers and habitat, elephants included.

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 Zimbabwe* ~~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.”

* Although the proposal does not specifically identify the changes allowing Zimbabwe to trade in commercial hides these changes are noted here underlined. References to resolutions that have been amended at subsequent CoPs have not been updated.

If accepted, the proposal's main effect would be to allow exports of registered raw ivory. The Parties therefore need to be satisfied that the Precautionary Measures in Res. Conf. 9.24 (Rev. CoP17) Annex 4 are met with respect to this proposed trade. 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 proponent states 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 Supporting Statement 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. Botswana, Namibia, and Zimbabwe all have recently adopted management plans and South Africa's National Norms and Standards for the Management of Elephants is currently being updated. However, for all the countries, details of the precautionary measures are lacking in the Supporting Statement. 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 (NDFs) and legal acquisition findings). The SS does not provide details as to how the proposed trade would be assessed for sustainability and controlled. However, according to the amended annotation, trade would only be in registered government-owned stocks, originating in

the State (excluding seized ivory and ivory of unknown origin).

The Proportion of Illegally Killed Elephants (PIKE), which provides a CITES adopted measure of poaching trends, estimates given in CoP19 Doc 66.5 for southern Africa (including the four countries in question as well as Angola, Malawi, Mozambique, and Zambia) show that between 2003 and 2021, the highest PIKE estimate for the subregion was in 2011. PIKE likely increased between 2003 and 2011 and subsequently decreased from 2011 to 2021. In the last five years, from 2017 and 2021, there is strong evidence of a downward trend. The unweighted PIKE estimate for 2021 in southern Africa is 0.27 (range: 0.20–0.34) and below the average continental PIKE estimate of 0.40 (range: 0.34–0.46) for the same year.

None of the countries concerned have been identified for consideration in the National Ivory Action Plan (NIAP) process (for which Parties most affected by illegal trade in ivory are selected), in the latest ETIS report to CoP19. In the period 2012–2020 exports of leather items have been reported by Zimbabwe in quantities greater than any of the other countries concerned. Some trade has already been reported as commercial trade. Trade in hides is permitted under the current resolution and therefore it is not clear why added value from processing hides within country would not be permitted. This would still be subject to NDFs. It is difficult to ascertain if exports of tusks have been within the quota since trade is recorded in kg rather than number of tusks (annual report guidelines recommend that tusks are reported by number).

Analysis: The *Loxodonta africana* populations of Botswana, Namibia, South Africa, and Zimbabwe are not small, nor do they have a restricted distribution nor are they undergoing a marked decline. Therefore, these populations, already in Appendix II, do not meet the biological criteria for inclusion in Appendix I. 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, the proposed amendments can be interpreted as new special measures under the terms of Annex 4 of Res. Conf. 9.24 (Rev. CoP17). Adoption of the proposed changes would remove some provisions which are no longer valid, with timeframes having passed and decisions no longer in effect. Trade in leather goods for commercial purposes from Zimbabwe would be permitted, which appears already to be taking place. It is unlikely that the demand for leather goods would be problematic.

If accepted, the main effect of this amendment would be to allow exports of registered raw ivory, but without the level of oversight and control mechanisms previously required by the Conference of the Parties for such trade. The Supporting Statement does not detail safeguards, although it is claimed that there are robust control measures in place at the national level and the only trade permissible would be in registered government-owned stocks, originating in the State (excluding seized ivory and ivory of unknown origin); it is unclear from the Supporting Statement whether the intention is to export only current stockpiles, or future stocks from natural mortality or management related take as well. 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.

Overall, there are significant risks to be considered in relation to the proposed amendments that are not convincingly addressed, particularly with respect to ivory trade. It is not possible to conclude that this proposal includes satisfactory Precautionary measures, as detailed 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.

Range

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

IUCN Global Category

Vulnerable A2a (assessed 2008 ver 3.1) (as a single species).

- African Savanna Elephant *Loxodonta africana* Endangered A2abd assessed 2021 (Gobush et al., 2021a)
- African Forest Elephant *Loxodonta cyclotis* Critically Endangered A2abd assessed 2020 (Gobush et al., 2021b)

Biological criteria for inclusion in Appendix I

A) Small wild population

The elephant populations of Botswana, Namibia, South Africa, and Zimbabwe comprise around 256,000 elephants or 61.6% of all remaining elephants in Africa at the time that their continental status was reviewed most recently.

According to the latest African Elephant Status Report (AESR) released in 2016, the 2015 total continental population based on “estimates” was $415,428 \pm 20,111$. There may be an additional 117,127–135,384 *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 Loxodonta africana population, with an estimated $293,447 \pm 16,682$, with nearly 75% of the population occurring in the Kavango Zambezi Transfrontier Conservation Area (KAZA TFCA) (Thouless et al., 2016).

In 2015 the four countries with Appendix II listed populations held a total of 255,851 *Loxodonta africana* and Botswana remains the stronghold of the species.

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 km² in 2015.

The range of the species within Botswana, Namibia, South Africa, and Zimbabwe covers approximately 504,000 km².

C) Decline in number of wild individuals

Botswana

The AESR 2016 gave an estimate of $131,626 \pm 12,508$ (95% confidence limits), based largely on the Great Elephant Census of 2014. Aerial sample count surveys in 2014 produced an estimate of $129,939 \pm 12,501$ while equivalent surveys in 2010 gave $128,340 \pm 9,938$. These differ considerably from estimates made by the Botswana Department of Wildlife and National Parks ($154,658 \pm 21,253$ in 2006 and $207,545 \pm 21,771$ 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 $22,754 \pm 4,305$ 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 km², 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 km², 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 $82,630 \pm 8,589$ at the time of the last survey for each area. It noted there may be an additional 1,635–1,805 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 km², 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 Matabeleland.

The African Elephant Specialist Group plans to produce a status report for the African Savanna Elephant sometime in 2023 that will incorporate the most up-to-date data from KAZA (Angola, Botswana, Namibia, Zambia, and Zimbabwe) transboundary population. If possible they will provide more information and references through an inf. doc at the CoP. Their preliminary analysis for the countries concerned considers the trends as follows (Co-chairs IUCN/SSC/AfESG, in litt., 2022):

- Botswana: Stable population
- Namibia: Stable and increasing in some populations
- South Africa: Stable and probably increasing in some populations
- Zimbabwe: Stable and unknown in some populations

Recent Government information shows:

The Botswana Elephant Management Plan and Action Plan 2021–2026. estimates 120,000–160,000 elephants in the country (Republic of Botswana, Ministry of Environment, Natural Resources Conservation and Tourism, 2021).

For Namibia, figures presented at the African Elephant Conference 23–24 May 2022 estimated the population at $23,663 \pm 4,397$ (Namibia Ministry of Environment, Forestry and Tourism (2022)).

Zimbabwe National Elephant Management Plan (2021 – 2025) estimate their population at about 82,000 based on 2014 survey results (Zimbabwe Parks and Wildlife Management Authority (2021).

*South Africa presented a figure of 44,326 (43,884 – 44,775) and increasing for their population at the African Elephant Conference, 24–26 May 2022 held at Hwange National Park, Zimbabwe, based on their non-detriment finding (NDF) for *Loxodonta africana* (Scientific Authority of South Africa (2022)).*

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 Supporting Statement reports that robust monitoring systems and control by permits and licences for domestic trade is in place, regular inspections are also done to check on compliance to set standards and security arrangements. Botswana placed a ban on trophy hunting in 2014 and a zero-export quota for exports of hunting trophies was set in 2015 (CITES, 2019b; Thouless et al., 2016). This was lifted in 2019. All four countries now have annual quotas for tusks from trophy hunting in place.*

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 (~ 50 t), and the second to Japan and China in 2008 (~ 102 t) (South Africa was also permitted to sell its ivory in this sale). In accordance with the agreement reached at CoP14, this resulted in a nine-year moratorium until 2017 specified in the current annotation.

*According to the CITES Trade Database, reported exports in *Loxodonta africana* directly from these four range States over the nine year period 2012–2020 has principally involved wild-sourced hunting trophies and trophy tusks*

(Table 1). All trade in non-commercial tusks and ivory carvings reported by weight was by Zimbabwe, with exports including around 54,000 kg of tusks and almost 14,900 kg of ivory carvings. South Africa was the only country to report small leather exports by weight (65 kg), compared to Zimbabwe who exported 7,778 items (see Tables 1 and 2).

Table 1. Exporter reported quantities from Botswana, Namibia, South Africa, and Zimbabwe by weight (kg) and “blank” in the selected products terms, between 2012 and 2020 (equivalent quantities were not reported by importers) (Source: CITES Trade Database).

| Item | Botswana | Namibia | South Africa | Zimbabwe |
|---|----------|---------|--------------|----------|
| Ivory carvings | | | | |
| kg | | | | 14,872 |
| Number of items (Unit not specified or No. not specified) | | | 1,579 | 3,821 |
| Tusks (as hunting trophies) | | | | |
| kg | | | | 54,192* |
| Number of items (Unit not specified or No. of specimens) | 1,824 | 334 | 810 | 6 |
| Trophies | | | | |
| kg | | | | |
| Number of items (Unit not specified or No. of specimens) | 39 | 548 | 892 | 748 |
| Leather products (large) | | | | |
| Number of items (Unit not specified or No. of specimens) | | 13 | 273 | 651 |
| Leather products (small) | | | | |
| kg | | | 65 | |
| m ² | | | 4,576 | |
| Number of items (Unit not specified or No. of specimens) | 75 | 92 | 2,769 | 7,778 |

It appears that Zimbabwe has already been reporting exporting some leather items for commercial purposes according to CITES trade data (see Table 2).

Table 2. Direct Trade in leather items (small and large) reported by Zimbabwe (exports) and importers from Zimbabwe (number of items). Source: CITES trade data (2022).

| Purpose | Importer reported quantity | Exporter reported quantity |
|---------------------------------|----------------------------|----------------------------|
| Hunting Trophies | 787 | 1,418 |
| Personal | 1,170 | 5,572 |
| Circus or travelling exhibition | | 396 |
| Commercial | 3,621 | 1,053 |
| Total | 5,578 | 8,439 |

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

Botswana, Namibia, South Africa, and Zimbabwe have all set annual export quotas for tusks and elephant trophies (see Table 3 below).

Table 3. Quotas posted on CITES website.

| Range State | Quota (years) | Notes |
|--------------|--|--|
| Botswana | 800 tusks (2012–2014, 2020) 0 (2015–2017) | Tusks as part of hunting trophies from 400 elephants Reflecting the ban on trophy hunting |
| Namibia | 180 tusks (2012–2020) | Tusks as part of hunting trophies from 90 elephants |
| South Africa | 300 tusks ((2012–2020, except 2019 where no quota recorded) | Tusks as part of hunting trophies from 150 elephants |
| Zimbabwe | 1,000 tusks (2012–2020, except 2019 where no quota recorded) | Tusks as part of hunting trophies from 500 elephants |

According to SC74 Doc 68, Annex 1 between 2016 and 2019, Botswana and Namibia exceeded quotas for estimated number of tusks in 2016 and Botswana in 2017. As Zimbabwe has reported the export of significant quantities of trophy tusks by weight rather than number, it is difficult to determine whether quotas are being adhered to.

Although the Supporting Statement outlines penalties for illegal hunting and/or trade in Botswana, Namibia, South Africa, and Zimbabwe and enforcement controls and compliance with CITES, it does not provide details as to how the proposed trade will be carried out and controlled beyond the fact 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.”

Analyses of the Elephant Trade Information System (ETIS) prepared for CoP19 (CITES, 2022) indicated that although overall trade levels may have decreased, this could be an artefact of the COVID pandemic. Furthermore, very large-scale seizures are still a cause for concern. None of the countries to which this Proposal pertains were identified through the analyses as those with the highest levels of illegal trade. For the most recent analyses (2017–2019) Botswana and Zimbabwe had made seizures above 500 kg but had not been implicated in any reported seizures made outside their territories. No large-scale seizures had been made by South Africa and Namibia, nor were they implicated in any large-scale seizures made outside the respective countries. But Namibia had a relatively high number of seizures made in country for small and medium raw ivory types, and overall weight of seizures made in country less than 500 kg, which can be interpreted as positive indicators of enforcement effort. Previous ETIS analyses have identified Zimbabwe and South Africa as candidates for the NIAP process, but after reviewing relevant information the Standing Committee agreed that they did not merit inclusion at SC71 in 2019.

There is no information provided in the Supporting Statement on the quantity of ivory currently stockpiled by the proponents.

Additional information

Threats

Land conversion and habitat loss, climate change impacts, human-elephant conflict and poaching are all considered to be threats to the African Elephant.

Not all populations of elephants in all regions are at risk. If all populations are treated the same, there is a huge risk of disincentivising those range States that are managing to contain the threats. In the 1980s, an estimated 100,000 elephants were killed each year and several populations were lost or severely reduced in some regions especially in East, West and Central Africa. Healthy populations have continued to thrive in most parts of southern Africa. In recent years, growing demand for ivory, particularly from Asia, has been linked to the surge in poaching in those range areas where law enforcement is neither strong nor effective.

Data from the CITES programme for Monitoring the Illegal Killing of Elephants (MIKE) indicate that overall poaching levels peaked in 2011, since the programme began in 2002, with a moderately declining trend thereafter (CoP19 Doc. 66.5). According to the latest figures from the MIKE programme, the Proportion of Illegally Killed Elephants (PIKE) between 2003 and 2021, the highest PIKE estimate for the continent was in 2011 and the lowest in 2020. Prior to the maximum value of PIKE in 2011, a trendline analysis shows sufficient evidence to confirm an upward trend (increase in PIKE) from 2003–2011, and a downward trend (decrease in PIKE) from 2011–2021. Over the last five years (2017–2021), the trend analysis shows a downward trend in PIKE, with a level of certainty of over 95%. The 2021 PIKE estimate (0.4) is higher than the 2020 estimate (0.34), while it is within the confidence limits of the 2020 estimate.

In the **MIKE Southern African region**, of which Botswana, Namibia, South Africa, and Zimbabwe are part, PIKE has reduced from 0.52 in 2016 to 0.27 in 2021, a marginal increase from the 2020 value of 0.23 (see Table 4 and Figure 5). The total number of elephant carcasses in Zimbabwe increased from 72 in 2019 to 381 in 2020, however the number of illegally killed carcasses decreased from 8 (2019) to 4 (2020) (MIKE, 2022b).

Table 4. Subregional PIKE data for Africa from 2016–2021 (MIKE, 2022a) For data including confidence limits see https://cites.org/sites/default/files/MIKE/data/Continental_and_Sub_regional_PIKETrend_Africa_2022-03-31_GLMM.xlsx.

| | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 |
|------------------------|-------------|-------------|-------------|-------------|-------------|-------------|
| Central Africa | 0.77 | 0.70 | 0.79 | 0.69 | 0.44 | 0.60 |
| Eastern Africa | 0.43 | 0.36 | 0.44 | 0.29 | 0.30 | 0.28 |
| Southern Africa | 0.52 | 0.50 | 0.38 | 0.30 | 0.23 | 0.27 |
| West Africa | 0.84 | 0.78 | 0.57 | 0.49 | 0.52 | 0.70 |

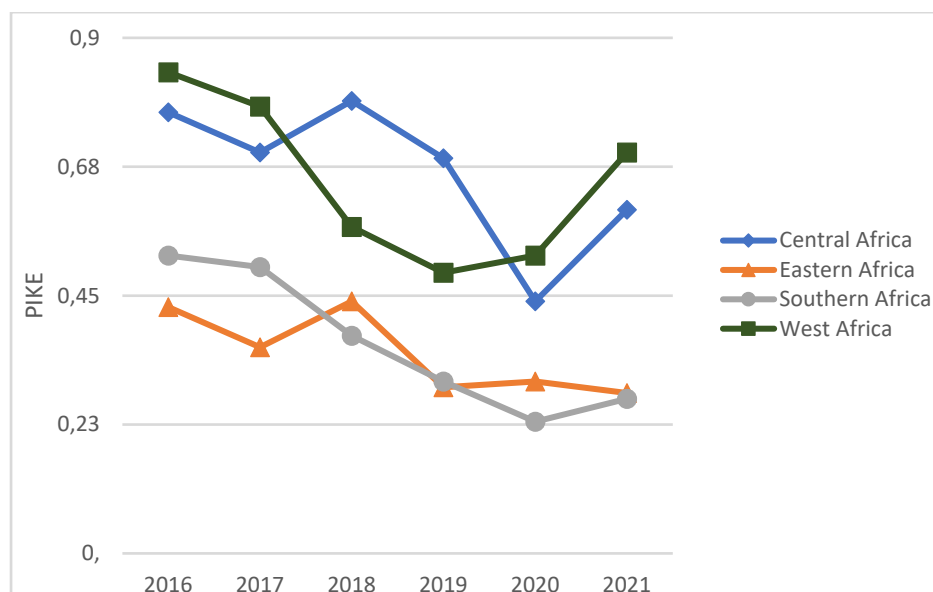


Figure 5: Subregional PIKE data for Africa from 2016–2021 (MIKE, 2022a) For data including confidence limits see https://cites.org/sites/default/files/MIKE/data/Continental_and_Sub_regional_PIKETrend_Africa_2022-03-31_GLMM.xlsx

The National Ivory Action Plans 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 Res. 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, 2019). South Africa has also been reported as a country with an involvement in illegal ivory trade for several years (CITES, 2019), for example, in 2017, Viet Nam seized ca. 2.5 t of ivory from South Africa (CITES, 2019). Previous ETIS analyses have identified Zimbabwe and South Africa as candidates for the NIAP process, but after reviewing relevant information the Standing Committee agreed that they did not merit inclusion at SC71 in

2019. None of the countries concerned have been identified in the latest ETIS report as candidates for inclusion in the NIAP process (CoP19 Doc. 66.6).

Conservation, management and legislation

Management measures vary greatly though the continent. They range from creation of migration corridors and transfrontier parks and conservation areas, translocation of animals, creation of artificial waterholes, fencing and deterring animals from crops with, for example chili peppers or beehives, to shooting of problem animals. Culling has not been employed as a management tool since Zimbabwe halted the practice in 1988 and South Africa in 1995.

Botswana's Elephant Management Plan 2021–2026 was published in 2021. In 2020 Namibia published its National Elephant Conservation and Management plan 2021/2022–2030/2031. Zimbabwe has an updated Elephant Management Plan (2021–2025). South Africa has developed the Norms and Standards of Elephants in 2020 which is in a process of consultation, and is yet to be published in the gazette for implementation. Since 2008 elephants have been managed in accordance with the National Norms and Standards for the Management of Elephants in South Africa (Government Gazette No. 30833, 29 February 2008). The species is listed as protected in terms of section 56 of NEMBA and various provincial ordinances and acts provide further legislative protection. South Africa has a National Red-List Assessment for African Savanna Elephants, published in 2016 that list the species as Least Concern. In 2019 a process has been initiated to develop a National Elephant Conservation Strategy, which is anticipated to be completed in 2023. The strategy will provide a high-level vision and strategic objectives for the management and long-term conservation of elephants in South Africa, while contributing to the well-being of people. The strategy will also assist in the further revision of the current Elephant Norms and Standards (Co-chairs IUCN/SSC AfESG, in litt., 2022).

The proponents recognise the importance of conservation incentives for local communities, whose agricultural livelihoods should clearly be supported within national development frameworks, and whose partnership is essential for effective and sustainable conservation of biodiversity. However, the success of community-based conservation is in no way dependent on international ivory sales, which are never likely to play any significant role in the development of community engagement in wildlife-related benefit sharing, conservation-compatible rural land use, and protection of wildlife from illegal trade.

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). Although up to 70% of the species' range is believed to lie in unprotected land, most large populations occur within protected areas (Blanc, 2008). 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).

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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, Equatorial Guinea, Mali, Senegal

Summary: This Proposal applies only to the African Elephant *Loxodonta africana* populations of four contiguous southern African countries: Botswana, Namibia, South Africa, and Zimbabwe. The most comprehensive and reliable information on distribution and population of this species is contained in the African Elephant Database (AED), maintained by the IUCN SSC African Elephant Specialist Group (AfESG) and presented in the African Elephant Status Reports (AESR), the latest of which was published in 2016 (the AfESG plans to produce an updated report in 2023). The 2016 AESR estimates a combined range in the four countries considered here as approximately 500,000 km² and a total population estimate of at least 255,000. This amounts to around 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. |

CoP19 Doc 66.5 (Report on Monitoring the Illegal Killing of Elephants (MIKE)) contains the most up-to-date synthesised information on illegal killing of elephants, based on information from 2003 until the end of 2021. It reports on the proportion of illegally killed elephants (PIKE) at 69 sites in 32 countries in Africa and 30 sites in 13 countries in Asia. A PIKE level of over 0.5 has been flagged in past reports as being of particular concern although reference to this “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 0.27 in the most recent assessment (2021), having decreased from 0.41 in 2016. This is the lowest PIKE for any of the African subregions. Over the last five years (2017–2021), the trend analysis shows a downward trend in PIKE, however, the 2021 PIKE estimate (0.4) is higher than the 2020 estimate (0.34).

The Supporting Statement (SS) of the Proposal deals extensively with the whole *Loxodonta africana* population, which is not the subject of the amendment proposal. It argues that there has been a reduction of more than 50% in the continental population in the past three generations. The proponents argue that a continental decline in African Elephants continues under the present split-listing and despite safeguards including a nine-year moratorium on ivory trade proposals, a recommendation of domestic ivory market closures, demand reduction strategies, and, country-

specific National Ivory Action Plans (NIAPs). In order to rectify this, the proponents therefore consider a transfer of Appendix II elephant populations to Appendix I as the next logical, essential and urgent step to reverse the continental decline.

Analysis: The *Loxodonta africana* populations of Botswana, Namibia, South Africa, and Zimbabwe are not small, nor do they have a restricted range and they are not undergoing a marked decline. Therefore, these populations do 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 Supporting Statement of the current Proposal and of CoP19 Prop. 4 submitted by 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.

This Proposal seeks to list all African Elephant populations in Appendix I in order to offer maximum protection under CITES in the face of the ongoing threat posed by the high demand from the ivory trade, a reduction of more than 50% of the continental population in the past three generations that is understood to be continuing and likely irreversible, and the enforcement problems that current split listing creates. The proponents argue that the threatened status of the species on a continental-scale calls into question the current split listing established in 1990 and that “No one country, region or agency can tackle illegal wildlife trade alone. Collective action across source, transit and destination states is essential”.

Taxonomy

Under current CITES Taxonomy the African Elephant *Loxodonta africana* remains a single taxon. Authorities, including IUCN, now recognise two distinct species—African Savanna Elephants (*Loxodonta africana* Blumenbach, 1797) and African Forest Elephants (*Loxodonta cyclotis* Matschie, 1900). The Animals Committee has discussed the nomenclature for African elephants and concluded that it would be prudent not to add this case to nomenclature matters for adoption at CoP19 under Res. Conf. 12.11 *Standard Nomenclature*. Decisions have been drafted in CoP19 Doc 84.1 that would direct the Secretariat and committees to consider the recommendations for consideration at CoP20.

Range

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

IUCN Global Category

Loxodonta africana assessed as Vulnerable 2008 (as a single species).

African Savanna Elephant *Loxodonta africana* Endangered A2abd assessed 2021.

African Forest Elephant *Loxodonta cyclotis* Critically Endangered A2abd assessed 2020.

Biological criteria for inclusion in Appendix I

a. Small wild population

According to the most recent African Elephant Status Report (AESR) released in 2016, the 2015 total continental population based on “estimates” was $415,428 \pm 20,111$. *There may be an additional 117,127–135,384 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 $293,447 \pm 16,682$ 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 elephant populations in Appendix II had a corresponding 2015 total of 255,851 and country totals as follows: Botswana 131,626, Namibia 22,754, South Africa 18,841, and Zimbabwe 82,630. *These figures are detailed in Table 1.*

Table 1. Populations of *Loxodonta africana* in 2002, 2006, and 2015 in Botswana, Namibia, South Africa, and Zimbabwe (based on AESRs).

| 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 | |

The African Elephant Specialist Group plans to produce a status report for the African Savanna Elephants sometime next year 2023 that will incorporate the most up-to-date data from the KAZA (Angola, Botswana, Namibia, Zambia, and Zimbabwe) transboundary population. If possible they will provide more information and references through an inf. doc at the CoP. Their preliminary analysis for the countries concerned consider the trends as follows (Co-chairs IUCN/SSC/AfESG, in litt., 2022):

- Botswana: Stable population
- Namibia: Stable and increasing in some populations
- South Africa: Stable and probably increasing in some populations
- Zimbabwe: Stable and unknown in some populations

Recent government information shows:

The Botswana Elephant Management Plan and Action Plan 2021–2026 estimates 120,000–160,000 elephants in the country (Republic of Botswana, Ministry of Environment, Natural Resources Conservation and Tourism, 2021).

For Namibia, figures presented at the African Elephant Conference 23–24 May 2022 estimated the population at 23,663 ± 4,397 (Namibia Ministry of Environment, Forestry and Tourism (2022)).

The Zimbabwe National Elephant Management Plan (2021–2025) estimate the population at about 82,000 based on 2014 survey results. (Zimbabwe Parks and Wildlife Management Authority (2021).

South Africa presented a figure of 44,326 (43,884–44,775) and increasing for their population at the African Elephant Conference, 24–26 May 2022, held at Hwange National Park, Zimbabwe, based on their non-detriment finding for *Loxodonta africana* (African Savanna Elephant) (Scientific Authority of South Africa (2022)).

b. Restricted area of distribution

The total range area of the African Elephant across Africa (defined as “Known” and “Possible”) was approximately 3.1 m km² in 2015; this represents a 6% decrease compared to 2007 (3.3 m km² reported), and a 36% decrease from 2002, which was itself an 8% decrease from the 1998 estimate. Some of this apparent range contraction may be due to improved information. However, despite the caveats over drawing quantitative conclusions about the rate of range loss, it is safe to accept that there has been a steady trend of decline over time in the range available for elephants.

The most recent information on range from the African Elephant Database (2016) shows the elephants' range over the four Appendix II countries now covers just over 504,000 km² (Botswana 228,073 km², Namibia 164,069 km², South Africa 30,651 km², Zimbabwe 81,228 km²). AED range data for southern Africa indicate there was a regional decline of some 21% of "Known and Possible" range combined from 2002–2015 (however, there was a very slight increase between 2006–2015, attributable to range expansion in Botswana only). *An update on range changes since 2016 will be published in 2023 in the next iteration of the Status Report as they are currently under review (Co-chairs of IUCN/SSC AfESG in litt., 2022).*

c. Decline in number of wild individuals

"C. A marked decline in population size in the wild, which has been either...:

i) observed as ongoing or as having occurred in the past (but with a potential to resume); or

ii) inferred or projected on the basis of ...the following:

- a decrease in the area of habitat

- levels or patterns of exploitation;"

Cii inferred or projected on the basis of ...the following:

- a decrease in the area of habitat (see above)

ii) inferred or projected on the basis of ...the following:

- levels or patterns of exploitation;"

The AESR 2016 stated (p.29) that: "This is the first African Elephant Status Report in 25 years which has reported a continental decline in elephant numbers."

Botswana: In 2016, the IUCN African Elephant Status Report noted an increase in elephant poaching in Botswana. In 2019, researchers found that despite stable elephant populations, numbers of (fresh) elephant carcasses in northern Botswana increased by 593% between 2014–2018, indicating a growing poaching crisis, with estimates suggesting a minimum of 385 elephants were poached in Botswana between 2017–2018. There is evidence that ivory poaching on the scale of hundreds of elephants per year has been occurring in northern Botswana since 2017 or possibly earlier. Elephant poaching and ivory trafficking is a transboundary issue in southern Africa, with reports of poachers crossing into Zimbabwe, Botswana, Zambia, and Namibia to poach ivory. Since 2019, very little information has been published about elephant poaching in Botswana.

Namibia

In 2016, the IUCN African Elephant Status Report noted an increase in elephant poaching in Namibia's Zambezi region. In 2020, 62 tusks were seized and 64 arrests were made relating to poaching and ivory.

South Africa

In 2016, the IUCN African Elephant Status Report noted an increase in South Africa's Kruger Park elephant population.

Zimbabwe

In 2014, elephant population numbers fell by 76% in Zimbabwe's Sebungwe ecosystem following a noted increase in numbers of elephant carcasses which began in the early 2000s.

The trend in poaching levels, while stabilising recently, remains one of continuing threat, with even the Appendix II countries facing potentially large population declines, in line with the trend in numbers reported in the AED results.

The MIKE programme is considered to provide conservative estimates of poaching rates based on ranger patrol monitoring. The MIKE sites with the best quality data are relatively intensively managed; therefore, PIKE values may underrepresent poaching mortality in a country if heavily based on such sites. *The Proportion of Illegally Killed Elephants (PIKE) estimates given in CoP19 Doc 66.5 for southern Africa (including the four countries in question as well as Angola, Malawi, Mozambique, and Zambia) show that between 2003–2021, the highest PIKE estimate for the subregion was in 2011. PIKE likely increased between 2003–2011 and subsequently decreased from 2011–2021. In the last five years, from 2017–2021, there is strong evidence of a downward trend. The unweighted PIKE estimate for 2021 in southern Africa is 0.27 (range: 0.20–0.34) and below the average continental PIKE estimate of 0.40 (range: 0.34–0.46) for the same year. Values above 0.5 have been flagged as being of particular concern, suggesting that more of the elephant deaths reported are attributed to illegal killing than to natural causes (CITES, 2018a). However, this reference point is omitted from the latest MIKE report (CoP19 Doc 66.5).*

Analyses of the Elephant Trade Information System (ETIS) prepared for CoP19 (CoP19 Doc 66.6) indicated that although overall trade levels may have decreased, this could be an artefact of the COVID pandemic. Furthermore, very large-scale seizures are still a cause for concern. None of the countries to which this proposal pertains were identified through the analyses as those with the highest levels of illegal trade. For the most recent analyses (2017–2020) Botswana and Zimbabwe had made seizures above 500 kg but had not been implicated in any

reported seizures made outside their territories. No large-scale seizures had been made by South Africa and Namibia, nor were they implicated in any large-scale seizures made outside their respective countries. However, Namibia had a relatively high number of seizures made in the country for small and medium raw ivory types, and the overall weight of seizures made in the country was less than 500 kg, which can be interpreted as positive indicators of enforcement effort. Previous ETIS analyses have identified Zimbabwe and South Africa as candidates for the National Ivory Action Plan (NIAP) process, but after reviewing relevant information the Standing Committee considered that they did not merit inclusion at SC71 in 2019.

Forensic examination of shipments of seized ivory from 2002–2019 showed an increase in poaching in southern Africa. Of the 196 forensically analysed tusks from seizures, 172 were inferred to originate in the KAZA Transfrontier Conservation Area, pointing to a newly emerging poaching hotspot in southern Africa.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

The populations concerned are subject to an annotation (2) allowing the trade in some specified products.

Additional information

Threats

Across the continent, the long-term threat to elephants is the loss or conversion of habitat through human expansion into elephant range, associated human–elephant conflict and the impacts of climate change. In central African forests, the impacts of forestry activities including both deforestation (habitat loss) and the building of roads (increasing human access) pose serious long-term and ongoing threats.

Poaching remains a major threat to elephants.

Conservation, management and legislation

Management measures vary greatly though the continent. They range from creation of migration corridors and transfrontier parks and conservation areas, translocation of animals, creation of artificial waterholes, fencing and deterring animals from crops with, for example chili peppers or beehives, to shooting of problem animals. Culling has not been employed as a management tool since Zimbabwe halted the practice in 1988 and South Africa in 1995.

Botswana's Elephant Management Plan 2021–2026 was published in 2021, in 2020 Namibia published its National Elephant Conservation and Management plan 2021/2022–2030/2031, Zimbabwe has an updated Elephant Management Plan (2021–2015).

South Africa has developed the Norms and Standards of Elephants in 2020 which is in a process of consultation and is yet to be published in the gazette for implementation. Since 2008 elephants have been managed in accordance with the National Norms and Standards for the Management of Elephants in South Africa (Government Gazette No. 30833, 29 February 2008). The species is listed as protected in terms of section 56 of NEMBA and various provincial ordinances and acts provide further legislative protection. South Africa has a National Red List Assessment for African Savanna Elephants, published in 2016 that list the species as Least Concern. In 2019 a process was initiated to develop a National Elephant Conservation Strategy, which is anticipated to be completed in 2023. The strategy will provide a high-level vision and strategic objectives for the management and long-term conservation of elephants in South Africa, while contributing to the well-being of people. The strategy will also assist in the further revision of the current Elephant Norms and Standards (Co-chairs IUCN/SSC AfESG in litt, 2022).

The proponents recognise the importance of conservation incentives for local communities, whose agricultural livelihoods should clearly be supported within national development frameworks, and whose partnership is essential for effective and sustainable conservation of biodiversity. However, the success of community-based conservation is in no way dependent on international ivory sales, which are never likely to play any significant role in the development of community engagement in wildlife-related benefit sharing, conservation-compatible rural land use, and protection of wildlife from illegal trade.

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). Although up to 70% of the species' range is believed to lie in unprotected land, most large populations occur within protected areas (Blanc, 2008). 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).

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Transfer of Mexican Prairie Dog *Cynomys mexicanus* from Appendix I to Appendix II

Proponent: Mexico

Summary: The Mexican Prairie Dog *Cynomys mexicanus* is one of five species of prairie dog. It is a rodent endemic to north-western Mexico classified as Endangered on both the IUCN Red List (2018) and on the List of Species at Risk in Mexico (2019) due to a small area of occupancy, high habitat fragmentation, low quality of remaining habitat, and restricted number of remaining subpopulations. *Cynomys mexicanus* was listed in Appendix I in 1975. It is one of two species of prairie dog occurring in Mexico (the other being *C. ludovicianus*) and is the only prairie dog included in the Appendices.

Estimates of population sizes are not available for all colonies. According to the IUCN Red List assessment (2018), the population trend is decreasing. However, the number of colonies has reportedly remained stable since 1999 at 50–60. Ecological models indicate a potential range of over 4,300 km², but satellite imagery suggests that the real extent of occurrence is limited to 215 km², taking into account habitat quality and known occurrence.

The primary threats to *C. mexicanus* include habitat loss and fragmentation due to changes in land-use for agricultural purposes and through overgrazing. Additional threats are from hunting and poisoning as the species is seen as an agricultural pest.

No national use is recorded for the species and only two harvests have been registered (150 specimens in 2008 and 130 in 2010) for reintroduction purposes from one of the four Wildlife Conservation Management Units (UMAs - i.e., the only legal entity that allows wildlife management in Mexico). According to the Mexican Law Enforcement Authority (PROFEPA), a total of nine specimens of *C. mexicanus* were seized from 2013 to 2019 within the country. The last instance of international trade occurred in 2012, when wild-caught specimens were exported for scientific purposes.

Analysis: This proposal results from the Periodic Review of the Appendices (Res. Conf. 14.8 (Rev. CoP17). Remaining populations of *Cynomys mexicanus* are characterised by a small area of occupancy, high habitat fragmentation and low habitat quality. The number of colonies has remained relatively stable over the past 20 years. There is little information on population trends though there are no indications of a marked recent decline. The species may still meet the biological criteria for inclusion in Appendix I. However, there is no evidence of any trade demand and it is highly unlikely that a transfer to Appendix II would stimulate trade in the species. The Precautionary measures in Annex 4 of Res. Conf. 9.24 (Rev. CoP17) appear to be met. This proposal was supported by the Animals Committee.

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

Cynomys mexicanus (Merriam, 1892).

Range

Mexico

IUCN Global Category

Endangered B2ab (i,ii,iii,iv,v) (assessed 2018, ver 3.1)

Biological criteria for inclusion in Appendix I

A) Small wild population

Estimates of population sizes are not available for all colonies. A recent study by Medellín et al. (2019) reported a range of four to 42 individuals per hectare and 26 to 1,588 individuals per colony in the regions of San Luis Potosí and Zacatecas. Previous studies (González-Urbe, 2011; Scott-Morales et al., 2005) recorded the number of colonies and average density of individuals per colony within the states of Coahuila, San Luis Potosí and Nuevo León. The results are summarised in Table 1.

Table 1. Number and average area of colonies and density of *Cynomys mexicanus* in Mexico.

| State | Number of colonies | Average area per colony (ha) | Average density (ind/ha) | Area of distribution (ha) | Percentage of the distribution compared to total area (%) |
|-----------------|--------------------|------------------------------|--------------------------|---------------------------|---|
| Nuevo León | 13 | 949.6 | 3.2*** | 12,345 | 57.2 |
| Coahuila | 21 | 437.8 | 6.1** | 8,759 | 40.6 |
| San Luis Potosí | 12 | 36.3 | 1.6** / 15.6* | 435 | 2.0 |
| Zacatecas | 3 | 14.3 | 20.6* | 43 | 0.2 |

* Medellín et al. (2019), **Scott-Morales et al. (2005), *** González-Urbe (2011).

B) Restricted area of distribution

Endemic to Mexico, the species has a distribution limited to a small region (approximately 215 km²) in the northeastern states. It occupies grassland habitats along valleys and intermontane basins within the states of Coahuila, Zacatecas, San Luis Potosí and Nuevo León. Declines in extent of occupancy have been recorded, and a loss of approximately 60% of its historical range was estimated in 1996 (Scott-Morales et al., 2005). More specifically, the habitat range of *C. mexicanus* progressively decreased from 800 km² in 1985, down to 478 km² in 1993, 322 km² in 1999, 284 km² in 2011 and finally 215 km² as estimated in the Periodic Review for *C. mexicanus* from 2019. The final estimate implies a 73% reduction in the historical range of the species. Moreover, according to Scott-Morales et al. (2005), the future of the species in the marginal southern parts of its distribution is not assured due to the low abundance and small size of most of the colonies.

C) Decline in number of wild individuals

According to the IUCN Red List assessment this species is Endangered (EN) and its population trend is decreasing. However, the number of colonies has remained stable overall since 1999.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

No national use is recorded for the species. Only two international trade transactions have been recorded since the species was listed in CITES Appendix I in 1975: one in 2012 (200 biological samples destined for Germany), and one in 2004 (300 tissue samples destined for the USA). All the specimens were wild-sourced (source code: "W") and all exports were for scientific purposes. There is no evidence of illegal international trade in the species. International trade does not pose a threat to the survival of the species.

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.

There are no official records of the sale of specimens or specimens of this species, nor does there appear to be a national or international market that threatens wild populations. The only records of international trade are for scientific specimens.

No species similar to *Cynomys mexicanus* are currently listed in the appendices.

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

There are no current quotas in place for this species.

Additional information

Threats

The main anthropogenic impacts on the species include habitat loss and fragmentation due to intensive agriculture, livestock farming (overgrazing, which affects reproduction and offspring survival, as well as social interaction between individuals in the colony), and hunting and poisoning as *C. mexicanus* is considered to be an agricultural pest.

Conservation, management and legislation

The National Commission for Protected Areas (CONANP/SEMARNAT) in Mexico has established the Action Program for the Conservation of Species (PACE) that includes goals and targets for the conservation of *Cynomys mexicanus* alongside other species. No regular monitoring of the species occurs—populations are mainly monitored by academics from various Mexican institutions.

In Mexico, wild specimens can only be used for commercial purposes if conservation activities are being carried out under the Wildlife Conservation Management Units (UMA) scheme. Any use related to scientific research must meet criteria established by legislation and get approval from the authorities. Moreover, hunting and any other uses of *C. mexicanus* are prohibited in the following protected areas: Llano de la Soledad, La Trinidad and La Hedionda (all situated in the State of Nuevo León). These reserves protect 30% of the current distribution of the species. In the USA, the species is listed on the US Endangered Species Act (ESA) as Endangered and therefore any actions leading to its capture, import, export, interstate or foreign commerce are prohibited.

Captive breeding

Cynomys mexicanus has been captive-bred at the Desert Museum in Saltillo, Coahuila, since 1999. According to Medellín et al. (2019), this population totals 50–60 individuals.

Implementation challenges (including similar species)

Similarities to the Black-tailed Prairie Dog *Cynomys ludovicianus*, which also inhabits parts of Mexico, exist. However, the two species can be distinguished by the shape and morphological characteristics of the skull. *Cynomys ludovicianus* is not traded for its meat nor is it found within the pet trade.

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Transfer of Aleutian Cackling Goose *Branta canadensis leucopareia* from Appendix I to Appendix II

Proponent: United States of America

Summary: The Aleutian Cackling Goose *Branta canadensis leucopareia* is a migratory bird found in Japan, Mexico, Russian Federation and USA. Most individuals breed on the Aleutian and Semidi Islands in Alaska, USA, and overwinter in California or Oregon. After its near-extinction caused by predation by non-native foxes introduced for the fur trade between the mid-18th and 20th centuries, the subspecies has bounced back due to intensive conservation efforts (including hunting closures, re-introductions and habitat conservation measures) and currently numbers over 160,000 individuals. The subspecies was included in Appendix I in 1975. It is now considered to be a subspecies of *Branta hutchinsii* the Cackling Goose rather than *Branta canadensis* the Canada Goose. Two other geese in the genus *Branta* are also included in the CITES appendices: the Nene or Hawaiian Goose *B. sandvicensis* (Appendix I) and the Red-breasted Goose *B. ruficollis* (Appendix II).

Having been originally included in the U.S. Endangered Species Act in 1973, in response to the recovery of the various populations (especially the western Aleutian population), the subspecies was down-listed to threatened in 1990 and in 2001 removed from the list altogether.

In the USA, the subspecies is now managed as a game bird. Hunting remains restricted in key breeding areas within the Aleutian Islands and is prohibited in northern coastal Oregon to protect the much smaller Semidi Islands population. Incidental take of the species may occur but does not appear to constitute a significant threat.

According to the CITES Trade Database, most international trade has been to assist in conservation measures, including the international transport of primarily captive-bred birds for either re-introduction efforts or captive breeding. Since the inclusion of the subspecies in CITES Appendix I in 1975, only three records indicate international trade of wild specimens for commercial or trophy purposes. Furthermore, no illegal trade has been reported by any state in the USA.

Analysis: This proposal is based on the outcome of the Periodic Review of the Appendices in accordance with Res. Conf. 14.8 (Rev. CoP17). *Branta canadensis leucopareia* (*Branta hutchinsii leucopareia*) no longer meets the biological or trade criteria for inclusion in Appendix I, as the wild population is neither small nor in decline, its distribution range is not currently restricted, nor is the subspecies in demand for international trade. Transfer of the taxon to Appendix II is in accordance with the Precautionary measures in Annex 4 of Res. Conf. 9.24 (Rev. CoP17) and will be accompanied by the continued implementation of distinct management measures including monitoring surveys and harvest strategies. This proposal was supported by the Animals Committee.

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

Branta canadensis leucopareia (Brandt, 1836).

The CITES standard reference for birds uses *Branta canadensis leucopareia*, and *Branta hutchinsii leucopareia* is considered to be synonymous.

The scientific classification and name of this taxon has changed to *Branta hutchinsii leucopareia* (Banks *et al.*, 2004). The Canada goose and the Cackling Goose were split into two distinct species (*Branta canadensis* and *Branta hutchinsii*, respectively) and this specific taxon was determined to be a subspecies of *Branta hutchinsii*.

Range

Japan, Mexico, Russian Federation, USA.

IUCN Global Category

Branta hutchinsii (whole species Cackling Goose): Least Concern (assessed 2020, ver 3.1)

Biological criteria for inclusion in Appendix I

A) Small wild population

The distribution of *Branta canadensis leucopareia* extends across the USA, the Russian Federation, Japan and Mexico. The majority of individuals breed on 10–12 Alaskan islands in the Aleutian and Semidi Island chains and overwinter in the Central Valley of California or near Pacific City, Oregon. A small population of mostly captive-bred specimens was re-introduced in the Kuril Islands of the Russian Federation—these birds normally overwinter in northern Japan.

There are three major, genetically distinct populations of *Branta canadensis leucopareia*—one in the western Aleutian Islands, one in the Semidi Islands, and finally the Eurasian population.

Overall, the subspecies numbers 160,000 individuals in the western Aleutian breeding segment, approximately 300 individuals in the Semidi Islands breeding segment, and more than 1,700 individuals in Eurasia.

According to the IUCN Red List assessment (2020), the current population trend of the whole species (Branta hutchinsii) is increasing.

B) Restricted area of distribution

Historically, the breeding range of the Aleutian Cackling Goose was believed to stretch from Kodiak, Alaska, westward through the Aleutian Islands of the USA, the Commander Islands of the Russian Federation, and the Kuril Islands of Japan. In between the 1960s and 1980s, only three known remnant breeding sites remained within the USA: Buldir Island in the western Aleutians, Chagulak Island in the central Aleutians, and Kiliktagik Island farther east in the Semidi Islands. Today, the breeding range of *B. canadensis leucopareia* includes 10 to 12 islands in Alaska and Ekarma Island in the Russian Federation.

The historical wintering range of *B. canadensis leucopareia* likely extended from British Columbia, Canada, to northern Mexico and included Japan, and amounted to hundreds of sites overall. At present, the subspecies inhabits over 30 wintering and staging areas in the western USA (i.e., near the San Joaquin River National Wildlife Refuge, on ranches near Modesto and in the Sacramento-San Joaquin Delta in California; near Nestucca Bay National Wildlife Refuge, Oregon Islands National Wildlife Refuge and on dairy farms in Oregon).

Additionally, *B. canadensis leucopareia* has been recorded in Baja California and Baja California Sur, in the Colorado River delta, on Ekarma Island in the Russian Federation, and in northern Japan.

The subspecies is no longer believed to occur in Canada.

C) Decline in number of wild individuals

Western Aleutian population:

The introduction of foxes on goose breeding islands to meet demand for fur severely intensified in the early 1900s, and over 450 islands had introduced fox populations by the 1930s. By the 1960s, fox predation on geese had reduced the wild population to a few hundred individuals found on a handful of Alaskan islands. Consequently, the Aleutian Cackling Goose was one of the first species to be included and protected by the Endangered Species Act in the USA in 1973 and *Branta canadensis leucopareia* was listed in CITES Appendix I in 1975.

However, intense conservation efforts (including fox eradication, hunting closures, re-introductions and habitat conservation) led by the U.S. Fish and Wildlife Service and its partners allowed the population to increase progressively—first through the 1970s and 1980s (at a rate of approximately 15% per year) and consequently in the 1990s and 2000s, reaching 62,800 birds in 2002. Currently, the population occupying the Aleutian Islands number 160,000 individuals.

Semidi Island population:

The population is relatively stable with little increase recorded over the years due to poor juvenile recruitment. At present, approximately 300 individuals breed there.

Eurasia population:

Branta canadensis leucopareia nearly disappeared from Eurasia in the 1960s—only a single individual was observed in Japan in 1964 and a few more throughout the 1970s. A recovery programme supported by the Japanese Association for Wild Geese Protection, the Yagiyama Zoo, U.S. Fish and Wildlife Service and the

Russian Academy of Sciences was launched in 1983, and several captive-breeding programmes were initiated in Japan and the Russian Federation. A total of 551 captive-bred birds from the Russian Federation were released on Ekarma Island between 1995 and 2010, and monitoring showed an increase from 161 birds in 2010–2011 to 402 birds in 2012–2013. The number of wintering geese in Japan increased to over 100 specimens in 2010–2011 and reached over 1,700 individuals in 2014–2015, the most recent assessment.

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

At the international level, the vast majority of legal trade in *Branta canadensis leucopareia* has been for captive breeding and re-introduction purposes, with mostly captive-bred sources. A total of 131 live specimens (and six eggs) have been traded internationally for captive breeding programmes since the subspecies listing in 1975. Among these, 18 individuals were wild-caught, whereas the remaining birds were recorded as captive-bred. During the same period, an additional 21 wild-caught birds have been traded for either commercial or hunting purposes.

Most of the international trade in *B. canadensis leucopareia* was motivated by a longer-term goal of breeding birds in captivity to re-introduce them to additional Aleutian Islands and to Asia. According to the CITES Trade Database, 38 live specimens have been transported from the USA to the Russian Federation, 15 from the USA to Japan and 66 live specimens and six eggs from Japan to the Russian Federation since 1975. An additional 12 birds have been transported for captive breeding purposes from Canada to Germany. Moreover, six live specimens were traded for zoological purposes from Canada to Hong Kong SAR in 1981.

Minimal reported trade has been for commercial purposes, including two live captive-bred individuals transported from Canada to the USA in 1995. Trade records also show shipments of six additional live specimens from the Netherlands to South Africa. Some wild birds have been transported as hunting trophies, including 19 wild-caught trophies exported from the USA to Taiwan POC in 1996 and one wild-caught trophy traded from Canada to the USA in 2018. One additional wild-caught body was traded from Canada to the USA in 2000.

In addition, there are a few records of international trade in parts and derivatives of *B. canadensis leucopareia* occurring over the years. In 2000, two pre-Convention unspecified parts were transported from the USA to France and back to the USA for exhibition purposes. In 2010, four pre-Convention parts were transported from France to Switzerland for personal purposes. Finally, one wild goose skeleton was traded for scientific purposes from the USA to Canada in 2012.

According to the EU-TWIX Database, a total of 19 live *Branta canadensis* specimens (source: unknown) were seized in 2017, 2019 and 2020 (15 in 2017, two in 2019 and two in 2020) in France.

No illegal trade appears to be occurring at the subspecies level for *B. canadensis leucopareia*. The seizure of one feather of this subspecies from Mexico to the USA in 2007 has been reported by the latter.

The Mexico Scientific Authority has stated that no requests for the export, import or re-export of *B. canadensis leucopareia* have been made from 2005 to 2022. Similarly, the United States Fish and Wildlife Service's Migratory Birds programme has noted that no information was available by state for either legal or illegal trade.

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**

Most of the populations of the Aleutian Cackling Goose inhabit the USA, where domestic laws regulate hunting of the subspecies. These laws provide a safeguard against overharvesting, and therefore any increased risk to the subspecies from international trade as a result of the proposed transfer to Appendix II is not anticipated.

Export quota or other special measure

There are no current quotas in place for this subspecies.

Additional information

Threats

The introduction of Arctic Foxes *Alopex lagopus* and, to a lesser extent, Norway Rats *Rattus norvegicus* to many North Pacific Islands starting from the 1750s resulted in the near extinction of the Aleutian Cackling Goose in the 20th century due to predation of eggs and juveniles (NatureServe, 2022). To meet demand for fur, the introduction of foxes on goose breeding islands severely intensified in the early 1900s, and over 450 islands had introduced fox populations by the 1930s. As a result, the populations of the Aleutian Cackling Goose plummeted and a survey of the Aleutian Islands from the 1930s reported only a few pairs remaining. Over the years, remnant

populations of the subspecies were rediscovered on Buldir Island in 1962, on Kiliktagik Island in 1979 and on Chagulak Island in 1982. Intensive management and conservation efforts have been vital for the survival of the subspecies, which now numbers over 160,000 individuals within the Aleutian Islands alone. However, Arctic and Red Foxes *Vulpes vulpes* remain on many of the islands and predation continues to limit the re-establishment of the subspecies across their historic range.

Nowadays, primary threats include habitat alteration in wintering and migration areas (mostly caused by urbanisation and shifts in agricultural practices), continued predation from invasive species, and infectious disease (e.g., avian cholera). Droughts in California are perceived to become a serious threat for the subspecies, alongside other impacts from climate change. The Semidi Island population faces additional threats (largely unknown due to lack of research), reflected in the poor survival rate of young birds.

Current harvest levels, as they are generally considered to be low, no longer pose a serious threat to the subspecies. However, incidental take by private entities may continue to have negative impacts on population size.

Conservation, management and legislation

Conservation:

In Alaska, all breeding locations of the subspecies are protected through the USFWS National Wildlife Refuge System, in which conserving of the Aleutian Cackling Goose and their nesting habitats remains a priority. In California, key wintering sites are protected in the San Joaquin National Wildlife Refuge and other federal lands. In Oregon, Nestucca National Wildlife Refuge offers protected wintering habitat for the Semidi Island region, while other migration staging areas and wintering locations are protected by the Bureau of Land Management or exist on private lands.

Management:

The first fox-removal operations from breeding islands occupied by the subspecies started in the 1940s. The first Aleutian Goose Recovery Plan was developed in 1979, with initial efforts focusing on securing breeding habitat and re-establishing breeding colonies. As these efforts enabled the goose's recovery, the subspecies was delisted from the U.S. Endangered Species Act in 2001. Altogether, the U.S. Fish and Wildlife Service has managed and restored the Aleutian Cackling Goose populations through (i) fox removal, (ii) hunting closures and harvest strategies, (iii) management of overwintering and migratory staging areas, (iv) disease control, and (v) captive breeding and re-introductions.

In California, as numbers of the Aleutian Cackling Goose increased, conflict has arisen between landowners and local populations due to the extent of crop damage caused by the subspecies during spring staging (from February to April) near Crescent City. As a result, some landowners started harassing the geese and working groups developed plans to limit goose grazing on public lands in the 1990s. Permitted hunting on private lands (with a limit of up to 10 per day) has also been introduced in order to shift goose grazing to public lands.

Overall, current management efforts include continued monitoring surveys, managing a harvest strategy, and addressing complaints from the agricultural community.

Legislation:

Within the USA, the Cackling Goose (*Branta hutchinsii*) is protected under the Migratory Bird Treaty Act (U.S. Fish & Wildlife Service 2020b), which prohibits the pursuing, hunting, taking, capturing, killing, possessing, selling, purchasing, bartering, importing, exporting, or transporting of the species, unless a federal permit is issued by the U.S. Department of the Interior.

In Japan, hunting of *Branta canadensis leucopareia* has not been allowed since 1947 according to the Wildlife Protection, Control, and Hunting Management Act. It is classified as a "rare wildlife species" under this law, and therefore requires protection at both national and international levels. Additionally, the subspecies is protected under the Act on Conservation of Endangered Species of Wild Fauna and Flora (ACES), and the hunting, gathering, killing, domestic trade, international trade, display and advertisement for the purpose of sale or distribution is prohibited.

At the international level, *Branta canadensis leucopareia* has been listed in CITES Appendix I since 1975.

Captive breeding

Captive breeding operations and re-introductions have taken place on Aleutian Islands, Alaska, and Ekarma Island, Russian Federation. A total of 2,500 birds were released by the year 1991 on four fox-free Alaskan Islands, however Bald Eagle *Haliaeetus leucocephalus* predation limited the success of such re-establishment operations. The pairing of captive-bred birds from Buldir Island with wild-caught birds led to a higher level of success with re-

introductions. Eventually, the subspecies was re-introduced in Alaska on Agattu, Nizki-Alaid, and Little Kiska Islands and possibly on Amchitka, Amukta, Skagul, and Yunaska Islands.

In the Russian Federation, the subspecies has been bred in captivity at the Kamchatka Institute of Ecology and Nature Management. Between 1995 and 2001, a total of 551 captive-bred geese were released on Ekarma Island.

Implementation challenges (including similar species)

Branta sandvicensis (Hawaiian goose) is listed in Appendix I. The two taxa are very distinct and therefore easily distinguishable.

The Aleutian Cackling Goose is similar to other subspecies within this species complex, none of which are included in the appendices.

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

Harvesting in the USA is well regulated by domestic measures, thus, a transfer to Appendix II will not allow international trade to threaten the population. Moreover, management of the Aleutian Cackling Goose and their habitat will continue in the USA regardless of CITES protection levels due to (i) the Migratory Bird Treaty Act and (ii) the fact that the subspecies is managed as a migratory game species, and therefore their key wintering, staging, and breeding habitats will remain protected on public lands.

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Inclusion of the White-rumped Shama *Kittacincla malabarica* in Appendix II

Proponents: Malaysia, Singapore

Summary: The White-rumped Shama (known as *Copsychus malabaricus* under the current CITES taxonomic reference for birds) is a widespread Asian songbird native to 15 countries: Bangladesh, Bhutan, Brunei Darussalam, Cambodia, China, India, Indonesia, Lao People's Democratic Republic (PDR), Malaysia, Myanmar, Nepal, Singapore, Sri Lanka, Thailand, and Viet Nam. The species comprises multiple genetically distinct subspecies and subpopulations, with references recognising 14–17 subspecies and new taxonomic research continuing to uncover more genetically distinct subspecies.

The species has an extremely large range, with an estimated extent of occurrence of some 14 million km², and is described as common in at least parts of its range. It was evaluated by BirdLife International as Least Concern on the IUCN Red List in 2020, although its population overall is suspected to be in decline owing to ongoing habitat destruction and collection for the bird trade. Where not impacted by trapping, the population can occur at very high densities, and it has been shown to be adaptable to some level of habitat disturbance. There are no population estimates for the species across its range, but it is thought to number in the hundreds of thousands.

The species is believed to be threatened in parts of its range where it is under pressure from harvest and it is recognised among the highest priorities for action by the IUCN SSC Asian Songbird Trade Specialist Group.

Owing to its remarkable singing ability, the White-rumped Shama is one of the most prized and valuable species in the South-East Asian cagebird trade and among the most important species used in singing competitions, particularly in Indonesia. Outside South-East Asia, only small numbers of White-rumped Shammas have been observed for sale in Hong Kong SAR and India. Most of the trade historically has been domestic. However, as populations around the main centres of demand decrease and become more difficult to source (most importantly in Indonesia, where the species is locally depleted in places where it is most popular), trappers and traders appear to have turned to sources further afield.

In 2019, over three million White-rumped Shammas were estimated to be kept in captivity across the island of Java in Indonesia alone. It is not known what proportion of these is wild-caught, nor how many have been imported although there is recent evidence of the illicit movement of birds across international borders. In Viet Nam, a songbird consumer study found that the species is one of the most desired birds sought by bird keepers, and that most owned or preferred to keep wild-caught birds, which are thought to be mostly sourced from within the country. Alternatively, some traders have stated that captive-bred White-rumped Shammas are more desirable because of their longer lifespans and greater compatibility with life in a cage.

Captive breeding of the species appears to be active in some South-East Asian countries. To date, there are 52 captive breeders in Peninsular Malaysia who hold a commercial captive breeding permit for this species, and both small- and large-scale commercial captive breeding of White-rumped Shammas is ongoing in Indonesia, but the lack of published records makes it impossible to determine its extent.

Combined data from snapshot surveys across Indonesia, Malaysia, Singapore, Thailand, and Viet Nam, carried out between 2007 and 2018, found a total of 8,271 White-rumped Shammas openly for sale in physical local bird markets on the days surveyed. Another 917 were found for sale online in six snapshot internet trade studies in Indonesia, Malaysia, and Thailand between 2016 and 2018.

From 1997–2003, the White-rumped Shama was listed in Annex D of the European Union Wildlife Trade Regulations (EU WTR). It was delisted in 2003, as specimens were not imported into the EU in numbers substantial enough to warrant monitoring. During that period, almost 1,000 live individuals were reported in trade annually from South-East and East Asian countries to the EU. Around 65% of the transactions were exports by China and Indonesia, and these accounted for 50% and almost 30% of the traded individuals respectively. In 2005, the EU implemented a ban on wild bird imports to prevent the spread of avian influenza and other diseases, which is still in place.

Malaysia's regulations allow capture and trade of the species under licence, as seems to be the case in most South-East Asian countries except for Singapore, Cambodia, and Thailand, where trade is prohibited unless individuals are captive-bred. Despite this, in recent years a growing number of smuggled bird shipments containing White-rumped Shammas have been intercepted mainly from Malaysia into Indonesia, where they reach high prices in the cagebird market and are expensive compared to other species. Between 2008 and 2018, 432 seizures of White-rumped Shammas were recorded in data from Indonesia, Malaysia, Singapore, Thailand, and Viet Nam, including more than 15,000 birds with two-thirds of these in the period 2014–2018. Some 12% of recorded seizure incidents involved international trade and accounted for more than two-thirds (over 10,000) of all the White-rumped Shammas seized in this period. Likewise, TRAFFIC's data include 615 seizure records of more than 30,000 White-rumped Shammas between 2009–2022, with around a third being seized after 2018. These seizures were concentrated in the South-East Asian region with at least 13% of the total incidents having involved international trafficking, mainly from Malaysia to Indonesia.

The proposal to list the species in Appendix II is based partly on the volume of seized individuals, including what the proponents perceive as being rising incidences of international smuggling.

Analysis: The White-rumped Shama was assessed as Least Concern on the IUCN Red List in 2020. The species is not considered threatened and is described as common in significant parts of its large range. However, there is evidence that populations are locally depleted in places in South-East Asia, where it is popular as a caged bird. Domestic trade appears to be the most significant driver of harvest, but the species is also traded internationally as indicated by seizures particularly from Malaysia to Indonesia, which appear to be increasing due to a locally depleted population. At this time, the impact of collection for international trade on the species in its range other than in Malaysia and, in particular, Indonesia is unclear. Some trade is said to be in captive-bred individuals, although it is not known what proportion of the overall trade this is. Available information on status and trends of the wild population and on impacts of collection for trade do not support a conclusion that the White-rumped Shama meets the criteria for inclusion in Appendix II set out in 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

"Kittacincla malabarica". Copsychus malabaricus (Scopoli, 1786) is the taxonomic name that should be used regarding the CITES Standard Nomenclature: Dickinson, E.C. (ed.) (2003). The Howard and Moore Complete Checklist of the Birds of the World. Revised and enlarged 3rd Edition. Christopher Helm, London. pp.1039 in combination with Dickinson, E.C. (2005). Corrigenda 4 (2nd June 2005) to Howard & Moore Edition 3 (2003).

Synonyms: *Copsychus malabaricus* (Scopoli, 1786), *Muscicapa malabarica* (Scopoli, 1786)

A CITES listing under a different or updated name is possible if the Proposal explicitly designates an updated nomenclature standard for adoption as part of the Proposal. However, this Proposal does not designate such a standard. Although transfer to the genus Kittacincla is likely to happen at CoP20 if/when adopting the Handbook of the Birds of the World–BirdLife International Illustrated Checklist of Birds of the World as the new bird nomenclature standard for CITES (intention to do so is noted in Document 84.1 on Nomenclature), currently it should be discussed as Copsychus malabaricus for CITES purposes (IUCN-SSC Asian Songbird Trade Specialist Group, in litt., 2022).

The taxonomy of this species is complex. *Kittacincla malabarica* (syn. *Copsychus malabaricus*) is a polytypic species complex comprising multiple genetically distinct subspecies and subpopulations, with references

recognising 14–17 subspecies and new taxonomic research continuing to uncover more genetically distinct subspecies.

Range

Bangladesh, Bhutan, Brunei Darussalam, Cambodia, China, India, Indonesia, Lao PDR, Malaysia, Myanmar, Nepal, Singapore, Sri Lanka, Thailand, Viet Nam.

The species has been introduced to the Hawaiian Islands of O'ahu and Kaua'i (Roberts et al., 2020).

IUCN Global Category

Least Concern (assessed 2021, ver. 3.1)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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

Owing to its remarkable singing ability, the White-rumped Shama is one of the most sought-after and valuable species in the South-East Asian cagebird trade and among the most important species used in bird singing competitions.

Population status and trends

There are no population estimates for the species across its range, but it is believed to be large as the species is described as common in at least some parts of its range, with well above 10,000 individuals. Therefore, the species was evaluated by the IUCN as Least Concern in 2020. The population is suspected to be in decline owing to ongoing habitat destruction and the bird trade.

The species has an extremely large range, with an estimated extent of occurrence (using a minimum convex polygon) of 13,900,000 km² (BirdLife International, 2021). The global population size has not been quantified, but it is believed to be large as the species is described as common in at least parts of its range (Robson, 2002). Although it is difficult to provide even a rough estimate, it is reasonable to suppose that the population numbers the hundreds of thousands (Fernando and Berryman, in litt., 2022). Where not impacted by trapping, the population can occur at very high densities, and it has shown to be adaptable to some level of habitat disturbance (Berryman, in litt., 2022).

The species is considered common in Peninsular Malaysia with the main habitat mostly in lowland primary forest. However, based on wildlife inventory and observation data, numbers are declining and there has been a lower number of captured individuals during mist-netting activity. *It is also becoming increasingly scarce and difficult to find outside protected areas in Peninsular Malaysia (Berryman, in litt., 2022).*

In Indonesia, the species is believed to have become rare in most locations. On Java, the species was rare in the 1980s from trapping for trade and was thought to have been trapped to local extinction in the wild prior to 1997. In Sumatra, field observations of birds where the species was previously reliably recorded have dwindled.

The population of White-rumped Shamas in Singapore is unknown but it has shown an almost fourfold increase from 10 years of Annual Bird Census data, which suggests an upwards trend. *Singapore populations were found to consist of a mosaic of both native populations and escaped cagebirds of mostly Peninsular Malaysian origin, indicating that inadvertent reintroductions of caged White-rumped Shamas have led to the recovery of a local population that was nearly extinct (Ng et al., 2017). Surveys in Pulau Ubin, Singapore, which is an offshore island almost fully forested and not easily accessible, showed that it is one of the commonest bird species observed (Berryman, in litt., 2022).*

The species is considered to be common in Thailand and Viet Nam, but even there declines have been recorded, *for Thailand particularly in the southern provinces from Chumphon province southward, where it is almost nowhere to be found outside protected areas (Chng and Krishnasamy in litt., 2022). In Brunei, observations from local birdwatchers and field researchers indicate that the species is scarce and confined to the interior forest of Brunei Darussalam (Chng and Krishnasamy, in litt., 2022).*

Some island endemic subpopulations are believed to have only a few surviving individuals. Subspecies *hypoliza*, *barboursi* and *opisthochra* are, for example, most likely extinct in the wild, whilst subspecies *melanurus* is at critical levels on Siberut, and potentially extinct on other islands. *K. malabaricus ngae* is one of the recently described subspecies by Wu and Rheindt (2022) and it is among the most threatened taxa (Berryman, in litt., 2022).

In the Asian Songbird Trade Specialist Group (ASTSG) 2019 meeting, a discussion on the status of White-rumped Shama subspecies concluded that the only populations that remained of Least Concern were the South Asian

populations. The populations in peninsular South-East Asia were near threatened, and all others considered vulnerable, endangered, or possibly extinct in the wild. *Several distinctive subspecies have been identified by the ASTSG as separate "conservation units" (IUCN-SSC Asian Songbird Trade Specialist Group, in litt., 2022).*

Trade

The species is heavily trapped across Indonesia for the caged bird trade, specifically to supply bird song contests. In 2019, over three million White-rumped Shamas were estimated to be kept in captivity across Java alone. The proportion of wild-caught vs captive-bred individuals is not clear within the number of cagebirds recorded, nor if the individuals are acquired locally or imported. *The species is also popular in trade in Brunei Darussalam, Malaysia, and Singapore (Chng and Krishnasamy in litt., 2022).* In Viet Nam, a songbird consumer study found that the species is one of the most sought after by bird keepers, and that most owned or preferred to keep wild-caught birds, which are mostly sourced from within the country. *However, some traders have stated that captive-bred White-rumped Shamas are more desirable because of their longer lifespans (maximum longevity of seven years in the wild; Human Ageing Genetic Resources, 2015) and greater compatibility with life in a cage (wild-caught birds were said to remain silent during their first few months in captivity).*

Trade has been identified as the most urgent and pressing threat facing White-rumped Shamas, particularly for South-East Asian populations. Outside South-East Asia, small numbers of White-rumped Shamas have been observed for sale in Hong Kong SAR and India. Most of the trade historically has been domestic. However, as populations around the main centres of demand decrease and become more difficult to source, trappers and traders turn to sources further afield. *This is particularly the case in Indonesia, where the species is locally depleted in places where it is most popular (IUCN-SSC Asian Songbird Trade Specialist Group, in litt., 2022).*

Combined data from surveys across Indonesia, Malaysia, Singapore, Thailand, and Viet Nam, carried out between 2007 and 2018, found a total of 8,271 White-rumped Shamas openly for sale in local physical bird markets. Another 917 were found for sale online in six snapshot internet trade studies in Indonesia, Malaysia, and Thailand between 2016 and 2018 (Leupen et al., 2018).

From 1997–2003, the White-rumped Shama was listed in Annex D of the European Union Wildlife Trade Regulations (EU WTR). *According to the Commission Regulation (EC) No 1497/2003 of 18 August 2003 amending Council Regulation (EC) No 338/97, this and other wild species were deleted from Annex D because at the time it had been established that "a number of species currently included in Annex D were not imported into the Community in such numbers as to warrant monitoring and that these species should be deleted from that Annex".*

Due to being included in Annex D of the EU WTR, international trade records for White-rumped Shama are available from the CITES Trade Database from that time despite them not being listed in the Appendices. According to these data, 5,768 live individuals were imported by the EU between 1998–2004, mostly from South-East and East Asian countries. In October 2005, the EU implemented a ban on wild bird imports as a policy measure to prevent the spread of avian influenza and other diseases, which is still in place.

According to the CITES Trade Database, from 1998–2004, almost 1,000 live individuals were annually imported by the EU, virtually all wild-sourced and traded for commercial purposes. Around 65% of the transactions were exports by China and Indonesia, and these accounted for 50% and almost 30% of the traded individuals respectively. Only records of imports of this species to the EU are available as it was not listed in CITES.

A review of songbirds led by Species360 and the University of Southern Denmark (See AC31 Doc. 30 and AC31 Inf. 12) covering both legal and illegal trade data from 2015–2020, classifies the species as one with "high" volumes of trade in the EU and indicates that there have been wild-sourced specimens entering the EU after 2006, but also that the species is bred in captivity within the EU. In this review, domestic trade for the species was classified as "extreme", while international trade was considered "high" (the categories being low, moderate, high, and extreme). It also considers that trade, either international or domestic, is affecting the sustainability of the species or a particular population and classifies the combination of domestic and international trade volumes relative to the species population size, as "extreme". Trade routes identified are Malaysia to Indonesia and Indonesia to the EU.

Illegal trade and seizures

In recent years a growing number of smuggled bird shipments containing White-rumped Shamas have been intercepted from Malaysia into Indonesia. According to information from traders, demand for birds in Indonesian Borneo (Kalimantan) has resulted in the smuggling of 6,000 birds a month, including White-rumped Shamas from Malaysian Borneo to Indonesia.

A Report from TRAFFIC focussing on Oriental Magpie-robins Copsychus saularis pointed out that these are often smuggled in the same shipments as White-rumped Shamas, but the latter reach higher prices in the cagebird market, as does Straw-headed Bulbul Pycnonotus zeylanicus, and are considered to be expensive songbirds in comparison to others in Indonesia (Irham et al., 2020, Chng et al., 2021).

Between 2008–2018, 432 seizures were recorded from Indonesia, Malaysia, Singapore, Thailand, and Viet Nam, involving 15,480 birds; 67% of these seizures occurred between 2014–2018 (Leupen et al., 2018). Of all recorded seizure incidents, which were extracted from open-source media and obtained from NGOs and from the governments of Malaysia and Thailand, 12% involved international trade and accounted for 67% (10,376) of all White-rumped Shamans seized in this period (Leupen et al., 2018).

The Supporting Statement refers to a total of 615 White-rumped Shama seizure incidents recorded by TRAFFIC between January 2009 and May 2022, aggregated data from the Department of Wildlife and National Parks Peninsular Malaysia (PERHILITAN) for 2011–2017, and aggregated data from the Thailand government for 2010–2018. These seizures were concentrated in the South-East Asian region with the highest seizures in Malaysia (309), followed by Thailand (223), Indonesia (71), Singapore (8), Cambodia (3), and Viet Nam (1). A total of 32,133 White-rumped Shamans was confiscated. Seizure incidents and numbers peaked in 2016. The number of birds seized shows an upward trend after a decline in 2018, with 31.8% of birds seized after 2018. Based on this seizure data, at least 78 (13%) of the total incidents involved international smuggling.

The USA did not report trade—either legal or illegal—in the species in the LEMIS Database in the period between 2008–2020.

The proposal to list the species in Appendix II is partly justified by the proponents by the volume of individuals being smuggled, including what appears to be a rising case of international smuggling.

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**
- B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved**

Additional information

Threats

Besides the species being overexploited for the cagebird trade in some areas in South-East Asia, it is also threatened by extensive deforestation occurring throughout its range (Roberts et al., 2020; Global Forest Watch 2021). However, the White-rumped Shama has shown to be adaptable to some level of habitat disturbance (Berryman, in litt., 2022). Habitat loss and degradation exacerbate losses in the wild populations, as fragmentation increases the accessibility to hunters (Berryman, in litt., 2022).

Conservation, management and legislation

There are conservation breeding projects occurring in the islands of West Sumatra to protect, breed and reintroduce the endemic White-rumped Shama subspecies found there.

Ex-situ breeding programmes for K. malabarica subspecies hypoliza, opisthochra, melanurus, and barbouri are currently underway (IUCN-SSC Asian Songbird Trade Specialist Group, in litt., 2020). In Singapore, a captive-breeding programme has been called for to supply the demand for singing cagebirds allowing restocking of wild populations (Roberts et al., 2020).

Malaysia's regulations protect the species but do allow capture and trade of the species through licensing, as seems to be the case in most South-East Asian countries except for Singapore, Cambodia, and Thailand, where trade is prohibited unless individuals are captive-bred (Leupen et al., 2018).

Captive breeding

Some captive-breeding already takes place in range countries and involves both independent breeders and large-scale, multi-species bird farms. However, the lack of published records makes it impossible to determine the extent to which White-rumped Shamans are being bred, and how much of this is legal.

To date, there are 52 captive breeders throughout Peninsular Malaysia who hold a commercial captive breeding permit for this species, and in Indonesia, both small- and large-scale captive breeding of White-rumped Shama is ongoing (Jepson et al., 2011).

The Supporting Statement refers to various pros and cons of wild-caught vs captive-bred White-rumped Shamans, resulting in varying preferences amongst buyers and keepers. In Indonesia, there seems to be a general preference for wild-caught over captive-bred songbirds due to their apparent superior singing abilities, and the former fetch a higher price in the markets (Burivalova et al., 2017). On the other hand, some traders have stated that captive-bred White-rumped Shamans are more desirable because of their longer lifespans (maximum longevity in the wild is reported as seven years; Human Ageing Genetic Resources, 2015) and greater compatibility with life in a cage, and claim that wild-caught birds remain silent during their first few months in captivity.

Species360 indicates that wild-caught specimens are the primary source of White-rumped Shamans in trade, and categorises the difficulty of breeding this species as “normal”. It points out that there are conservation breeding programmes in place (See AC31 Inf. 12).

Implementation challenges (including similar species)

A foreseen challenge would be preventing the laundering of specimens as captive-bred to bypass CITES compliance.

Studies on the species’ wild populations would need to be developed and kept updated for range countries to be able to formulate non-detriment findings (NDFs) to comply with CITES regulations.

Potential risk(s) of a listing

Data show that the species is being illegally trapped and traded within range countries despite it being possible to trade it legally through permits and licences from most of them. There needs to be an effort to work out why current harvest and trade controls are being circumvented and call for co-operation between trading states, which are also the species’ range States.

Potential benefit(s) of listing for trade regulation

A CITES listing implies the mandatory recording of international trade, which would provide more information on trade levels, source and purpose, and would support range States in making informed decisions about trade regulations and the need of specific controls for the species being smuggled in rising volumes.

Additionally, information on the species’ wild populations would need to be produced, available and updated in order for range countries to be able to formulate NDFs as needed, and thus, the knowledge of the species would increase and the sustainability of its trade would be promoted.

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Transfer of Straw-headed Bulbul *Pycnonotus zeylanicus* from Appendix II to Appendix I

Proponents: Malaysia, Singapore, United States of America

Summary: The Straw-headed Bulbul *Pycnonotus zeylanicus* is a large, non-migratory bird found in Singapore, Malaysia, Indonesia and Brunei Darussalam inhabiting lowland successional habitats bordering rivers, streams and marshes, usually adjacent to broadleaf evergreen forest and secondary growth. Due to exploitation for the songbird trade and habitat loss, *P. zeylanicus* was assessed by BirdLife International as Critically Endangered in 2018 and is currently listed on Tier 1 of the IUCN SSC Asian Songbird Trade Specialist Group's priority taxa list (which includes those species considered to be the most threatened from trade) as a conservation priority. The species was included in CITES Appendix II in 1997.

At present, the wild population is small, ranging from 600 to 1,700 mature individuals. According to the IUCN Red List assessment, a marked decline is ongoing and has probably exceeded 80% over the past three generations (or 15 years). The largest and only stable population is found in Singapore (200–500 mature individuals). Little information exists on population structure.

The species has been subject to widespread extirpations throughout its entire range within the last 30 years (including from Java and Borneo (Indonesia), Myanmar and Thailand, and in various sites in Peninsular Malaysia). Logging and development, as well as change in land-use for agricultural plantations, are causing habitat loss throughout its range. Most of the secondary forest and woodland where *P. zeylanicus* occurs does not fall within protected areas, and in many cases has been cleared.

The primary threat to the species is trapping for the caged songbird trade, with specimens being moved mainly within and between South-East Asian countries. Although some captive breeding is known, it has been reported that wild-caught specimens are considered superior songsters and can fetch higher prices, so that captive-breeding does not appear to be alleviating demand for wild-caught birds. Following its original listing in 1997 the CITES Trade Database has registered 704 live birds in trade of which only three were declared as captive-bred rather than wild-caught. Only 46 live specimens have been reported in trade since 2000. A decline in the wild population is suspected to be the main reason for a decrease in availability in the market. Its market value has increased markedly over the past three decades (from USD20 in 1987 to over USD900 in 2018).

Instances of illegal trade have been registered over the past 20 years including incidents in Malaysia, Thailand, and Indonesia.

Analysis: *Pycnonotus zeylanicus* has been assessed as Critically Endangered on the IUCN Red List. Remaining populations are small and declining due to exploitation for the caged songbird trade as well as habitat loss and degradation. The species is now limited to a fraction of its historic range. Therefore, *P. zeylanicus* appears to meet the biological criteria for inclusion in Appendix I of Res. Conf. 9.24 (Rev. CoP17). Although the relative extent of domestic and international trade is not clear, there is sufficient evidence to conclude that the species is affected by 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.

CITES Background

Listed in CITES Appendix II since 1997.

Taxonomy

Pycnonotus zeylanicus (Gmelin, 1789)

Standard nomenclature: Dickinson, E.C. (Ed.) (2003). *The Howard and Moore Complete Checklist of the Birds of the World*. Revised and enlarged 3rd Edition. Christopher Helm, London. 1039 pp. in combination with Dickinson, E.C. (2005). Corrigenda 4 (02.06.2005) to Howard & Moore Edition 3 2003. (Available on the CITES website).

Range

According to the IUCN Red List assessment:

Extant (resident): Brunei Darussalam; Indonesia; Malaysia; Myanmar

Extant (breeding): Singapore

It is now considered with some certainty to be extinct in Thailand and likely Myanmar (BirdLife International, 2001).

IUCN Global Category

Critically Endangered A2cd+4cd (assessed 2020, ver. 3.1)

Biological criteria for inclusion in Appendix I**A) Small wild population**

The latest IUCN Red List assessment for the species revised the population estimate in 2020 based on an appraisal of the areas where populations were then persisting. It places the number in the range of 1,000–2,499 individuals, considered to represent 667–1,667 mature individuals.

The Singapore population is estimated to comprise 200–500 mature individuals. Based on wildlife inventory checklist data, the species can be found throughout Peninsular Malaysia, but mostly in very remote areas in protected areas and forest reserves.

Systematic field surveys undertaken by Chiok et al. (2021) in Singapore estimated a population of 388–758 Straw-headed Bulbuls nationwide, which potentially comprises 22.9–57.3% of the global wild population. In spite of its apparently secure status in Singapore, the species remains susceptible to local and foreign trapping pressures (Chiok et al., 2021).

Historically, it was widespread and common throughout its range, which extended from southernmost Myanmar and Thailand through Peninsular Malaysia to the islands of Borneo, Sumatra, and Java. There is evidence that the species has been extirpated from Thailand, Myanmar, and Java, Sumatra, Nias, and Sipora in Indonesia, and is now limited to Singapore, parts of Malaysia, and remote parts of Kalimantan in Indonesia.

The Straw-headed Bulbul has been extirpated from Thailand, Myanmar, and Java. On Sumatra, its status is unknown. There are records from the last five years but its population size must be very small at best and may have now been extirpated at worst (Berryman, in litt., 2022).

B) Restricted area of distribution

P. zeylanicus occurs in South-East Asia. Historically, its range extended from southernmost Myanmar and Thailand through Peninsular Malaysia to the islands of Borneo, Sumatra, Java. The species is now limited to Singapore, parts of Malaysia (Peninsular Malaysia, Sarawak and Sabah), remote parts of Kalimantan in Indonesia. Some individuals have been observed in Brunei Darussalam, but population numbers remain unknown. There is evidence that the species has been extirpated from Thailand, Myanmar, and Java, Nias and Sipora in Indonesia, and on Sumatra its status is unknown but may have now been extirpated (Berryman, in litt., 2022). However, the species' area of distribution is not restricted.

C) Decline in number of wild individuals

The IUCN Red List assessment, published 2021, reported the population decline is rapid and likely ongoing, estimated to exceed 80% over three generations (15 years). Due to the extremely rapid decline, the Straw-headed Bulbul has been listed as Critically Endangered since 2018, only two years after it was reclassified from Vulnerable to Endangered in 2016.

Singapore has the largest known population globally. However, Malaysia may well have a bigger population, but it has not been confirmed (Berryman, in litt., 2022). There are small remaining populations in parts of Peninsular Malaysia, Sarawak and Sabah. In Indonesia, very few individuals are believed to remain in Java and it is considered as likely extinct, while in Kalimantan it was increasingly rare by the mid-1990s despite being "common" two decades previously. In Myanmar and Thailand, it seems to have been extirpated, and in Brunei Darussalam the population status is unknown.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Following its listing in Appendix II in 1997 and up until 2020, the CITES Trade Database has recorded the commercial trade in 704 live birds. All were declared as wild-caught, except for three individuals declared by Kuwait (the importer) as captive-bred. All exported birds reportedly originated from Malaysia and were imported by Indonesia, Netherlands, Singapore, Kuwait, and Taiwan POC; 93% of these took place prior to 2000. In the last two decades, trade in only 46 live birds was recorded in CITES annual reports.

The number of birds available at market has dropped dramatically in the past few years *presumably* due to declining populations, *which could also explain the species' decrease in international trade*. Temporal analyses of market price data have shown an increase in the bulbul's monetary value—from USD20 in 1987 to over USD900 in 2018—likely due to the population decline.

Poaching and illegal trade of the Straw-headed Bulbul have been reported over the past 20 years, as evidenced through seizures, arrests, convictions and market observations. An important component of this trade is domestic, but it has also crossed borders. Bird dealers in Medan, Indonesia, claimed that birds were being sourced from Malaysia, specifically Peninsular Malaysia and Sabah. Sourcing of birds from Malaysia due to population declines of the species in Indonesia has been reported since the early 1990s and is still ongoing, impacting the survival of this and other Asian songbird species.

For known illegal trade, a minimum of 61 Straw-headed Bultuls were confiscated from 2006–2021. *WiTIS holds records for a total of seven seizure incidents between 2014 and 2021, involving the confiscation of 19 individuals in Malaysia, Thailand (where it is now considered to be extinct) and Indonesia, but there was no indication of the proportion of these that involved international smuggling. Between 2008 and 2020, LEMIS registered four transactions involving seven Straw-headed Bultuls sourced from the wild and exported from Singapore to the USA for scientific purposes. No seizures of the species were recorded in this period either in LEMIS or in EU-TWIX.*

Additional information

Threats

The main threat that the Straw-headed Bulbul faces is trapping of wild birds for the caged songbird trade, compounded by historical and ongoing habitat loss.

Decreases in the extent of lowland forests, especially near rivers and other watercourses, have contributed to the decline, although the Straw-headed Bulbul is capable of tolerating some degree of habitat degradation in areas where it is not under significant hunting pressure. However, road developments in such areas increase their accessibility to trappers of Straw-headed Bultuls.

The species's strong preference for forested habitat near to or even edging larger freshwater bodies makes them particularly accessible as well as susceptible towards targeted trapping for trade (IUCN-SSC Asian Songbird Trade Specialist Group, in litt., 2022).

The quality of the bulbul's song makes it a very popular cage bird, which has resulted in extensive trapping for both domestic and international trade. Its lack of shyness and habit of roosting and nesting in easily accessible locations have compounded its vulnerability to trapping (BirdLife International, 2021).

Conservation, management and legislation

Range States have specific legislation to protect the species and most can apply fines for persons illegally trapping, selling, importing or exporting the species. In Peninsular Malaysia, the species is considered Totally Protected, however captive breeding is permitted through government regulation. There is a blanket ban on the live import of birds from Malaysia into Singapore due to concerns over avian influenza.

Programmes to manage wild populations do not exist in any range State, and much of the secondary forest and woodlands where the Straw-headed Bulbul occurs do not fall within protected areas.

Captive breeding

P. zeylanicus has been bred in captivity with some success in Singapore (*breeding has taken place in zoos for conservation purposes*). However, the process is not yet deemed sustainable, with protocols being developed to improve breeding knowledge such as hand-rearing of chicks.

To date, there are eight captive breeders in Peninsular Malaysia actively conducting commercial captive breeding activity for this species, for which they require a permit.

Bird breeding does not appear to be alleviating demand for wild-caught birds and some breeding operations in Kalimantan have been considered fronts for trading wild-caught birds (Rentschlar et al., 2018).

The European Association of Zoos and Aquaria (EAZA) has initiated an EAZA Ex-situ Programme (EEP) aimed at managing a conservation breeding programme for P. zeylanicus. This EEP is co-ordinated by Jurong Bird Park which also holds the majority of this captive population (IUCN-SSC Asian Songbird Trade Specialist Group, in litt., 2022). EAZA has published Best Practice Guidelines for the captive breeding of this species (Kumar, 2018) that demonstrate the challenges involved in this approach (IUCN-SSC Asian Songbird Trade Specialist Group, in litt., 2022).

Implementation challenges (including similar species)

No particular challenges foreseen.

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

The value of the species is already high because of its scarcity, which could be increased further from trade prohibition. However, there are facilities breeding the species, and these could be registered in CITES if they are interested in exporting under government regulation (which has not happened in the past according to the CITES Trade Database).

Potential benefit(s) for trade regulation of a transfer from Appendix II to I

Stricter protection for this Critically Endangered species and increased penalties for traffickers at least by some Parties. Potentially it may also have incentives to increase efforts to breed the species in captivity to meet demand.

Other comments

An Appendix I listing can contribute to the conservation of the Straw-headed Bulbul. Controlled international trade can support better local protection, including governments introducing stronger penalties for those trapping and trading birds illegally, and possibly introducing strengthened deterrent factors. These measures can also be adopted up by other importing countries besides range States.

However, parallel to a transfer to Appendix I, the Straw-headed Bulbul would also benefit from the development and adoption of Conservation and Sustainable Management Plans or Programmes by the range States, which do not seem to be in place currently.

This is a species with a very high chance of imminent extinction caused almost solely by trade (Berryman and Fernando, *in litt.*, 2022).

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<https://doi.org/10.1177/1940082917753909>. Viewed 27th July 2022.

Transfer of Short-tailed Albatross *Phoebastria albatrus* from Appendix I to Appendix II

Proponent: United States of America

Summary: The Short-tailed Albatross *Phoebastria albatrus* is a large seabird that historically bred on some 15 islands in Japan, Taiwan POC, and Hawaii, USA, with non-breeding individuals ranging through the North Pacific Rim and off the western coast of North America. Prolonged exploitation from 1887–1933, when around 5 million birds are believed to have been harvested for their feathers, resulted in near-extinction. By the 1950s the species only bred on Torishima Island, Japan, and on the Senkaku Islands, a group of islands whose sovereignty is disputed. The introduction of protective measures led to a slow recovery. Currently the population is estimated at over 7,000 individuals, with 80% breeding on Torishima and the remainder on the Senkaku Islands. In 2018–2019, just over 2,000 breeding birds were counted at Torishima Island with an additional 380 birds projected as breeding on the Senkaku Islands. The population is increasing at a yearly rate of approximately 9%. The species was included in Appendix I in 1975. It was classified by BirdLife as Vulnerable on the IUCN Red List in 2018. It is the only species of albatross included in the Appendices.

Major identified threats currently comprise natural events including habitat erosion and volcanism (Torishima is an active volcano), commercial fishing by-catch, climate change, and the possible impacts of introduced mammals. The species is protected by national legislation in Canada, China, Japan, Mexico, the Russian Federation, and the USA and is included in Annex 1 of the Agreement on the Conservation of Albatrosses and Petrels and in Appendix I of the Convention on the Conservation of Migratory Species of Wild Animals.

There are no indications of any current commercial demand for the species. The CITES Trade Database registers a total of 157 imports and six exports from 1975–2019, of which only 1% were for commercial purposes (these involved pre-Convention specimens).

This proposal results from the *Periodic Review of the Appendices* (Resolution Conf. 14.8 (Rev. CoP17)), undertaken by the CITES Animals Committee.

Analysis: Following near-extinction in the 1950s as a result of overexploitation the Short-tailed Albatross has undergone significant recovery. The population is still relatively small (although is increasing year on year) and the species remains classified as Vulnerable by BirdLife/IUCN (2018). There is currently no evidence that harvest for international trade is or may be a threat to its survival. Therefore, the species does not appear to meet either the biological or trade criteria for inclusion in Appendix I. Transfer of the species to Appendix II is in line with the Precautionary measures in Annex 4 of Res. Conf 9.24 (Rev. CoP17). Furthermore, the species is widely protected. This proposal is supported by the Animals Committee.

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

Phoebastria albatrus (Pallas, 1769).

Synonym: *Diomedea albatrus* (Pallas, 1769).

Range

The IUCN Red List Assessment states the following:

- Extant (seasonality uncertain): China; Korea, Republic of; Mexico
- Extant (resident): Canada; Japan; Russian Federation (Eastern Asian Russia); Taiwan POC; United States of America; United States Minor Outlying Islands

- Extant & Origin Uncertain (seasonality uncertain): Northern Mariana Islands; Philippines

IUCN Global Category

Vulnerable D2 (assessed 2018, ver. 3.1)

Biological criteria for inclusion in Appendix I

A) Small wild population

Population size is estimated as 7,365 individuals and, although significantly small compared to their historic status (i.e., <1% of historical numbers—Orben et al., 2021), is steadily increasing at a yearly rate of 8.9% (US Fish and Wildlife Service, 2022). Records of population growth for Short-tailed Albatross are consistent across studies (Senzaki et al., 2019).

B) Restricted area of distribution

Phoebastria albatrus is found in the island chains south of Japan (including Mukojima, Nakodojima, and Yomejima in the Ogasawara Islands), the Aleutian Islands of Alaska and into both the Bering Sea and eastern Chukchi Sea, the north-western Hawaiian Islands and along the western coast of Canada, the Baja Peninsula of Mexico, ranging within the continental shelf and slope regions of the North Pacific Rim. Their southern distribution limit is currently unknown but presumed to be along the North Equatorial Current. The distribution of breeding pairs extends from Torishima Island (84%) to the Senkaku Islands (16%), and to the Ogasawara Islands (<1%; US Fish and Wildlife Service, 2020).

C) Decline in number of wild individuals

As previously mentioned, populations of *P. albatrus* are steadily increasing at a yearly rate of 8.9%. The most recent estimate of the breeding population from 2018–2019 amounts to approximately 2,022 individuals at Torishima and a projected 380 breeders in the Senkaku Islands in the East China Sea.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Prolonged exploitation from 1887 to 1933, during which approximately 5 million birds were harvested for their feathers, resulted in the near-extinction of *P. albatrus*. The CITES Trade Database reports a total of 157 legal imports and six legal exports from 1975–2019; imports were from the then Soviet Union, the UK, the USA, Switzerland, Japan, Canada, and New Zealand, whereas exports were from Japan, Switzerland, and the USA. There was only one instance of apparent illegal trade, when Canada reported exports of 632 pre-Convention bones for scientific purposes to the USA in 2017, which were reported as sourced from seizures and/or confiscations by the USA in import records. WITIS reports a seizure from a taxidermist in the UK in 2018. There is no trade demand for *P. albatrus*, and no evidence that international trade is or may be a threat to the survival of this species.

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

There is no trade demand for *P. albatrus*, and no evidence that international trade is or may be a threat to the survival of this species.

Phoebastria albatrus is the only member of the order Procellariiformes currently included in the Appendices.

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

There are no current quotas in place for this species. Several international, national and state laws prohibit the taking of *P. albatrus* from the wild unless the purpose falls under the recovery of the species.

Additional information

Threats

The primary threats include natural events such as habitat erosion and volcanic activity (especially on Torishima), as well as interactions with a variety of commercial and artisanal fisheries, climate change, and mammalian predation in the colonies if non-native species are introduced. The following is a comprehensive list of potential threats to *Phoebastria albatrus*.

Commercial fishing by-catch

Fishing by-catch is observed sporadically, and most are juveniles (Orben et al., 2021). According to the U.S. Fish and Wildlife Service, there have been 18 documented mortalities of *P. albatrus* in fisheries in the Bering Sea in Alaska, the Sea of Okhotsk in the Russian Federation, within the North Pacific Ocean off Oregon, USA, and within the North Pacific Ocean off Japan. These deaths have occurred in line, pelagic and demersal fisheries (Flint, in litt., 2022). Many fisheries with which this species overlaps in range have no observer programme and it is unknown how many birds are killed each year (Flint, in litt., 2022). However, according to Orben et al. (2021), as little as a ~2.5% increase in annual fisheries mortality has the potential to slow population growth substantially.

Contaminants

- **Plastic ingestion:** Plastics have been discovered throughout age groups in nearly all albatross species. Evidence of plastic ingestion in *P. albatrus* has been recorded at Torishima Island, where chicks regurgitate plastic waste in the colony (Donnelly-Greenan et al., 2018). However, the impacts of plastic ingestion on the survival and population growth of *P. albatrus* are unknown.
- **Radiation:** An earthquake off the north-eastern coast of Japan in 2011 resulted in a tsunami that damaged the Fukushima Daiichi Nuclear Plant and the subsequent release of approximately 520 PBq of radiation. Roughly 80% of this has entered the North Pacific Ocean, where *P. albatrus* can be found. Although recent studies indicate that there are no detectable levels of radiation found in *P. albatrus*, adverse effects from the radiation on the species' food resources could possibly occur.
- **Organochlorines, pesticides and metals:** Contaminants such as organochlorine, mercury and other metals have been discovered in the eggs, feathers and blood of *P. albatrus*. Bioaccumulation and biomagnification of these substances could have adverse effects throughout the growth and development of *P. albatrus*. Studies have shown that these specific contaminants have resulted in a decreased immune system response in *P. albatrus*.
- **Oil spills:** The risk of oil spills has significantly increased throughout the North Pacific and northwards into the Arctic Ocean, where *P. albatrus* distributions are the most prevalent. A future threat that *P. albatrus* may face is the possibility of oil development around the Senkaku Islands off Japan.

Competition

Populations of the Black-footed Albatross *Phoebastria nigripes* have been reported competing over nesting sites with *P. albatrus*. The Laysan Albatross *Phoebastria immutabilis* is believed to be in competition with *P. albatrus* over marine foraging areas along the south-western coastline of North America, as *P. immutabilis* has been seen expanding into waters historically occupied by *P. albatrus* (Henry III et al., 2021).

Predation

Crow predation has been recorded for *P. albatrus*, but crows are not present on Torishima today. Shark, cat, dog, mouse, and rat predation has been recorded for other albatross species but has yet to be recorded for *P. albatrus*.

Disease and parasites

Overall, *P. albatrus* is susceptible to disease due to low numbers of breeding sites and population size. Currently there is no evidence of parasites infesting populations nor having an impact on the mortality of the species.

Conservation, management and legislation

The habitats of *P. albatrus* are conserved through five Canadian marine protected areas (i.e., the Race Rocks Proposed Marine Protected Area (1988), the Bowie Seamount Marine Protected Area (2008), the Gwaii Haanas National Marine Conservation Area Reserve and Haida Heritage Site (2010), the Hecate Strait/Queen Charlotte Sound Glass Sponge Reef Proposed Marine Protected Area (2017), and the Scott Islands Marine National Wildlife Area (2018)) and five national wildlife refuges/national monuments in Japan (i.e. National Wildlife Protection Area on Mukojima (1954), Natural Monument (1958), Special Natural Monument (1962), Torishima Island Natural Monument (1965), National Park and Marine Park in the Ogasawara (Bonin) Islands (1972)). The Papahānaumokuākea Marine National Monument in US waters, established in 2006 and expanded in 2016, protects 1.5 million km² of marine habitat surrounding the north-western Hawaiian Islands from commercial fishing and disturbance of nesting areas on land (Flint, in litt., 2022).

Management measures for *P. albatrus* consist of national and international recovery (i.e., the 2008 Short-tailed Albatross Recovery Plan; USFWS, 2020) and conservation plans. National plans have been created by Canada, Japan, and the USA. The following have legislation that restricts trade of the species, along with ongoing conservation measures: Canada, China, Japan, Mexico, Russian Federation, and the USA.

At the international level, *P. albatrus* is protected by (aside from CITES Appendix I listing): Agreement on the Conservation of Albatrosses and Petrels—Annex 1, Convention on the Conservation of Migratory Species of Wild Animals—Appendix I, and Protection of Birds & Their Environments—a multilateral agreement between the Government of the USA and the Governments of Japan, Mexico and the Russian Federation for the protection of migratory birds and birds in danger of extinction, and their environment.

Captive breeding

The Short-tailed Albatross Recovery Plan from 2008 entails the translocation of breeding populations from the Tsubamezaki colony to the Hatsunezaki site on Torishima, and from Torishima to Mukojima. Both translocations involved recorded playback of breeding sounds and decoys of *P. albatrus*, as well as hand-reared chicks that were artificially raised and released at adulthood.

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Transfer the population of Broad-snouted Caiman *Caiman latirostris* of Brazil from Appendix I to Appendix II

Proponent: Brazil

Summary: The Broad-snouted Caiman *Caiman latirostris* is native to Brazil, Argentina, Bolivia, Paraguay, and Uruguay where it occurs in the Paraná, Paraguay, São Francisco and Uruguay River basins. The species was included in Appendix I in 1975. In 1997 the population of Argentina was transferred to Appendix II in accordance with the Resolution on Ranching (now Res. Conf. 11.16 (Rev. CoP15)). The species was assessed by IUCN in 2019 and classified as Least Concern, on account of its wide range, ability to colonise anthropogenic environments, and apparently stable global population.

In Brazil, the species is found in the Cerrado, Caatinga, Atlantic Forest, and Pampas biomes, extending from the coastal areas of Rio Grande do Norte to the São Francisco and Paraná-Paraguay watersheds and reaching the Lagoa dos Patos and Lagoa Mirim, in the state of Rio Grande do Sul. This range extends over 2.7 million km² and comprises over 70% of the total distribution of the species. The area of occupancy is believed to exceed 20,000 km².

The diversity and extent of habitats occupied by *C. latirostris* make it difficult to estimate population abundance accurately, but Brazil's population was estimated in 2016 to number over 400,000 based on the population density estimated for water bodies associated with a silvicultural landscape. The species remains widely distributed and abundant throughout much of its range. There are a number of reports of *C. latirostris* being found in urban and peri-urban areas, possibly indicating dispersal to new areas and suggesting an increase in the size of natural populations.

Although illegal hunting still occurs in some places, it is no longer identified as the major threat to this species, and illegal trade of skins has not been recently documented in Brazil. There is very limited export of captive-bred specimens from Brazil. According to the CITES Trade Database, from 2010 to 2020, Brazil reported exports of the species involving 101 skins from captive-bred individuals. Currently, there are five operating farms for *C. latirostris* in Brazil, only one of which is registered with CITES. It is not clear from the proposal what the long-term intention is in relation to trade in wild specimens of *C. latirostris*.

Precautionary measures included in the proposal are a "zero quota of ranched or harvested individuals", and the only form of management currently in place and proposed is farming. The proposal does not provide details on how long this quota is intended to be in place, nor if it would be lifted depending on results obtained through a nationwide monitoring programme. The quota is not an integral part of the proposal, so Brazil could decide to modify it or lift it at any time.

Brazil's objective for transfer of its population of the species to Appendix II is not clear, since trade from farms is already possible under the Appendix I listing. Brazil indicates it expects the social development of local communities through the management and conservation of Broad-snouted Caiman populations. However, it is not explained how a transference of the species to Appendix II would facilitate this.

Analysis: The *Caiman latirostris* population in Brazil no longer meets the biological criteria for being listed in Appendix I. It has a large population in Brazil in the hundreds of thousands, it is widespread in the country and the population is not declining. However, it is unclear what Precautionary measures are intended in line with Annex 4 of Res. Conf. 9.24 (Rev. CoP17). Brazil indicates in the supporting statement that it "will practise a zero quota for ranched or harvested individuals", but there is no formal export quota proposed for consideration with the proposed transfer to Appendix II. Brazil could include an export quota or other special measures to be approved at CoP19. Setting a zero export quota for wild harvested individuals for commercial purposes, as an integral part of the proposal to

transfer *C. latirostris* from Appendix I to Appendix II, would ensure that the relevant Precautionary measures 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.

CITES Background

The species was listed in Appendix I in 1975, and the Argentinian population was transferred to Appendix II in 1997.

Taxonomy

Caiman latirostris (Daudin, 1802).

CITES Standard reference: Wermuth, H., and Mertens, R. (1996). Schildkröte, Krokodile, Brückenechsen. xvii + 506 pp. Jena (Gustav Fischer Verlag), Reprint.

Range

Argentina, Bolivia, Brazil, Paraguay, and Uruguay

IUCN Global Category

Least Concern (assessed 2020, ver. 3.1).

Biological criteria for inclusion in Appendix I

A) Small wild population

Caiman latirostris is native to Argentina, Bolivia, Brazil, Paraguay, and Uruguay. It is distributed throughout the Paraná, Paraguay, São Francisco and Uruguay River basins, and has the southernmost distribution among Neotropical crocodilian species.

More than 70% of the species' global distribution is within Brazilian territory.

The total population size of caimans is hard to estimate, because of a number of methodological constraints. However, assuming the species only occurred in one-fifth of its >20,000 km² area of occupancy, equating to 4000 km² or 400,000 ha, and using an estimated density of 1–2 individuals/ha, the estimated total population ranges from 400,000 to 800,000 individuals in Brazil.

Density estimates of the Broad-snouted Caiman tend to vary greatly across the species' distribution range (Marques et al., 2016). Regarding the estimated population size in Brazil included in the proposal, Siroski (in litt., 2022) considers several assumptions used may not be correct and that extrapolations across a habitat that is not continuous, particularly in Brazil where the population's distribution is most fragmented, does not appear to be justified. Nevertheless, the species remains widely distributed and abundant throughout much of its range (Siroski, et al., 2020).

In Brazil the species is widely dispersed among many small habitat patches over a very large area but at a relatively low density (Verdade, 2001). Yves et al. (2017) reported 3.6 to 7.7 individuals/km in Rio Doce State Park, Minas Gerais, Brazil. The global population was classified on the IUCN Red List as Endangered from 1982–1988, as Lower Risk/Least Concern in 1996, and as Least Concern in 2019 (Siroski et al., 2020).

B) Restricted area of distribution

Caiman latirostris encounter data, obtained from the Federal Research Authorizing System (SISBIO), clearly indicates an increase in the geographic occurrence of *C. latirostris* within the country.

Caiman latirostris remains widespread and in many places abundant despite loss of habitat and disturbance in some areas (Siroski et al., 2020).

In Brazil, the species is widely dispersed among many small habitat patches over a very large area but at a relatively low density (Verdade 2001). The state of São Paulo, Brazil, is located in the central portion of the species' distribution. In this region, the species inhabits small wetlands and artificial reservoirs (Marques et al., 2016). Although canals connect these patches of habitat, there seems to be some isolation at a meta-population level possibly due to different pressures (Verdade et al., 2002). This does not appear to occur elsewhere (Siroski et al., 2020).

C) Decline in number of wild individuals

In the state of Santa Catarina, Luchese et al. (2021) report an increasing population of *C. latirostris*, particularly in suitable areas near urban centres. As an example, they mentioned a large population inhabiting a mangrove,

rivers and drainage channels close to the city of Florianopolis (South Brazil), where the species was rarely observed back in the 1990s.

There are a number of reports of *C. latirostris* being found in urban and peri-urban areas, which might indicate that these individuals are dispersing towards new areas, and suggesting an increase in the size of natural populations.

The 2020 IUCN Red List assessment considers the global population trend of C. latirostris to be stable. The species was considered to be Endangered in Red list assessments from 1982 to 1990, since 1996 it was reclassified as Lower Risk/Least concern.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

The Argentinian population of *C. latirostris* was transferred from CITES Appendix I to Appendix II in 1997. Argentina reported exporting 5,473 ranched skins in 2016 and 3,652 in 2017; and for 2018 importing countries, France, Germany, Spain and the USA, reported 2,811 skins. European imports of ranched skins of this species increased more than 20-fold within the period 2003–2012, with trade levels almost doubling between 2011 and 2012. No wild-sourced skins were imported to the EU from 2003 to 2012. Brazil reports exporting small numbers of skins from captive-bred animals most years.

According to the CITES Trade Database, from 2010 to 2020, Brazil reported eight exports of C. latirostris involving 101 skins from captive-bred individuals, 97 of them imported by Italy, and 30 specimens that had previously been seized or confiscated and were exported to the USA for scientific purposes.

For the period from 2010 to 2020, WiTIS includes details of five seizures involving C. latirostris (two concerning international trade and three domestic), none of them involving Brazil.

EU-TWIX and LEMIS register no records of illegal trade of Caiman latirostris originating in Brazil from 2011 to 2022 (with the exception of an import in 2017 to the USA of a sample for scientific purposes).

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 is already in demand for trade for skins from captive-bred specimens.

The legal trade of skins and leather would add value to the species and, consequently, to the whole ecosystem, becoming an important incentive to promote natural habitat conservation. Brazil states that it will implement a zero quota for ranched or harvested individuals.

The zero quota for ranched or harvested individuals indicated by Brazil is not being proposed as an integral part of the proposal to transfer the species' Brazilian population to Appendix II. As such Brazil could decide to lift the quota at any time. Inclusion of a zero export quota as part of the proposal to transfer the species to Appendix II would give CITES Parties the time to review population information and management strategies in the event Brazil decided to lift the quota.

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

The only form of management currently proposed is farming of Broad-snouted Caiman, following requirements of national laws and management plans. Brazil indicates it will practice a "zero quota of ranched or harvested individuals." However, it is not clear for how long this quota is planned to be implemented, or if it would be lifted depending on results obtained through the nationwide monitoring programme included in the "Programme for Conservation Biology and Management of Brazilian Crocodilians" co-ordinated by the Centre for Conservation and Management of Reptiles and Amphibians (RAN/IBAMA).

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

Brazil indicates that all CITES regulations are already applied in the country, including specific regulations for crocodilian trade and management. The Ministry of Agriculture and the State Sanitary Authority has strict measures to control meat exports, whereas cured skins can be exported, therefore, control measures can also be implemented at the tanneries. There are other government agencies that play an important role in controlling trade, particularly at the border with neighbouring countries. These include the Federal Police, the State Police in each respective state, and the Forestry Police, who can also control domestic trade.

Export quota or other special measure

Brazil indicates it will practice a “zero quota for ranched or harvested individuals”, *but it is not an export quota or an integral part of the proposal for consideration at the CITES CoP.*

It has been suggested that a zero export quota for wild specimens should be set as part of the amendment proposal (P. Siroski, in litt., 2022).

Additional information

Threats

Broad-snouted Caiman’s natural habitats are impacted by socio-economic activities since their geographic distribution coincides with the most densely occupied areas in Brazil. These are regions in which most of the natural environment has already been altered. Activities such as draining wetlands, deforestation, habitat reduction, pollution, urban expansion and intensive use of pesticides are constant threats. Despite this, *C. latirostris* is found in urban and peri-urban areas as well as in habitats affected strongly by human occupation, suggesting the species has some degree of resilience to human impacts.

In northeastern Brazil, illegal hunting still supplies local markets for meat in small cities along the São Francisco River basin. The meat is sold as salted carcasses, locally called “São Francisco codfish.” In Argentina, illegal hunting has been reduced as local rural people (“gauchos”) are currently rewarded for locating nests for the local ranching programme. Although still occurring in some places, illegal hunting is no longer the major threat to this species—probably due to a combination of reduced density, improved protection, increased cost of illegal hunting, and legal skins becoming more attractive to traders. *Illegal trade of skins has not recently been documented in Brazil.*

Conservation, management and legislation

Since 2003, Brazil has been implementing the Programme for Conservation Biology and Management of Brazilian Crocodilians, which is co-ordinated by the Centre for Conservation and Management of Reptiles and Amphibians (RAN/IBAMA). As part of this, a nationwide monitoring programme is in place employing systematic surveys applying a standardised methodology. The surveys are carried out by trained and equipped personnel to evaluate population trends in all states and habitats. Regular evaluations and systematic reports on the management programme will be provided by Brazil to ensure transparency in the programme.

The only form of management currently proposed is farming of Broad-snouted Caiman, following requirements of national laws and management plans.

Captive breeding

There are currently five operating farms for this species in Brazil, located in the south-east and north-east states that manage around 6,100 specimens. Specimens produced are used for both meat and skin.

One of the five operations that breed C. latirostris in captivity for commercial purposes is registered in CITES by Brazil (<https://cites.org/eng/common/reg/cb/BR>), in accordance with Res. Conf. 12.10 (Rev. CoP15) Registration of operations that breed Appendix-I animal species in captivity for commercial purposes.

There is insufficient information provided on the farms, their history and the extent of production through captive breeding. The farms that are established are facing financial problems, as are caiman farms elsewhere. The value of skins in the international market is very low indicating that domestic use of skins will be more important than international trade (G. Webb, in litt., 2022).

Implementation challenges (including similar species)

There are already CITES-related measures for *C. latirostris* in place in Brazil, and considering the zero quota for ranched or harvested individuals that Brazil proposed, *no additional implementation challenges are envisioned. However, if this quota is lifted, and Brazil should decide to harvest specimens from the wild for trade purposes (including through a ranching scheme), there would need to be enough solid and updated information available for the development of Non-Detriment Findings.*

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

Brazil indicated that it will implement a zero quota for ranched or harvested individuals, *although it did not specify a duration for the quota, nor is it an integral part of the proposal.*

Risks related to the transfer of the species to Appendix II would be reduced if the zero quota, indicated by Brazil, is proposed as an integral part of the proposal for consideration at the CoP. In the event Brazil wishes to remove the quota, this measure will enable time for information on wild populations and a sustainable management plan for the species to be compiled and approved by relevant experts (e.g., the IUCN-SSC CSG) and by the Conference of the Parties.

Potential benefit(s) for trade regulation of a transfer from Appendix I to II

As an outcome of the proposal, Brazil expects the social development of the country's local communities through the management and conservation of Broad-snouted Caiman populations.

Other comments

Setting the zero "export" quota for ranched or harvested individuals that Brazil has stated it would implement nationally as an integral part of the proposal to transfer *C. latirostris* from Appendix I to Appendix II would satisfy the precautionary safeguards laid out in Annex 4 of Res. Conf. 9.24 (Rev. CoP17).

Caiman latirostris is listed as endangered in the US Endangered Species Act (ESA), except for the Argentinian population, which is listed as threatened, and which is currently the only population of the species that has been transferred from CITES Appendix I to Appendix II. Specimens and products of species and populations listed as endangered or threatened in the ESA can not be imported into the US for commercial purposes even when they have been bred in captivity in line with Res. Conf 10.16 (Rev) Specimens of animal species bred in captivity. The Argentinian population was reclassified in the ESA from endangered to threatened after it had been transferred from CITES Appendix I to Appendix II, and a special rule was applied to allow the US to trade in skins, other parts, and products of this species originating from Argentina if certain conditions are met prior to exportation to the US.

There are other examples where the transfer of a crocodilian species (or population) from Appendix I to Appendix II precedes a reclassification or deletion from the ESA, or the adoption of a special rule to allow for trade to the USA of the species' parts and derivatives in accordance with CITES (e.g., the Australian population of *Crocodylus porosus*, and *Crocodylus moreletii*).

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Transfer of the Philippine population of Saltwater Crocodile *Crocodylus porosus* in Palawan Islands from Appendix I to Appendix II with a zero export quota for wild specimens

Proponent: Philippines

Summary: *Crocodylus porosus* is one of the most widely distributed crocodilians, found in East and South-East Asia and Australasia where its current range encompasses 17 range States. It is essentially extinct in the wild in Cambodia, Thailand, and Viet Nam. Originally included in Appendix II in 1975, all but one national population of the species was transferred to Appendix I in 1979. Subsequently various populations (in Australia, Indonesia, Malaysia and Papua New Guinea) have been transferred to Appendix II. The species was assessed as Least Concern by IUCN in 2019. The Philippines is proposing the transfer of its population in the Palawan Islands to Appendix II including a zero export quota for wild specimens.

Commercial hunting of Saltwater Crocodiles in the Philippines, mainly between 1950 and 1970, compounded by habitat loss and negative public attitudes towards crocodilians resulted in depletion of wild populations. Between 1987 and 1992, a founder stock of 301 *C. porosus* from various locations, including 140 individuals from Palawan, was relocated from the wild to the Crocodile Farming Institute (CFI) for captive breeding and to establish a local crocodile farming industry. By 1992, the total wild population on Palawan was estimated at fewer than 200.

No large wild populations of *C. porosus* remain in the Philippines. The highest numbers of *C. porosus* are reported to occur on the island of Mindanao, southern Palawan, Sulu Archipelago in southwestern Philippines, northeastern Mindanao and some part of northeastern Luzon. Population surveys were conducted in Palawan from 2014 to 2019 in 19 rivers, obtaining a relative density of 2.94 ± 1.23 crocodiles per km) with considerable variation between different river systems. Based on these relatively limited density estimates the current Palawan *C. porosus* population is estimated to be around 5,000 individuals although this is expected to be refined as more of the island is surveyed. The current figure represents a mean annual rate of increase of around 13% between 1992 and 2019 indicating significant population recovery.

Currently, all international trade is restricted to farms authorised and registered by the Department of Environment and Natural Resources (DENR; the Philippines CITES Management Authority) and the CITES Secretariat. There are three CITES-registered facilities for *C. porosus* in the country, with only two exporting at this time around 4,500 skins and leather products per year. The third, not currently exporting, is located in Palawan and was the source of the breeding stocks of the two exporting facilities. Aside from these, there are five other establishments outside Palawan holding *C. porosus* in the Philippines. Current *C. porosus* captive farm stocks number around 35,000 individuals.

The wild *C. porosus* population in the Philippines is protected by law, and no domestic or international trade in wild animals occurs. Very little illegal trade in Saltwater Crocodile products originating from the Philippines has been observed in the last decade.

It appears that the proponent intends to export ranched specimens in the future. The initial action that Philippines states it will undertake as part of the transition from a transfer of the Palawan population of *C. porosus* to Appendix II to a formal ranching programme, will be to expand the successful nest protection incentive scheme implemented in 2017, specifically to:

- a) encourage more local communities to identify wild *C. porosus* nesting sites on Palawan;
- b) protect more nests until hatching, quantify nest success, and release hatchlings in exchange for financial support;
- c) test whether strategic habitat interventions can increase *C. porosus* nest abundance; and,
- d) identify local communities and sites with the best potential for future ranching.

The Philippines contemplates that this transition will require continued population monitoring, increased commitment and investment from stakeholders, and the active participation of local communities along the process.

Analysis: Current information available on the Palawan *Crocodylus porosus* population indicates that it no longer meets the criteria for inclusion in Appendix I. Although the population could still be considered small, it does not meet any of the sub-criteria relevant to criteria (A) of Annex 1 of Res. Conf. 9.24 (Rev. CoP17), it does not have a restricted area of distribution (B), nor is it declining (C).

The Philippines proposes a zero export quota for wild specimens and has stricter domestic measures than those of CITES in prohibiting trade in wild terrestrial fauna. The proposal therefore complies with Annex 4 of Res. Conf. 9.24 on precautionary safeguards to transfer species from Appendix I to Appendix II. Any future exports of wild or ranched specimens would require the zero quota to be amended at a future Conference of the Parties. This annotation would make the listing in Appendix II stricter for wild specimens than an Appendix I listing, as it would not allow export for scientific, education and other purposes permitted under an Appendix I listing. Referring to a “zero export quota for wild specimens for commercial purposes” might reflect better the intention of the proposal.

Other Considerations: Adoption of this proposal would result in a split listing for the species within the Philippines with populations outside Palawan remaining in Appendix I. There can be practical regulatory complications associated with split-listings of a species within a country, which may create implementation challenges for importing Parties. The Philippines is proposing a zero export quota for wild specimens, but one of the three captive breeding farms registered in CITES is located in Palawan. It is also the source of specimens in the other two CITES-registered facilities in the country. Parties may therefore wish to consider the possible regulatory complications that may arise from a split-listing of *Crocodylus porosus* populations in the Philippines.

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

Crocodylus porosus was listed in Appendix II in 1975, and all populations except that of Papua New Guinea were transferred to Appendix I in 1979. Amendments to the listing took place in 1985, 1995 and 2017, and the species is currently listed in Appendix I except for the populations of Australia, Indonesia, Malaysia (where wild harvest is restricted to the state of Sarawak and a zero quota for wild specimens for the other states of Malaysia (Sabah and Peninsular Malaysia), with no change in the zero quota unless approved by the Parties) and Papua New Guinea, which are included in Appendix II.

Through Notification to the Parties No. 2010/038 (29 November 2010), the Philippines provided information about its stricter domestic measures around trade in specimens of CITES-listed species. Concerning trade in specimens of terrestrial fauna: “the export for commercial purposes of wild-caught specimens of wild terrestrial fauna is prohibited. Only specimens bred in captivity by breeders authorized and registered by the DENR, the CITES Management Authority for terrestrial wildlife, may be exported. This restriction has been in effect since 15 February 1994.”

Taxonomy

Crocodylus porosus (Schneider, 1801).

Synonyms: *Crocodylus biporcatus*, *Crocodylus oopholis*, *Crocodylus raninus* and *Oopholis pondichianus*.

CITES Standard reference: Wermuth, H., and Mertens, R. (1996). (reprint): *Schildkröte, Krokodile, Brückenechsen*. xvii + 506 pp. Jena (Gustav Fischer Verlag).

Range

Australia, Bangladesh, Brunei Darussalam, India, Indonesia, Malaysia, Myanmar, Palau, Papua New Guinea, Philippines, Singapore, Solomon Islands, Sri Lanka, Timor Leste, and Vanuatu. It is possibly extinct in the wild in Cambodia, Thailand, and Viet Nam and is extinct in the Seychelles.

IUCN Global Category

Least Concern (assessed 2021, ver. 3.1).

Biological criteria for inclusion in Appendix I**A) Small wild population**

Population surveys have been conducted in Palawan from 2014 to 2019 in 19 rivers; 8 of these on the mainland and 11 on nearby islands, covering 79.62 km where a total of 235 crocodiles was sighted, obtaining a relative density of 2.94 ± 1.23 crocodiles per km. The highest relative density (24.2 sightings/km) was found on Bugsuk Island in the Balabac island group, where an extensive swampland is associated with the river.

Of the entire river and creek systems in Palawan (N=271; 7,143 km), current records establish *C. porosus* in 21% of rivers (N=56; 1,500 km). If Bugsuk river, which has a significantly higher relative density than the rest of the rivers, is excluded, the visible population if all were surveyed by spotlight is estimated as 2,640 individuals (1,500 km x the mean density of 1.76), and with Bugsuk included, about 3,000. This estimate is considered conservative as it ignores scattered individuals throughout the other 215 waterways and the population within vegetated swamps where spotlight counting is impractical and was not undertaken. The current Palawan *C. porosus* population estimation is of more than 5,000 individuals, with 52% exceeding two metres (m) in length.

Survey information available from Palawan is not extensive and refers mainly to tidal rivers and creeks and not the freshwater swamps, where most nesting appears to take place and from which juveniles disperse into the tidal systems. Survey results indicate a highly heterogeneous abundance between different tidal rivers and creeks and a bimodal distribution, with high densities in some rivers and low densities in others. Therefore, the estimate needs to be refined as additional survey information is obtained, but at this point, the estimate of more than 5,000 individuals seems to be reasonable (Webb, in litt., 2022).

B) Restricted area of distribution

Crocodylus porosus is one of the most widely distributed of all crocodilians, ranging from southern India and Sri Lanka, throughout South-East Asia, east through the Philippines to Micronesia, and down through Indonesia, Papua New Guinea and the Solomon Islands to northern Australia (Webb et al., 2021). The species also existed in the Seychelles but the population was extirpated by the early 1800s (Webb et al., 2021).

Within the Philippines, intense historical hunting left remnant breeding populations largely restricted to the southern islands of Palawan, Mindanao, and also the northeast coast of Luzon. In Palawan Province, with 271 mostly short rivers, *C. porosus* occurred at low densities in rivers, mostly draining to the east. The current population is largely restricted to 56 locations, particularly in southern Palawan. The rivers tend to be short and lined with fringing mangroves from the mouth to a few km upstream.

*By 1994, Saltwater Crocodile populations and habitats were reduced throughout the Philippines, and no large populations of *C. porosus* remained (Ortega et al., 1994). Today they exist as a few single individuals, small groups, and some iconic populations scattered through remaining wetland habitats. Areas with the highest numbers of *C. porosus* are thought to be on the island of Mindanao (e.g., Ligawasan Marsh), rivers and estuaries around southern Palawan, Sulu Archipelago in southwestern Philippines, northeastern Mindanao and some part of north-eastern Luzon.*

*A survey in 2012 found evidence of basking areas that are potential nesting sites in the Agusan River Basin, but no large viable population of *C. porosus* is known to exist (Manalo et al., 2012). The connectivity of the islands of Palawan and Mindanao with the northeast coast of Borneo suggests a shared crocodile population (Webb et al., 2021).*

C) Decline in number of wild individuals

Commercial hunting since the 1920s, which intensified between 1950 to 1970, resulted in depletion. In addition to commercial hunting, habitat conversion, private collection and negative public attitudes towards crocodiles were all implicated in population declines up to the 1970s. Between March 1987 and October 1992, 140 *C. porosus*, mainly juveniles, were relocated from the wild in Palawan to the Crocodile Farming Institute (CFI). By 1992, the wild population on Palawan was estimated at between 57 and 131 individuals with 16 to 38 non-hatchlings. Spotlight counts in Palawan averaged 0.05 sightings per km versus 2.95 per km in 2019. If the total population in Palawan in 1992 was 200 individuals, and the current population is more than 5,000, the mean rate of increase would be 12.7% per annum; a significant recovery since 1992.

The above estimation uses the 2.95 sightings per km relative density and does not take into consideration the Bugsuk Island density (the highest density observed) or the high standard deviation of 1.23 sightings per km. In section 4.2 of the proposal, a mean density of 1.76 ± 0.38 sightings per km, excluding Bugsuk is given and this results in an estimation of 2,640 individuals (3,000 if Bugsuk was added).

However, it is clear that the population in Palawan is increasing rather than declining, and the estimates are lacking information on the populations within freshwater swamps, some of which are likely to be high as it is where most nesting appears to take place, and from which juveniles disperse into the tidal systems (Webb, in litt., 2022).

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

The wild *C. porosus* population in the Philippines is protected by law, and no domestic or international trade in wild animals occurs. All trade, domestic and international, is restricted to registered farms authorised by the DENR as wildlife facilities, and/or registered with the CITES Secretariat as commercial captive breeding operations for Appendix I species.

There are currently three CITES-registered farms, two of them outside Palawan and currently exporting mainly to Singapore. The third one, not currently exporting, is located in Palawan and has been the source of the other farms' breeding stocks. Exports of *C. porosus* specimens from the Philippines are largely restricted to raw salted skins, with an occasional skull and teeth as souvenirs. Crocodile meat is restricted to the domestic market. Tanned crocodile leather products (mainly belts, bags, wallets, and key holders) are made with re-imported Philippine *C. porosus* skins and are marketed in selected souvenir stores with an accompanying authenticity card, inspected by authorities.

The Philippines indicates that there are no (post-CITES accession: in 1981) records of illegal trade in crocodile skins, products or meat originating in the country.

During 2010–2020, the CITES Trade Database registers the export of 49,348 Saltwater Crocodile specimens (mainly skins, garments, and leather pieces) from the Philippines, which account for around 4,500 specimens per year, all of which derive from captive-bred individuals and are registered with the source code D, as reported by the exporter. Singapore imported 98% of the specimens exported by the Philippines.

For the period from 2000 to 2020, WiTIS registered only one incident (in 2016) involving the illegal export of two live Saltwater Crocodiles from the Philippines to Indonesia, which were seized in Indonesia.

*From 2011 to 2020, EU-TWIX registers only one seizure involving *C. porosus* from the Philippines. In 2014, three specimens of *C. porosus* were seized in the Netherlands and registered as being exported from the Philippines with an unknown origin.*

According to the LEMIS database, recording international import and export trade from and to the USA between 2008 and 2020, there are 10 seizure events registered involving Saltwater Crocodile specimens originating from the Philippines. These seizures consisted of 10 small-manufactured leather products and garments, and a medicinal product.

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 is in demand for trade, which has consisted of skins from captive-bred specimens from the Philippines. The endemic Philippine Crocodile *Crocodylus mindorensis* inhabits freshwater habitats in upland areas, and rarely coexists with *C. porosus*. It is easily distinguished from *C. porosus* by its size and scale pattern.

Export for commercial purposes of wild-caught *C. porosus* is prohibited throughout the Philippines, and will remain so, with the zero export quota proposed.

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

There are currently three CITES-registered farms, which export raw skins—all exports have been to Singapore, overseen by the CITES Management Authority in the Philippines and compliant with CITES.

Crocodiles throughout the Philippines will remain protected under the Philippines Wildlife Resources Conservation and Protection Act of 2001, and the Philippines will retain stricter domestic measures than those of CITES (CITES Article XIV) with regard to trade in CITES-listed specimens. Export for commercial purposes of wild-caught *C. porosus* is prohibited throughout the Philippines, and will remain so, with the zero export quota of *C. porosus* from Palawan. The zero export quota will remain in place until adaptive management approaches have been tested and approved by CITES Parties.

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

C. porosus is listed as Critically Endangered in the Philippines national list of threatened species for enforcement purposes but not based on extinction risk. Under this category, illegal acts are punishable by six months to 12 years of imprisonment or a fine of PHP5,000 (USD100) to PHP1,000,000 (USD20,500).

The implementation of CITES in the Philippines is embodied within Section 11 of R.A. 9147 “Wildlife Resources Conservation and Protection Act.” CITES import and export permits are required for trade, and international trade in non-CITES species requires export permits.

The commercial breeding or propagation of wildlife resources in the country requires a national permit. Penalties for violations committed in relation to captive breeding of *C. porosus* in the country are very steep with fines ranging from PHP5,000 to PHP300,000 (USD100 to USD6,000) or imprisonment of up to five years.

Export quota or other special measure

Export for commercial purposes of wild-caught *C. porosus* is prohibited throughout the Philippines and will remain so with the zero export quota of *C. porosus* from Palawan until the adaptive management approaches are tested and meet the approval of CITES Parties.

Additional information**Threats**

Human–crocodile conflict is a major concern to *C. porosus* in much of the Philippines. The rate of crocodile attacks on people in the country is increasing in areas where the *C. porosus* population has been recovering: 32% of reported attacks (2000–2020) are fatal, and 68% of all attacks are in Palawan. Coastal communities pressure local authorities to cull *C. porosus* and sometimes do so themselves (unauthorised) to protect their families.

The largest risk posed to successful C. porosus conservation is the cohabitation of humans with an apex predator (Webb et al., 2021). Local people do not welcome the arrival of crocodiles in many of the rivers and creeks in which they live (Webb, in litt., 2022). Other threats to C. porosus include urban encroachment, sand mining, tourism and agriculture as well as pollutants in some range States (Amarasinghe et al., 2015; Samarasinghe and Chandrasiri, 2013).

Conservation, management and legislation

The Philippines is proposing a zero quota for wild *C. porosus* specimens from the Palawan population as an interim precautionary measure. The Government can ensure management interventions achieve their goals before seeking the ability to trade internationally. The restriction to Palawan is a further precautionary measure. The initial action will be to expand the successful nest protection incentive scheme implemented on Palawan in 2017, specifically to:

- a) encourage more local communities to identify wild *C. porosus* nesting sites on Palawan,
- b) protect more nests until hatching, quantify nest success, and release hatchlings—all in exchange for financial support.
- c) test whether strategic habitat interventions can increase *C. porosus* nest abundance; and,
- d) identify local communities and sites with the best potential for future ranching.

It appears that the proponent intends to export ranched specimens in the future. The transition from the first phase (anticipated to take a minimum of two years), to a formal ranching programme will require increased commitment and investment from stakeholders. It will be trailed in sites and with communities deemed to have the best potential for success. The technologies are within the existing industry, and information from other ranching programmes is readily available.

The proposed initiative on Palawan will allow the Government and stakeholders to evaluate its sustainability and determine how best to use commercial incentives to foster tolerance and stewardship for wild *C. porosus*—rather than calling for their eradication. It will be an adaptive programme. No extension of the scheme beyond Palawan is anticipated, but may occur in the future based on the results from Palawan.

Captive breeding

There are three CITES-registered facilities for *C. porosus* in the country, with only two exporting at this time around 4,500 skins and leather products per year. The third one, not currently exporting, is located in Palawan and was the source of the breeding stocks of the two exporting facilities. Aside from these, there are five other establishments holding *C. porosus* outside of Palawan.

All trade, domestic and international, is restricted to farms registered and authorised by DENR as wildlife facilities, and/or registered with the CITES Secretariat as commercial captive breeding operations for Appendix I species. Current farm stocks are around 35,000 individuals.

Implementation challenges (including similar species)

There can be implementation challenges associated with split-listings of a species population within a country, as importers of products and specimens from that Party might have doubts about where exactly these are originated within the country. The Philippines is proposing a zero export quota for wild specimens, but one of the three captive breeding farms registered in CITES is located in Palawan, and it is where the specimens from the other two CITES-registered facilities originated. Although Res. Conf. 11.12 (Rev. CoP15) on the universal tagging system for the identification of crocodilian skins, which CITES Parties must comply with in order to export crocodile skins, allows the traceability of the origin of such skins and would help the implementation of the split-listing, it still involves challenges, for example concerning captive breeding facilities' registration needs.

Potential benefit(s) for trade regulation of a transfer from Appendix I to II

An Appendix II listing is seen by the Philippines as essential for engaging industry partnership and investment. Linking farms to the wild population and livelihoods of local people will assist sustainability.

Despite public education there is growing hostility towards *C. porosus*, which are large and dangerous predators, generally feared, responsible for fatal and non-fatal attacks on local people. Creating positive, tangible incentives for local communities to tolerate *C. porosus* is the main "compelling reason" (Annex 2, criterion bB of Resolution Conf. 9.24 (CoP17)), as implemented with *C. porosus* in Australia and Sarawak. Paying local communities to protect *C. porosus* nests and hatchlings as a trial did alter attitudes and tolerance.

In pursuing improved management in Palawan, co-operation with diverse stakeholders and experts (e.g. IUCN SSC Crocodile Specialist Group) will occur. The experience gained in Palawan will provide practical insights into *C. porosus* management in other parts of the Philippines, where public and political opposition to *C. porosus* populations are building.

Other comments

The proponent expects that an Appendix II listing will help engage industry partnership and investment and enable the transition to a formal ranching programme. Stretching links between farms, the wild population and the livelihoods of local communities, with the support of the government, is key to achieving sustainability and promoting the conservation of the Saltwater Crocodile.

The IUCN SSC Crocodile Specialist Group (Webb, in litt., 2022) points out that the closed-farming concept, fundamental to captive breeding of Appendix I specimens for commercial purposes, does not lend itself to village-level farming and trade in specimens that have been captive-bred for around eight generations, which are sold and exchanged locally, and which generate significant livelihood advantages. This issue does not seem to be exclusive to the Philippines, and it could merit a more in-depth analysis within the Convention.

*Crocodylus porosus is listed as endangered in the US Endangered Species Act (ESA), except for the Australian population (which is listed as threatened) and the Papuan New Guinean population (which is not listed), both of which are currently listed in CITES Appendix II. Specimens and products of species and populations listed as endangered or threatened in the ESA cannot be imported into the USA for commercial purposes even when they have been bred in captivity according to CITES. The Papuan New Guinean population has never been listed in Appendix I nor in the ESA, and the Australian *C. porosus* population was reclassified in the ESA from endangered to threatened after it had been transferred from CITES Appendix II to Appendix I. At the same time (1996), a special rule was adopted that allows for the importation into the USA of certain specimens of Saltwater Crocodiles from Australia and Nile crocodiles (*Crocodylus niloticus*) from those countries in which this latter species is listed in Appendix II of CITES.*

*There are other examples where a transference of a crocodilian species (or population) from Appendix I to Appendix II precedes a reclassification or deletion from the ESA, or the adoption of a special rule to allow for trade to the USA of the species' parts and derivatives in accordance to CITES (e.g. Argentinian populations of *Caiman latirostris*, and *Crocodylus moreletii*).*

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Transfer of the Thai population of Siamese Crocodile *Crocodylus siamensis* from Appendix I to Appendix II with a zero quota for wild specimens

Proponent: Thailand

Summary: *Crocodylus siamensis* historically occurred over much of mainland South-East Asia as well as parts of Indonesia. Extant populations occur in Cambodia, Indonesia, Lao People's Democratic Republic (PDR), Thailand, and Viet Nam. An IUCN Red List assessment, conducted in 2012, listed the species as Critically Endangered on the basis of a severe reduction in global populations, principally due to hunting and collection of live animals to stock farms, with all remnant subpopulations being small and fragmented and the population believed to be in continuing decline.

In Thailand, the Siamese Crocodile is thought to have been formerly widely distributed in low-altitude freshwater wetlands mainly in central and eastern areas. Extant populations are now found in a number of scattered localities in central and western Thailand. Reports in 2021 indicate that wild populations occur in six protected areas. An additional population has recently been found in a natural swamp adjacent to Cambodia. Wild populations in the country have been estimated recently as totalling more than 100 individuals.

There is currently no harvesting or trade of wild *C. siamensis* in Thailand. However, the species is traded in national and international markets derived from a large captive breeding industry in the country. In 2020, there were 731,457 individuals registered from 928 owners in Thailand. According to the CITES website, 28 operations that breed the species for commercial purposes have been registered according to Res. Conf. 12.10 (Rev. CoP15) on *Registration of operations that breed Appendix-I animal species in captivity for commercial purposes*. Thailand indicates that current production is sufficient to cover trade demand, which mainly consists of skins, meat and leather products, making it unnecessary to take specimens from the wild.

Seizure records are scarce, which suggests there is no significant illegal trade of *C. siamensis* specimens originating in Thailand, and the majority of these transactions were for personal purposes.

The Proponent argues that the crocodile industry, which once was a major driver of the decline of wild populations, currently has an important role to play in the conservation of the species as it could support the reestablishment of viable wild populations. However, there has been public and political opposition to reintroducing *C. siamensis* in Thailand for decades, which appears to be the major ongoing constraint to establishing wild populations outside protected areas, and thus, an obstacle for *in situ* conservation and species recovery.

Analysis: The Thai population of *Crocodylus siamensis* is still small and fragmented, with each subpopulation being very small. It appears to continue to meet the biological criteria for inclusion in Appendix I set out in Res. Conf. 9.24 (Rev. CoP17). Large quantities of captive-sourced specimens are exported from Thailand, although it is unlikely that wild-sourced specimens would enter trade due to extreme rarity owing to earlier overharvest.

Should the Parties decide that irrespective of the status of the wild population in Thailand, continued listing in Appendix I is not proportionate to the anticipated risks to the species (see Precautionary measures in Annex 4 of Resolution Conf. 9.24 (Rev. CoP17)), it is worth noting that Thailand is proposing as an integral part of its proposal: "a zero quota for wild specimens." This annotation would make the listing in Appendix II stricter for wild specimens than an Appendix I listing as it would not allow export for scientific, education and other purposes permitted under an Appendix I listing. Referring to a "zero export quota for wild specimens for commercial purposes" may better reflect the intention of the proposal. Any change to a zero quota would require the approval of 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 I in 1975.

Thailand entered a reservation for *C. siamensis* in 1983 when it ratified the Convention, and withdrew it in 1987. In 2013, at CoP16, Thailand submitted a proposal to transfer its populations of *C. siamensis* from Appendix I to Appendix II with a zero quota for wild specimens, which was not accepted by the Parties.

Taxonomy

Crocodylus siamensis (Schneider, 1801)

Synonyms: *Crocodylus galeatus* (Cuvier, 1807)

CITES Standard reference: Wermuth, H., and Mertens, R. (1996). (reprint): *Schildkröte, Krokodile, Brückenechsen*. xvii + 506 pp. Jena (Gustav Fischer Verlag).

Range

Cambodia, Indonesia, Lao PDR, Thailand, Viet Nam

IUCN Global Category

Critically Endangered A2cd (assessed 2012, ver. 3.1)

Biological criteria for inclusion in Appendix I

A) Small wild population

On the basis of the limited data available, the global wild population of *C. siamensis* comprises around 1,000 individuals. *No global population estimate is available for C. siamensis and national population estimates are available only for Cambodia. On the basis of the limited data, the global wild population of C. siamensis may comprise fewer than 1,000 mature individuals (Bezuijen et al., 2012).*

Surveys since the early 1990s have confirmed many fragmented and scattered populations in Thailand, largely within protected areas. *Chumarnkid (2021) reports that there are wild Siamese Crocodile populations inhabiting six protected areas and recently another viable wild population has been found outside a protected area at a natural swamp adjacent to Cambodia. Wild populations in Thailand were estimated to total more than 100 individuals.*

B) Restricted area of distribution

C. siamensis historically occurred over much of mainland South-East Asia as well as parts of Indonesia.

In Thailand, *C. siamensis* appears to have been widely distributed in low-altitude freshwater wetlands of central and eastern Thailand. Extant populations occur in a number of scattered localities in central and western Thailand. They have been reduced to non-breeding remnants in marginal habitats. Confirmed sites include Bueng Boraphet (which covers an area of 224 km²), Pang Sida and Kaeng Krachan National Parks (covering 845 km² and 2,915 km² respectively) and Khao Ang Rue Nai Wildlife Sanctuaries (comprising 108 km²).

C) Decline in number of wild individuals

Currently in Thailand there are six wild crocodile populations inhabiting protected areas and one in a non-protected area. Nests have been discovered during annual systematic surveys by a research team from the Department of Fisheries.

Chumarnkid (2021) reported that *C. siamensis* was found in 2014 and 2015 at Thung Saraeng Luang National Park in natural canals with nests and young crocodiles, indicative of a viable population.

Crocodylus siamensis is among the most threatened crocodilians. Historically, it occurred over much of mainland South-East Asia as well as parts of Indonesia. By the 1960s, the population of Siamese Crocodile was already a remnant of its historical abundance (Webb, in litt., 2022). In 1992 it was reported as virtually extinct in the wild (Thorbjarnarson, 1992), and its current distribution remains greatly diminished and fragmented (Bezuijen et al., 2012).

According to the IUCN Red List assessment conducted in 2012, global populations had been severely reduced, almost certainly by over 80% in the past 75 years equating to three generations (assuming a generation length of 25 years), principally due to hunting (Bezuijen et al., 2012) but also for some time due to the collection of live animals to stock farms, and all remnant subpopulations are small and fragmented. At the time of the assessment, the global population trend was considered to be decreasing and the species was classified as Critically Endangered, where it remains.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

There is currently no trade of wild *C. siamensis* in Thailand. However, captive-bred Siamese Crocodiles have been traded in national and international markets. According to the CITES Trade Database, 2007–2011, *C. siamensis* trade comprised skins, meat and leather products. The largest exporter was Thailand (117,875 skins, 894,628 kg of meat and 105,490 leather products), followed by Viet Nam (55,715 skins, 15,098 kg of meat and 17,755 leather products).

Reported export quantities peaked at 39,000 skins in 2008 and fluctuated between 25,000 and 36,000 until 2016 when they decreased to fewer than 15,000 for the first time since 2003. Between 2017 and 2019 they declined further to around 12,000 (with some differences among what was reported to the CITES Secretariat by importers and exporters, e.g., in 2018 importers registered more than 22,000 skin and leather products from Thailand, while the latter reported having exported around 14,000).

In 2020, the CITES Trade Database records 18,005 specimens including skins, leather products and garments exported by Thailand as reported by the importers, mainly to China, Japan, Singapore, Republic of Korea, and the Russian Federation, and 10,075 as reported by Thailand as the exporter. Additionally, 320 kg of meat was exported to Hong Kong SAR, and three live specimens to the Republic of Korea for zoological purposes.

After the enactment of the Wildlife Reservation and Protection Act, B.E. 2535 (1992), no illegal trade of wild Siamese Crocodiles was recorded in Thailand. The current commercial production from captive breeding stocks is sufficient to meet the trade demand. Therefore, it is unnecessary to take individuals of this species from the wild.

*For the period from 2000 to 2020, WiTIS registered eight seizure events involving specimens of *C. siamensis* illegally exported from Thailand to Sweden, China, and Poland. These involved low numbers of specimens including skins, leather products and garments (one to four) seized in airports, apart from a seizure in Qingdao Liuting International Airport in China that involved 244 *C. siamensis* leather products.*

*From 2011 to 2022, EU-TWIX registered 10 illegal transactions of *C. siamensis* originating in Thailand and involving 24 commodities including skins, small leather products, garments and medicine. Another 16 seizures involving 69 commodities of Siamese Crocodiles from an unknown origin are registered for this period as exported by Thailand.*

Trade data from LEMIS for 2008–2020 recorded 128 incidents of trade involving five skins and 213 leather products of Siamese Crocodiles and around 700 kg of meat and 5,531 units of medicine, originating in Thailand. Of these, 55 incidents involved wild-sourced specimens. Of the total incidents, 85% were refused upon import to, or transit via, the USA and were subsequently seized. Records show that 70% of these transactions were for personal purposes.

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

There is no harvesting of wild *C. siamensis* in Thailand as they occur in protected areas such as National Parks, Wildlife Sanctuary areas and non-hunting areas.

Currently, all commercial trade of *C. siamensis* in Thailand is derived from captive breeding operations. In 2020, there were 731,457 *C. siamensis* from 928 owners.

The proposal indicates there are 29 Thai registered crocodile farms in accordance with Resolution Conf. 12.10 (Rev. CoP15) of CITES. However, according to the CITES website, Thailand has 28 registered captive breeding operations.

It appears that current commercial production from captive breeding stock is sufficient to meet trade demand.

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

The Siamese Crocodile was included in CITES Appendix I in 1975. Since then, the Convention has proven its effectiveness in controlling international trade of the species and preventing illegal activities that may affect wild populations. Res. Conf. 11.12 *Universal tagging system for the identification of crocodilian skins*, and Res. Conf. 12.10 (Rev. CoP15). *Registration of operations that breed Appendix-I animal species in captivity for commercial purposes*, have established mechanisms to ensure control of international trade in crocodilians. In Thailand, the government and private sector have jointly planned a countrywide monitoring programme for wild populations and habitats of *C. siamensis*.

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

Thailand and all ASEAN countries are Parties to CITES. In relation to this, ASEAN has established the ASEAN Working Group on CITES and Wildlife Enforcement (AWG-CITES and WEN) and the roles of AWG-CITES and WEN emphasise the need for engaging ASEAN governments and their agencies, individuals and experts from international organisations, academia, the private sector, civil society organisations (CSOs) and the general public in combating illegal wildlife trade.

The AWG-CITES and WEN is committed to furthering co-operation among all sectors and agencies; increasing law enforcement capacity and support for investigations; encouraging strong laws and appropriate sentencing to deter criminals, and increasing public awareness of wildlife crime and its impacts to reduce consumer demand. AWG-CITES and WEN are helping to build capacity to dismantle the organised criminal networks behind wildlife crime.

Export quota or other special measure

The proposal includes a zero quota to ensure that Thailand's wild populations of Siamese Crocodiles do not become endangered by international trade.

Thailand is proposing: "a zero quota for wild specimens." This annotation would make the listing in Appendix II stricter than Appendix I for wild specimens, as it would not enable exports for scientific, education and other purposes which are allowed for Appendix I wild specimens. If the proposal is adopted, referring to a "zero export quota for wild specimens for commercial purposes" would reflect better the intention of the quota.

Additional information

Threats

Siamese Crocodiles in Thailand are affected by environmental impacts such as habitat destruction and degradation, hydropower construction, road construction, and local villager encroachment that alters habitats. The species is not currently impacted by hunting or poaching in Thailand.

Thailand has a long history of farming crocodiles, exporting crocodilian products, and importing crocodiles and crocodile skins from other countries.

Threats include the illegal collection of eggs, juveniles and adults, habitat loss, incidental capture with fishing gear, and the inherent vulnerability of remnant populations due to their small size (Bezuijen et al., 2012). Potential threats in some or all range States include climate change, hybridisation of wild populations, and the risk of genetic depression due to severely low numbers (Jelden et al., 2005, 2008).

*Public and political opposition to reintroducing *C. siamensis* appears to be the major ongoing constraint against trying to establish viable wild populations outside of protected areas in Thailand.*

Conservation, management and legislation

In Thailand, a reintroduction programme was initiated by the Royal Thai Forest Service and Crocodile Management Association of Thailand, with 20 crocodiles released in Pang Sida National Park in 2005 and 2006 (Temsiripong 2001, 2007). Few crocodiles were detected during subsequent monitoring (Temsiripong, 2007) and further releases are being considered (Bezuijen et al., 2012).

Development and implementation of reintroduction programmes in Thailand will be continued and strengthened according to the national plan. The information gained from ongoing monitoring and evaluation of these programmes will be applied to future release programmes subject to adjustment as appropriate.

Strengthening linkages between commercial captive breeding, trade, and conservation in the South-East Asian region is a prime objective. Several countries in the region have already developed crocodile farming associations and other commercial enterprises linked to the farming industry. The crocodile industry has an important role to play in the conservation of wild populations. The ultimate goal is to re-establish viable wild populations and their sustainable utilisation.

Farms in Thailand maintain large populations of captive-bred pure stocks of *C. siamensis*. However, there is also hybridisation with *C. porosus*, and farms are encouraged to segregate genetically pure *C. siamensis* for conservation. Currently, more than 7,000 animals from private farms are designated for the reintroduction programme in Thailand.

However, there is still public and political opposition to reintroducing C. siamensis outside protected areas, as most wetlands are now used by people for other purposes, and efforts to grow some remnant populations in rural areas are unable to secure the willingness of local people to participate (Webb, in litt., 2022).

Thailand also has programmes to educate and raise awareness among villagers, communities and the general public about Siamese Crocodile conservation such as reintroduction activities at various protected areas, recapturing and returning escaped crocodiles during the wet season, disseminating information and raising awareness of the law.

Captive breeding

By 2020, there were 928 crocodile farms registered with the CITES Management Authority of Thailand. There are 29 farms registered as captive breeding operations that breed Appendix I species in captivity for commercial purposes in accordance with Res. Conf. 12.10 of CITES (*CITES website includes only 28 registered operations*). Total annual production is around 200,000 individuals and all reported exports from Thailand are of skins from captive-bred specimens.

The farming industry was largely pioneered in Thailand, and C. siamensis is well-adapted to captive propagation and raising. That farming technology has allowed crocodile farming to be pursued throughout the country, similar to a domestic animal production industry, from the 1960s onward, from small village-level farms with basic facilities, to large operations using sophisticated technologies. Within the low-level farms, crocodiles are integrated into conventional animal production, being fed waste meat to produce meat for human consumption. The larger farms are often linked to smaller ones for the supply of captive-bred hatchlings (Webb, in litt., 2022).

Implementation challenges (including similar species)

At the International IUCN-SSC-Crocodile Specialist Group Regional Species Meeting in 2011, Bangkok, Thailand, the Crocodile Specialist Group (CSG) addressed many issues and recommendations for ASEAN range States, particularly illegal trade issues. It recommended dialogue among range States to resolve the issues. Accordingly, dialogue between range States can be arranged through a regional working group under an appropriate body (e.g. AWG-CITES and WEN and/or Mekong Sub-regional) in order to address regional issues concerning *C. siamensis*, with an emphasis on actions to control illegal trade.

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

To control the raising and trading of parts and derivatives from similar species with different CITES Appendices listings effectively, the range States will be required to harmonise regional regulation of registration of captive-breeding institutions with management authorities and a marking system for live animals and products.

With the zero export quota for wild specimens proposed by Thailand, there would not be substantial differences with the current listing. Captive breeding operations would not need to be registered with CITES, but they will still need to meet the CITES definition for the term "bred in captivity" according to Res. Conf. 10.16 (Rev.) Specimens of animal species bred in captivity, to export their products.

Potential benefit(s) for trade regulation of a transfer from Appendix I to II

The objective of transferring C. siamensis to Appendix II with a zero quota for wild specimens is not made clear in the proposal, and there would not be substantial differences regarding CITES regulations for the species. Therefore, no apparent benefits derived from the amendment are envisioned.

Other comments

Crocodylus siamensis is listed as endangered in the US Endangered Species Act (ESA), which does not allow the import to the US of specimens and products of the species for commercial purposes even when they have been bred in captivity according to CITES.

There are examples where a transference of a crocodilian species (or populations) from Appendix I to Appendix II precedes its reclassification or deletion from the ESA, or the adoption of a special rule to allow for trade to the US of the species' parts and derivatives in accordance with CITES (e.g., the Australian population of Crocodylus porosus, the Argentinian population of Caiman latirostris, and Crocodylus moreletii). This may underlie Thailand's proposal to transfer its Siamese Crocodile population to Appendix II.

The IUCN SSC Crocodile Specialist Group (Webb, in litt., 2022) points out that the closed-farming concept, fundamental to captive breeding of Appendix I specimens for commercial purposes, does not lend itself to village-level farming and trade in specimens that have been captive bred for around eight generations, which are sold and

exchanged locally, and which generate significant livelihood advantages. This issue does not seem to be exclusive to Thailand, and it could merit a more in-depth analysis within the Convention.

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Inclusion of Chinese Water Dragon *Physignathus cocincinus* in Appendix II

Proponents: European Union, Viet Nam

Summary: The Chinese Water Dragon *Physignathus cocincinus* is a large, brightly coloured semi-aquatic arboreal lizard widespread in lowland riverside forest across continental South-East and East Asia, occurring in Cambodia, China, Lao People's Democratic Republic (Lao PDR), Myanmar, Thailand, and Viet Nam. It is the only member of its genus. There is little information available on national distribution of the species. However, there are indications that it is patchy in some countries, being known, for example, from only a few locations in eastern Thailand. It has been introduced in various places outside its range, including Hong Kong SAR, Malaysia, Taiwan POC and the USA. This species is harvested throughout most of its range for human food as well as collection for the international and domestic pet trade. The species is reported to be easy to collect due to its sedentary nature and harvest is believed to represent a significant threat to wild populations.

Listed as Vulnerable in the IUCN Red List in 2017, and, despite being locally abundant, assessed as likely to be experiencing an ongoing population decline across its range. Detailed population and abundance estimates are lacking. Studies in northern Viet Nam, 2016–2017, found population densities to range from 1.98–2.64 individuals per 100 m along inhabited streams, and in central Viet Nam, 2014–2016, 0.85–0.95 individuals per 100 m in an undisturbed site and 0.07–0.43 individuals per 100 m (mean density 0.25 individuals per 100 m) in a disturbed sites experiencing pressures such as harvesting. In Cambodia, no numerical estimates exist, but in one site the population was inferred to have declined by approximately 50% over 18 years (three generations with an estimated generation length of six years).

Across range States, *P. cocincinus* is nationally traded in considerable numbers for human food consumption for and for the national and international pet trades.

Large numbers of mainly wild-sourced individuals are reported in the international pet trade (just over 59,000 wild-sourced individuals a year). Between 2011 and 2020 the EU reported just over 80,000 live individuals as directly imported, the majority being wild-sourced and imported from Viet Nam (~67,000). In the same period the USA reported imports of around 520,000 wild-sourced individuals (with ~35,000 captive-bred) from Viet Nam. It is suspected that some may have originated from neighbouring range States, such as Lao PDR and Thailand. Small numbers of live wild-sourced individuals originating in Thailand were also re-exported by Viet Nam to USA, possibly because of reduced availability within Viet Nam.

Physignathus cocincinus is protected in Cambodia from any form of collection, possession or persecution. Since February 2021 in China, any hunting or collection requires approval from provincial or local government. The species is legally protected in Thailand. In Viet Nam, the collection of *P. cocincinus* wild specimens in protected areas is prohibited without permits, although effective implementation is reportedly difficult. No information was available on its legal status in Lao PDR or Myanmar.

Analysis: *Physignathus cocincinus* has a wide range in mainland South-East Asia (principally in Viet Nam, Lao PDR and Cambodia), where it is extensively harvested for human food consumption and the pet trade. Population information is largely lacking for all range States although there are indications of at least locally reduced population densities in Viet Nam, ascribed to harvesting. The species was categorised in the IUCN Red List as Vulnerable in 2017 due to a suspected population decrease across its entire range of more than 30% over the past three generations. No global trade data are available, however, data from imports to the USA and EU show an annual average of just over 59,000 wild-sourced individuals, primarily imported from Viet Nam. There are also indications that Viet Nam is re-exporting individuals originating in neighbouring range States, possibly due to decreased

availability within Viet Nam. Due to the large numbers reported in trade, the vast majority of which is in wild-sourced individuals, and probable impacts of collection on wild populations, *P. cocincinus* appears to meet the criteria for inclusion in Appendix II under part B of Annex 2a of Res. Conf. 9.24 (Rev. CoP17) in that regulation of trade 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 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

Physignathus cocincinus Cuvier, 1829 is the only member of its genus in the Agamidae lizard family. There is no CITES Standard reference for the species.

Range

Native: Cambodia, China, Lao PDR, Myanmar, Thailand, and Viet Nam
Introduced: Hong Kong SAR, Malaysia, Taiwan POC, USA.

IUCN Global Category

Vulnerable (assessed 2017, ver. 3.1).

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

Physignathus cocincinus has a wide distribution in South-East and East Asia from southern China through Viet Nam, Lao PDR, Cambodia to eastern Thailand. Its occurrence in Myanmar requires verification and no detailed or regional information exists.

The population status of *P. cocincinus* is largely unknown in most range States except for Viet Nam. *The IUCN Red List Assessment describes the species as locally abundant in its range, but there is no global population estimate and the population trend is reported to be subject to ongoing declines (Stuart et al., 2019). There are no national population estimates and only a few local-level estimates that include population densities (Stuart et al., 2019).*

In Viet Nam, *P. cocincinus* populations across 14 sites in three protected areas in central Viet Nam in 2016–2017 were estimated to total between 232–250 individuals. The mean density along inhabited streams varied from 1.98–2.64 individuals every 100 m, with the density ranging from 0.80–6.62 individuals per 100 m. Across the sites, *P. cocincinus* population densities were highest at lower elevations, and decreased according to the level of harvest and habitat degradation. In northern Viet Nam, population estimates were carried out in 15 sites across four locations between 2014–2016. No *P. cocincinus* individuals were observed at eight of the 15 sites and the total population across all sites was estimated at 80 individuals. Population densities along inhabited streams ranged from 0.85–0.95 individuals per 100 m in an undisturbed site and from 0.07–0.43 individuals per 100 m (mean density 0.25 individuals per 100 m) in a disturbed site. In southern Viet Nam, population surveys were carried out in suitable habitats in two national parks (Phu Quoc and Bu Gia Map) and three districts in Phu Yen Province in 2022. A total of 97 individuals were observed across 25 night surveys, with populations in the three areas estimated to total 170 individuals in Phu Yen Province, 15 in Phu Quoc and 14 in Bu Gia Map. *Physignathus cocincinus* population densities ranged from 0.07–1.73 individuals per 100 m along inhabited streams. In Thailand, recent studies report that the *P. cocincinus* population is restricted to the east of the country and is small.

The wild population of *P. cocincinus* is reported to be decreasing due to habitat loss and degradation across parts of its range. In Cambodia, the population decreases were due to traditional harvesting and habitat loss. No numerical estimates were available to assess the scale of decline, however, it was inferred that in one location the population had declined by approximately 50% in 18 years (corresponding to about three generations). In Viet Nam, *P. cocincinus* was listed as vulnerable in the Viet Nam Red Book in 2007, due to an estimated population decline of 20% over 10 years across the country. Demographic studies in 2014 and 2017 observed extremely small populations and the lack of mature individuals in some localities. In Thailand, a relatively recent study revealed only a small population of *P. cocincinus* occurred in the east. It has been suggested that *P. cocincinus* requires a population size of between 3,000–7,000 mature individuals to maintain long-term stable populations.

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

In Cambodia, P. cocincinus is harvested by local people for human consumption in all Cambodian provinces (Thy, in litt., 2022).

In Viet Nam, the collection and sale of *P. cocincinus* for human consumption at local food markets and restaurants have been frequently reported. Both adults and eggs of this species were reported to be collected by local hunters for food. In 2006, in central Viet Nam *P. cocincinus* was sold for food for VND50,000–70,000 (approximately USD2–3 per kg; Nguyen and Bain, 2006). In 2016, the harvest by local hunters of 1,000 kg of *P. cocincinus*, equating to more than 2,000 individuals, was reported in central Viet Nam. Prices per kg ranged from VND250,000 (approximately USD12) in remote localities to VND450,000 (approximately USD20) in Hue City restaurants. In southern Viet Nam, prices for *P. cocincinus* sold for human consumption were reported to reach VND300,000 per individual (approximately USD15). *Physignathus cocincinus* individuals in the pet trade in Viet Nam were reported to fetch prices ranging from VND150,000–450,000 (approximately USD7–20) and VND60,000–480,000 (USD3–21).

In Lao PDR, P. cocincinus is frequently observed in local markets (Phimmachak, in litt., 2002). This species has been noted for sale in northern and southern Lao PDR, with a few individuals observed in two local markets in 2013. Another study observed ten individuals on three occasions in local markets in central Lao PDR.

Legal trade

According to CITES trade data, just over 80,000 live individual P. cocincinus were imported directly by the EU and the UK in the most recent ten years, 2011–2020, as reported by importers. Almost all of this live trade was exported by Viet Nam (92%; around 75,000 individuals), with most of this trade in wild-sourced individuals (90%; ~67,000) imported for commercial and unreported purposes. Live wild-sourced individuals originating in Viet Nam were predominantly imported by Germany (50%) and the Netherlands (26%). The imports during the period 2011–2020 of live wild-sourced P. cocincinus individuals into the EU and UK remained relatively stable.

According to trade data from the US (LEMIS), imports of P. cocincinus to the USA 2011–2020 almost entirely comprised live individuals totalling just over 560,000 (>99%), the vast majority of which were wild-sourced (94%). The remainder were reported as captive-born (source codes C and F) or ranched. Live wild-sourced P. cocincinus were virtually all directly imported from Viet Nam (>99%; ~520,000). During the ten-year period, 2011–2020, trade in live wild-sourced individuals declined by 28%. Of the almost 27,000 live wild-sourced individuals re-exported by the USA, Canada and the UK both imported 25% and the Republic of Korea imported 11%. A small number of individuals (300) originating in Thailand were re-exported by Viet Nam to the USA, indicating trade from other range States to Viet Nam does occur.

Illegal trade

In Viet Nam, it is illegal to collect wild animals including *P. cocincinus* without permits in protected areas (Viet Nam Governmental Decree 06/2019/ND-CP). Enforcing this law was noted to be challenging as this species occurs in both protected and unprotected areas, and rangers and relevant authorities cannot easily verify the origin of individuals in trade.

In Thailand, illegal trade in *P. cocincinus* has been reported in physical and online markets. A total of 39 illegal international trade seizures have been made with eight domestic poaching cases having been recorded since 2010.

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**
- B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved**

Additional information**Threats**

The major threats to *P. cocincinus* are from habitat loss and degradation and harvesting for local human food consumption and the domestic and international pet trade. In Viet Nam, the harvesting of *P. cocincinus* was noted to be the most serious threat and considered unsustainable. Habitat loss and degradation due to road construction and timber logging were noted to add additional pressure on the species. *In Cambodia, P. cocincinus is affected by habitat loss in lowland forests as well as continual countrywide harvesting primarily for human food (Thy, in litt., 2022). In Lao PDR, overharvesting also impacts this species which is frequently observed in local markets (Phimmachak, in litt., 2002).*

Conservation, management and legislation

This species occurs in multiple protected areas across its range and its population in Thailand occurs entirely within a protected area.

Cambodia

Physignathus cocincinus is protected under the Forest Law (2002) prohibiting any form of collection, possession or persecution.

China

Physignathus cocincinus has been listed as a key state species since February 2021 and protected by the Law of the People's Republic of China on the Protection of Wildlife. As such, a special hunting and catching licence is required to hunt or catch this species only for the purpose of scientific research, population control, epidemic focus and disease monitoring or other special circumstances. *P. cocincinus* is listed as endangered in the Red List of China.

Lao PDR

No species-specific measures are in place and its national status has not yet been assessed.

Myanmar

No species-specific measures are in place and its national status has not yet been assessed.

Thailand

Physignathus cocincinus is listed as a protected species under the national wildlife protection law, as well as the Wild Animal Conservation and Protection Act B.E.2562 (2019). Accordingly, the possession, hunting, import or export of this species is prohibited. *Physignathus cocincinus* is listed as endangered on the Red List of Thailand.

Viet Nam

No species-specific measures are in place but the collection of wild specimens in protected areas, such as national parks and nature reserves, is prohibited without government permits. However, the enforcement of this law was noted as difficult to implement effectively due to the difficulty in verifying the origin of traded individuals. Due to *P. cocincinus* being listed as vulnerable in the Red Data Book of Viet Nam and the IUCN Red List, the online sale of this species is also prohibited.

Physignathus cocincinus was included in Annex D of the EU (European Union) Council Regulation (EC) No. 388/97 in 2010, whereby all species on the list imported into the EU are recorded and made available in the CITES Trade Database.

Captive breeding

This species has been bred in captivity in Europe since at least 1975. The species readily breeds in captivity and is noted to be extremely common in the pet trade, especially in Europe.

Implementation challenges (including similar species)

The Eastern Water Dragon *Intellagama lesueurii*, formerly *Physignathus lesueurii*, is a water dragon species native to Australia with a similar ecology to *P. cocincinus* but both are readily distinguishable due to different morphology.

Potential benefit(s) of listing for trade regulation

Given the patchy nature of both trade and population data for P. cocincinus a listing would provide data in the future of the full-scale of international legal trade of P. cocincinus. A listing may also promote the further collection of empirical data on populations and densities of P. cocincinus throughout its range.

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Inclusion of Jeypore Hill Gecko *Cyrtodactylus jeyporensis* in Appendix II

Proponent: India

Summary: *Cyrtodactylus jeyporensis* is a medium-sized, bent-toed gecko that inhabits high elevation semi-evergreen forests and hills with coffee plantations in southern Orissa and northern Andhra Pradesh, India. The species is currently known to occur in two separate areas (covering less than 600 km², possibly less than 100 km²), and its habitat is being rapidly degraded due to forest fire, grazing, fuel wood collection, monoculture plantations and mining activities. No quantitative estimation of the population size is available, although surveys indicate that the remnant populations are declining at a fast rate.

The species was classified as Critically Endangered on the IUCN Red List in 2013 and later changed to Endangered in 2019 based on limited extent of occurrence, fragmented populations and continued decrease of both extent and quality of the species' habitat and population size. The decision to revise the assessment from Critically Endangered to Endangered in 2019 did not reflect an improvement of the species' status, but rather resulted from the discovery in 2012 of subpopulations in new localities in Andhra Pradesh and therefore from an increase of the species' estimated area of occurrence.

In India, the National Biodiversity Authority requires permission to collect the species for research, commercial utilisation, bio-survey or bio-utilisation by foreign citizens, foreign corporate bodies, foreign associations, or non-resident Indians, or the transfer of the species to such persons, under Sections 3, 19 and 20 of the Biological Diversity Act (2002). No international legal instruments are currently in place to protect the species.

At present, habitat loss and degradation represent the main threats to the species. However, *C. jeyporensis* is now perceived to be in demand for the international pet trade on the basis of a rise in numbers of online advertisements for sale (mostly aimed at the European market) and its apparently increasing popularity among pet traders and breeders outside of India. Nevertheless, no legal or illegal species-specific trade records exist for *C. jeyporensis*.

The species *Cyrtodactylus jeyporensis* is proposed for inclusion in Appendix II under the criteria in Annex 2a paragraph A of Res. Conf. 9.24 (Rev. CoP17).

Analysis: Remaining populations of *C. jeyporensis* are characterised by a restricted area of occurrence in two separated areas, a continued decrease in population size and ongoing threats of habitat degradation, so that biological criteria in Annex 1 of Res. Conf. 9.24 (Rev. CoP17) already appear to be met. There is evidence that the collection of the Jeypore Hill Gecko for commercial purposes, specifically for the pet trade, has occurred based on online advertisements, its availability among international reptile breeders and pet traders, and opinions from various experts from India, who have also corroborated its occurrence in the illegal market outside the country. Although actual levels of harvest and trade are unknown, market signals indicate that there is a credible risk that it will be affected by trade. Given the vulnerability of this species to any degree of wild harvest, it meets the criteria for inclusion in Appendix II Criterion A of Annex 2a.

Other Considerations: The Government of India has implemented "stricter domestic measures" regarding CITES-listed species whereby the export for commercial purposes of all wild-taken specimens of species included in Appendices I, II and III (except some cultivated varieties of plant species) is prohibited. Therefore, if the proposal to include *C. jeyporensis* is adopted by the Parties, the species would subsequently be prohibited from commercial trade from India and it may be appropriate to submit to the CITES Secretariat a zero quota for wild-harvested specimens for commercial trade reflecting the "stricter domestic measures" that would come into force with the 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.

Taxonomy

Cyrtodactylus jeyporensis (Beddome, 1878).

Synonyms: *Gymnodactylus jeyporensis* (Beddome, 1878); *Geckoella jeyporensis* (Kluge, 1993); *Geckoella jeyporensis* (Agarwal and Karanth, 2015); *Cyrtodactylus (Geckoella) jeyporensis* (Wood et al., 2012).

There is no CITES Standard reference that applies to this species. The following could be used: Uetz, P., Freed, P. and Hošek, J. (eds). 2021. The Reptile Database. Available at: <http://www.reptile-database.org>.

Range

India

IUCN Global Category

Endangered B1ab (iii)+2ab (iii) (assessed 2019, ver 3.1)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

Cyrtodactylus jeyporensis is endemic to India, occurring in the Eastern Ghats (a broken mountain chain along the east coast of India) (Agarwal et al., 2012). It is known only from two separated areas in high elevation moist forest of Jeypore Hills in Orissa and neighbouring Andhra Pradesh and the extent of occurrence is highly restricted to less than 100 km² of fragmented high elevation habitat (Mohapatra, 2021). The core distribution range of *C. jeyporensis* is reported to cover Patinghe (Potangi) Hill near Jeypore in Orissa. The species' habitat is rapidly shrinking due to forest fire, grazing, mining activities, quarrying and tourist activities. Moreover, the species is presumed to have disappeared from Potangi Hill, where no suitable forest remains (Mohapatra, 2021).

There is very little information on the biological characteristics of *C. jeyporensis*. The species is nocturnal and forages on the ground as well as on trees and rocks, likely feeding on insects similarly to its congeners. Individuals can be found below rock boulders during the day in primary and well-shaded secondary forests as well as on high elevation hills. It has an altitude range of 1,100–1,400 m above sea level.

Surveys indicate that habitat degradation is leading to a decline of the population. A population observed by a team of researchers in 2012 in the Galikonda and Aaraku valleys of Andhra Pradesh was revisited in 2021—the second expedition resulted in no observations during the same month (October). Similarly, six individuals were encountered in Paderu hill during a survey conducted in 2012 but only one was at the same location in October 2021.

Cyrtodactylus jeyporensis is not used or traded domestically and any collection of the species for scientific research, commercial or any other purposes requires permission from the National Biodiversity Authority under Sections 3, 19 and 20 of the Biological Diversity Act, 2002.

At present, international trade volumes of the species are not known for certain, although illegally-sourced individuals do appear to be offered for sale outside of India for the pet trade based on species advertisements on social media (e.g., Instagram, Facebook and Twitter) and the species seems to be in demand among reptile breeders outside of the country. In July 2022, a well-known website involved in the sale of exotic animals showed two advertisements for a total of three specimens of *C. jeyporensis* (listed in September 2021 and February 2022) from Belgium and the Netherlands respectively. Additional evidence that *C. jeyporensis* is being exported internationally for the pet trade is reflected by the increasing demand for the species (Mohapatra, in litt., 2022).

However, no species-specific legal or illegal trade records are available.

Although the occurrence of *C. jeyporensis* in the illegal market outside of India is corroborated by multiple experts, habitat loss is currently considered to be the most pressing threat (Agarwal, in litt., 2022).

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

B. Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Additional information

Threats

The major threats identified for *C. jeyporensis* throughout its range are habitat loss and degradation. *Specifically, habitat loss and degradation due to coffee plantations in and around Galikonda hills, grazing by livestock and fuel wood collection in Deomali and bauxite mining in the Koraput District is reported (Mohapatra, 2021). Furthermore, monoculture plantations for silviculture (mostly silver oak and some eucalyptus) occur across the species' distribution range (Mohapatra, 2021).*

Tourism and collection for the illegal trade are considered secondary but emerging threats for the species, reflected in its availability among pet traders and breeders outside of India.

Conservation, management and legislation

At present, neither national management measures, international control measures nor any species-specific conservation measures exist for the species. The distribution range of *C. jeyporensis* does not fall within any protected area (PA) under the Wild Life (Protection) Act (1972), and therefore no conservation measures regarding the species' habitat are in place either. *However, awareness campaigns aimed at local forest managers regarding endemic species prone to illegal trade that occur within their localities (and outside PAs) are occasionally carried out (Mohapatra, in litt., 2022).*

At the national level, the collection of the species for research, commercial utilisation, bio-survey or bio-utilisation by foreign citizens, foreign corporate bodies, foreign associations, or non-resident Indians, or the transfer of the species to such persons, requires permission of the National Biodiversity Authority under Sections 3, 19 and 20 of the Biological Diversity Act, 2002. Additionally, the species is projected to be included in Schedule I of the Wild Life (Protection) Act (1972) in the near future. *If included, the Act will prohibit all trade, possession, and any form of extraction or private captive breeding programme for trade (Fernandes, in litt., 2022).* Furthermore, according to Central or State Forest legislation, when found in any forest area covered by the legislation, the species will qualify as "forest produce" and thereby its removal will be regulated.

There are no international legal instruments in place to protect the species at present.

Captive breeding

No information on captive breeding is currently available for *C. jeyporensis*.

Implementation challenges (including similar species)

This species is distinctive and unlikely to be confused with any other species currently listed in the Appendices. *One other species group of the subgenus Geckoella (G. nebulosus complex) with superficially similar features is distributed across the same range as C. jeyporensis within the Eastern Ghats (Agarwal et al., 2012), but differentiating between C. jeyporensis and species of the G. nebulosus complex is not challenging with some training (Agarwal, in litt., 2022).*

Potential benefit(s) of listing for trade regulation

If listed the species would subsequently be included in India's "stricter domestic measures" regarding CITES-listed species (according to Article XIV of the Convention), whereby the export for commercial purposes of all wild-taken specimens of species included in Appendices I, II and III (except some cultivated varieties of plant species) is banned.

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Inclusion of Helmethead Gecko *Tarentola chazaliae* in Appendix II

Proponents: Mauritania, Senegal

Summary: The Helmethead Gecko *Tarentola chazaliae* is a relatively small gecko, one of around 20 members of the genus *Tarentola*, endemic to the Atlantic Coastal Sahara, occurring along the coastline of western North Africa in Morocco, the non-self-governing territory of Western Sahara, and Mauritania. Its range extends almost continuously along 1,400 km of coast, extending some 20 km inland (although there is a single record from 150 km inland indicating a potentially much wider distribution) with reports that it is common across its known range. *T. chazaliae* was assessed as Vulnerable on the IUCN Red List in 2004 based on an extent of occurrence estimated at less than 20,000 km², a small number of known locations, continued and predicted decline in extent and area of habitat and decline in mature individuals ascribed to collection for the pet trade. Habitat degradation, caused by urbanisation and coastal development, was identified as the primary threat to the species; more recent studies indicate the species may be notably sensitive to the impacts of anthropogenic climate change.

According to the Supporting Statement, the attractive patterns and vocal behaviour of the species have resulted in numbers of *T. chazaliae* being sold into the international pet trade from the early 1970s onwards. Prices range from EUR40–60 (equivalent to USD41–61 as of August 2022) for captive-bred individuals in Europe, to USD200 for wild-caught adult females in the USA. The species is known to be offered for sale in Canada, the USA, in Europe (mainly Germany, Sweden, the UK, France, Belgium and the Czech Republic), and also in Asia (mostly China and Taiwan POC). The only national trade data are for US imports and exports (from LEMIS). A total of 420 wild-sourced individuals were imported by the USA between 2011 and 2018, from Egypt, France and Germany; and 11 wild-sourced individuals exported by the USA in 2012, originating from Egypt and Morocco. Additionally, 651 individuals reported as captive-bred were imported by the USA between 2011 and 2020, and 110 captive-bred and four captive-born individuals were exported by the USA during the same period, almost all since 2016.

A substantial portion of *T. chazaliae* individuals being offered for sale on online platforms are adult specimens, either labelled as wild-caught or captive-bred. Instances of illegal trade of *T. chazaliae* specimens have been recorded in recent years including a seizure of more than 500 wild-caught individuals in Sweden in 2018.

In Morocco, the capture, sale, acquisition and export of *T. chazaliae* without a permit is prohibited. No other legal instrument is currently in place for the protection of the species.

Analysis: No recent population size or trend estimates are available for *Tarentola chazaliae*; information on its status is conflicting. An IUCN Red List assessment in 2004 inferred likely declines owing to coastal development and collection, while more recent studies indicate that the species remains common and relatively widely distributed. The species is in demand as a pet and has featured in international trade. There is very little recent information on the scale of international trade or on any impact of collection for trade although there are indications that at least a portion of the demand for the species is now met through captive breeding. Overall, there is insufficient information to determine whether or not the species meets the criteria for inclusion in Appendix II set out in 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

Tarentola chazaliae (Mocquard, 1895)

It was previously considered to be part of the genus Geckonia, but an integrative study with morphological and molecular markers revealed that it does in fact belong to the genus Tarentola (de Melo, 2016).

The proponents place this species in the family Phyllodactylidae, but this is not in accordance with the standard nomenclatural reference adopted by the Conference of the Parties, which recognises the species to be part of the family Gekkonidae.

Synonym: *Geckonia chazaliae* (Mocquard, 1895)

Range

Morocco, Western Sahara, Mauritania, and Senegal (*debated*).

IUCN Global Category

Vulnerable A3cd; B1ab(iii,v) (assessed 2004, ver 3.1)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

Tarentola chazaliae inhabits the coast of western North Africa. The species ranges from Agadir in Morocco, south through coastal Western Sahara to Cap Blanc in Mauritania. *Its distribution in Morocco is almost continuous from Agadir southwards and amounts to approximately 1,400 km, with a 50-km gap separating the Souss Valley populations from those further south (de Rueda, 2019).* In Morocco, the species has been recorded in Ma Fatma, El Outia, Régrara, Al Argoub, Laâyoune, Sidi Warziq, Tafnidilt, Abattekh, Graret AnNwayeb, Sebkha Imlilil, Chtoukane, Atf, Oued Lakra and Ghrad Al 'Angra. Outside of Morocco it has been recorded from Nouâdhibou, Cap Blanc and Boû Lanouar in the Division of Dakhlet Nouadhibou.

The species' distribution range extends up to 20 km from the coast and within an elevation range of 0–100 m above sea level. *Sanchez-Vialas and de Rueda (2016) reported a juvenile being found during a field expedition up to 144 km inland, around Smara.* Its extent of occurrence is believed to cover less than 20,000 km².

The occurrence of *T. chazaliae* in Senegal is strongly debated and not yet confirmed.

According to the 2004 IUCN Red List assessment, the overall population trend of T. chazaliae was inferred to be decreasing with declines predicted to exceed 30% of the population over three generations.

According to the Supporting Statement, *Tarentola chazaliae* is regularly encountered in the international pet trade and relatively high numbers of specimens (especially on online websites) reflect high, ongoing demand. It was first labelled as a species of concern due to (i) lack of data on imports and (ii) the sale of specimens taken from the wild back in 2005. The species has also been found in the Asian pet trade market (mostly in mainland China and also in Taiwan POC). In North America, the species is sold in Canada and the USA for an average of CAD350 (equivalent to USD270 as of August 2022) for a pair and USD200 per adult female specimen. In Europe, *T. chazaliae* has been reported to be sold at reptile trade fairs since 1998.

According to USA trade data (2011–2020), in 2012 11 live wild-sourced specimens of T. chazaliae were re-exported for commercial purposes, one from Morocco and the remainder reported as originating from Egypt (not a range State). Additionally, the USA reported direct exports of 54, and re-exports of 60, live captive-bred and captive-born individuals for commercial purposes (Table 1).

Imports to the USA, from 2011–2020, of T. chazaliae for commercial purposes totalled 1,071 live individuals, of which 40% were reported as wild-sourced (420) and 60% captive-bred (651; Table 1). The majority of the wild-sourced trade was reported from Egypt (88%; 368), with lesser quantities from France (10%; 40) and Germany (3%; 12).

Table 1. Exports and imports of *T. chazaliae* from 2011–2020 as reported in the LEMIS database (extracted records of trade for both *Tarentola chazaliae* and its previous name, *Geckonia chazaliae*, are summarised).

| Exports | | | | | | | | | | |
|--------------|------------------------|-------------------|------------|-----------|-----------|-----------|------------|-----------|------------|--------------|
| Source* | Origin | Destination | Year | | | | | | | TOTAL |
| | | | 2011 | 2012 | 2013 | 2017 | 2018 | 2019 | 2020 | |
| C | Germany | Canada | | | | | 4 | | | 4 |
| C | Germany | Republic of Korea | | | | | 2 | | | 2 |
| C | France | Canada | | | | | 2 | 22 | | 24 |
| C | France | Czech Republic | | | | | | | 2 | 2 |
| C | France | Japan | | | | | 3 | | | 3 |
| C | France | Republic of Korea | | | | | | 3 | | 3 |
| C | Hong Kong SAR | Japan | | | | | | | 16 | 16 |
| C | Hong Kong SAR | Netherlands | | | | | | | 6 | 6 |
| C | USA | Canada | | | | 6 | 4 | 2 | | 12 |
| C | USA | Hong Kong SAR | | | | | 5 | | | 5 |
| C | USA | Japan | | | 2 | | | 16 | 9 | 27 |
| C | USA | Republic of Korea | | | | | 2 | | | 2 |
| C | USA | Taiwan POC | 4 | | | | | | | 4 |
| F | USA | Thailand | 4 | | | | | | | 4 |
| W | Egypt | Republic of Korea | | 10 | | | | | | 10 |
| W | Morocco | Republic of Korea | | 1 | | | | | | 1 |
| TOTAL | | | 8 | 11 | 2 | 6 | 22 | 43 | 33 | 125 |
| Imports | | | | | | | | | | |
| Source* | Location imported from | Year | | | | | | | | TOTAL |
| | | 2011 | 2012 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | |
| C | Germany | | | 3 | 45 | 20 | | | | 68 |
| C | France | | | | | 25 | 90 | 35 | | 150 |
| C | Hong Kong SAR | | | | | | | 30 | | 30 |
| C | Netherlands | | | | | 3 | | | | 3 |
| C | Denmark | | | | | | | | 400 | 400 |
| W | Germany | | | | 10 | 2 | | | | 12 |
| W | Egypt | 210 | 158 | | | | | | | 368 |
| W | France | | | | | | 40 | | | 40 |
| TOTAL | | 210 | 158 | 3 | 55 | 50 | 130 | 65 | 400 | 1,071 |

*Source: Specimen bred in captivity (C), specimen born in captivity (F), specimen taken from the wild (W).

Figure 1 illustrates the overall trend in exported live specimens of *T. chazaliae* from 2011–2020 for commercial purposes as reported by the USA. A 450% increase in numbers of exported captive-bred specimens was recorded from 2016–2020. No commercial exports in wild-sourced specimens have been reported since 2012.

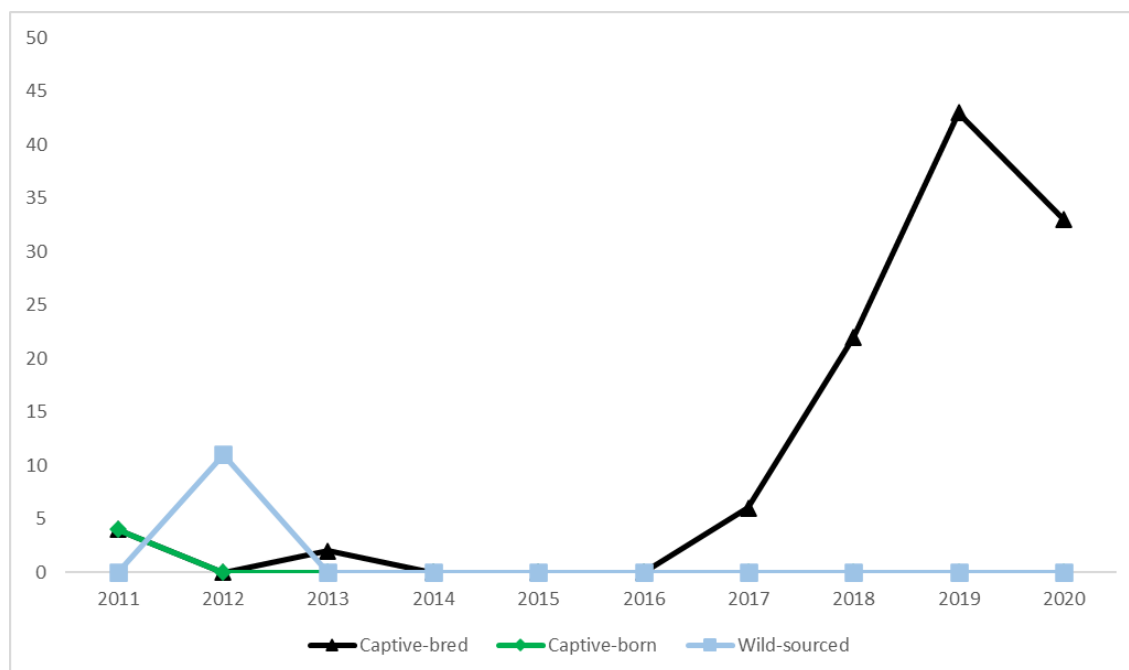


Figure 1. Number of live specimens of *T. chazaliae* exported by the USA for commercial purposes 2011–2020. The vertical axis indicates the number of individuals.

An online survey carried out from September 2017 to September 2018 in Germany found a total of 72 specimens being sold for the pet trade, among which 13 were classified as captive-bred and the remaining were of unknown origin. Prices ranged from EUR40 to EUR60 (equivalent to USD41–61 as of August 2022) per specimen. A search on a well-known website used by reptile breeders in July 2022 resulted in four advertisements (from Germany, Belgium and Austria of people seeking specimens of *T. chazaliae* to buy). The same search found one advertisement of *T. chazaliae* for sale in Switzerland in 2021.

Further preliminary searches by TRAFFIC using an online investigation software and intelligence platform into the online trade of *T. chazaliae* resulted in seven extra advertisements of live specimens for sale, with prices ranging from USD50–USD200 per individual. Five out of the seven advertisements came from the USA and two from the UK. Three of the advertisements specified that the individuals on sale were captive-bred.

Over 500 *T. chazaliae* specimens were seized in Sweden in March 2018. As there was no information on the origins of the individuals, 50 of them were given to European zoos (The Local, 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

According to the IUCN Red List assessment (2004), harvest of mature individuals for the international pet trade has already caused a decline of the wild population. However, the impact of the international pet trade has yet to be quantified.

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
- B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Additional Information

Threats

Tarentola chazaliae is impacted by the collection of mature specimens for the international pet trade and coastal fragmentation due to development and urbanisation, especially along Morocco's coast. This particular stretch of habitat was predicted to be completely transformed from 2004–2014 by development activities, and as a result the population of *T. chazaliae* was expected to undergo a 30% decline over that ten-year period, and certainly within three generations (19.5 years). Remote sensing observations (August 2022) indicate that much of the coastline within the range of the species remains undeveloped, contrary to the predictions of the 2004 Red List assessment.

Furthermore, the species was reported to be highly sensitive to climate change, due to its low intrinsic capabilities to disperse inland. A study by de Melo (2016) on the biogeographic patterns of four *Tarentola* species

across the West Sahara found that *T. chazaliae*, contrary to the other three species, occurs in areas where the temperature is relatively stable (around 23 °C) and confirmed that it is restricted by this requirement. Additional impacts on *T. chazaliae*, linked to climate change, include storm surges caused by sea level rises. This would lead to regular flooding events of low-lying beaches and subsequently reducing the area of suitable habitat. Further anthropogenic impacts include road fatalities.

Conservation, management and legislation

Most of the distribution range of *Tarentola chazaliae* falls outside protected areas. Some populations overlap protected areas in Morocco (e.g., the Oued Massa National Park) and Mauritania (e.g., the Banc d'Arguin National Park).

At present, no species-specific management plans, population monitoring, or domestic/international control measures are in place.

In Morocco, harvest, sale, acquirement, or export of *T. chazaliae* without a permit is prohibited under Category IV of Law No. 29-05, which protects species categorised as threatened by IUCN. No regulations on the protection of the species are in place in either Mauritania, Western Sahara, or Senegal.

Artificial propagation/captive breeding

Successful long-term captivity and captive breeding in *Tarentola chazaliae* specimens have been reported. Identified challenges linked to captive breeding include loss of breeding females and general difficulty in reproduction, as reported by breeders on online websites.

Implementation challenges (including similar species)

There are a total of 21 species within the genus *Tarentola*, ranging across North Africa, coastal regions of the Mediterranean Sea, Macaronesia and the West Indies (de Melo, 2016). Four of these species inhabit the West Sahara, namely *T. chazaliae*, *T. annularis*, *T. hoggarensis* and *T. parvicarinata*. Although *T. chazaliae* shares some morphological characteristics with these three other species of the genus *Tarentola* (e.g., large head, short plump body, slender limbs and short tail), in general it is still considered to be morphologically distinct from these and can be easily identified (de Melo, 2016).

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Inclusion of the Desert Horned Lizard *Phrynosoma platyrhinos* in Appendix II

Proponent: United States of America

Summary: The Desert Horned Lizard *Phrynosoma platyrhinos* is one of around 21 species of small, horned lizard in the genus *Phrynosoma*. It occurs in the west of the USA, extending into northwestern Mexico and is found in desert shrublands and the lower reaches of interior chaparral and Great Basin conifer woodlands. The species was assessed as Least Concern on the IUCN Red List in 2016, with an estimated population size of over 100,000 and a stable or slowly declining population trend. There are anecdotal accounts of local population declines but quantitative information is lacking. Four species of *Phrynosoma* (*P. blainvillii*, *P. cerroense*, *P. coronatum* and *P. wigginsi*) are currently included in Appendix II. The entire genus *Phrynosoma* is the subject of a proposal (number 18) by Mexico for inclusion in Appendix II.

Horned lizards have specialised dietary requirements (they feed almost exclusively on ants) and relatively high productivity, maturing at 10–12 months and producing one or two clutches of around seven eggs annually. Annual survivorship of young has been assessed at 26–38%. *P. platyrhinos* is primarily affected by habitat loss and degradation due to anthropogenic development, invasive non-native grasses, and climate change. Habitat loss and fragmentation have locally reduced or eliminated previously suitable habitat within the range and climate change has reportedly resulted in a shift in range to higher elevations.

Phrynosoma platyrhinos is found in the pet and reptile trade but, because of its dietary requirements, is reportedly very difficult to keep alive in captivity. The species has been recorded in international trade, with the USA reporting commercial exports of just under 20,000 live wild-sourced individuals between 2008 and 2017, with an additional 900 reported as from captive sources. Since 2018 reported trade has effectively ceased, with only three individuals reported for export for commercial purposes. The reasons for this change are not clear, although there are indications that the challenge of keeping them alive in captivity has led to their virtual disappearance from trade. There is no indication of large quantities of the species or the genus in seizure records.

Collection and sale of the species is currently regulated in the six US states in which the species occurs, with bag and possession limits ranging from three *P. platyrhinos* (Utah) per person per day to no collection allowed (Oregon).

Analysis: Although recent information is lacking, available information indicates that the Desert Horned Lizard is a relatively abundant species with a large wild population. The species has featured in the international pet trade, but is evidently very difficult to keep alive in captivity with indications that demand has latterly declined. Negligible trade has been reported in recent years (post 2017). In the decade before this, reported trade averaged around 2,000 per year. Offtake from the wild is regulated in all six states in the USA where the species occurs. Were trade to increase to pre-2018 levels, it seems unlikely that the species would meet criteria for inclusion in Appendix II as set out in Res. Conf. 9.24 (Rev CoP17). None of the species currently included in Appendix II can be easily confused with *P. platyrhinos*.

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 is no CITES Standard Reference covering the whole genus Phrynosoma.

Two subspecies are recognised:

Phrynosoma platyrhinos platyrhinos: Northern Desert Horned Lizard, endemic to the USA and occurring in the states of California, Idaho, Nevada, Oregon, and Utah.

Phrynosoma platyrhinos calidarium: Southern Desert Horned Lizard, occurring in the USA (Arizona, California, Nevada, Utah) and Mexico (Baja California).

Phrynosoma platyrhinos goodei was previously considered a subspecies of *P. platyrhinos* but is now considered the distinct species *P. goodei*.

Synonyms: *Anota calidarium*, *Anota platyrhina*, *Doliosaurus platyrhinos*, *Phrynosoma (Doliosaurus) platyrhinos*

Range

Mexico, United States of America.

IUCN Global Category

Least Concern (assessed 2016, 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

Population status

Phrynosoma platyrhinos was assessed as Least Concern on the IUCN Red List in 2016 with an estimated population size of over 100,000 individuals and a stable or slowly decreasing population trend. While no population census data exist for *P. platyrhinos* it is thought to be comparatively common among *Phrynosoma* species. The total adult population size was estimated to be greater than 100,000 individuals according to the 2016 IUCN Red List assessment, and the species is thought to have good viability or ecological integrity. *Newbold (in litt., 2022) considers that there is insufficient data to estimate a total population size.*

Regionally, populations are declining in Nevada, while being considered stable, abundant, and widespread in Arizona. Populations in Utah are also considered stable according to the Supporting Statement. *An expert noted that fewer individuals were encountered in northwest Utah in 2022 compared to 2000, suggesting some populations may be in decline (Newbold, in litt., 2022).* In the northwest extreme of its range (Oregon and Idaho), the population trends are unknown, and the species is considered vulnerable in both states. *Populations in California are estimated to be around 3,000–4,000 individuals in the most densely populated areas, and those at lower elevations are likely decreasing (Barrows, in litt., 2022). No population data were available for Mexico.*

Population range

Phrynosoma platyrhinos occurs in the west of the USA, extending into northwestern Mexico. In the USA, the species' range includes southeastern Oregon, southwestern Idaho, northern Utah, eastern and southern California, Nevada, and western Arizona; and in Mexico the species exists in northern Baja California. The extent of the range is estimated to be between 207,200 and 2,590,000 km². Populations were previously thought to exist in northwestern Sonora, Mexico and south of the Gila River in southwestern Arizona, USA. However, species in these regions are now recognised as the distinct species *Phrynosoma goodei*.

Phrynosoma platyrhinos may persist in areas where there is a low intensity of threat pressures such as urban or agricultural development, roadkill mortality, human-associated native predators, predation and injuries by domestic animals, and collection for pets.

Biological characteristics

This species has a specialised diet consisting primarily of ants, but also including a variety of slow-moving insects, spiders and plant material.

Females have an average clutch size of seven eggs, laying one to two clutches of eggs from April to July. Individuals live seven to eight years and become sexually mature at 10–12 months (*Young, in litt., 2022*). The species exhibits high reproductive investment relative to other sympatric lizard species, which is likely due to their large body size and cryptic colouring and behaviour. Compared to other lizards, horned lizards produce large numbers of eggs and expend a large amount of energy on each clutch. Annual survivorship of young is 26–38% and that of adults is 55–75%, and varies annually.

International trade

Phrynosoma platyrhinos are collected from the wild for the pet trade or by individuals as curiosities. Reports of collection date back to the early 1900s when collectors moved to new areas to supply the demand for *Phrynosoma coronatum* after it became scarce due to overexploitation. The Supporting Statement notes that this

is indicative of the boom-and-bust pattern seen in reptiles in which exploitation and trade shift from one species to another when a species becomes harder to obtain in the wild or more strictly regulated. This leads to it becoming commercially unprofitable. It is also thought that the high mortality of captive *P. platyrhinos* due to its specialised diet results in increasing numbers being collected from the wild to replace dead pets. In the USA, *Phrynosoma platyrhinos* is the most commonly traded species within the genus. *Phrynosoma platyrhinos* is traded internationally, primarily for collectors and the pet trade.

The Supporting Statement of CoP19 Prop. 18 (to list the Genus *Phrynosoma* in Appendix II) notes that there is an increasing demand for horned lizards. However, in a query to Reptiles Magazine, it was noted that horned lizards had virtually vanished from the pet trade, owing to their dietary requirements, occasionally being found for sale for reptile keepers that were “up to the challenge” (Love, 2017). It is unclear whether this decline is due to lower population numbers in the wild or due to difficulties of keeping the species in captivity.

From 2008–2020, a total 20,602 live individuals, 126 scientific specimens, and 18 bodies (including both cleared and refused records) were declared for import and export by the USA (LEMIS, 2022). When looking at cleared records only, the majority of exported live individuals were reported as wild-sourced (96%), with the remainder reported as captive-bred (3%) and captive-born (1%; Table 1). The export of live, wild-sourced individuals decreased during this period (Figure 1), with very low levels of trade reported from 2018–2020 (LEMIS, 2022). The majority of all individuals reported in trade were reported as originating in the USA (>99%). The main importers of live individuals for commercial purposes were the European Union (EU27; 46%, primarily the Netherlands and Germany), United Kingdom (UK; 12%), and Japan (11%). All imports reported by the USA were from the Netherlands (94), and the UK (46), with all re-exported wild-sourced individuals originating from the USA. An expert noted that the wild individuals in international trade were likely sourced from Nevada (Young, in litt., 2022).

Table 1. Imports and exports for commercial purposes of live individuals of *Phrynosoma platyrhinos* reported by the USA between 2008 and 2020. Wild sources include trade reported as wild (W) and unknown (U), and captive sources include captive-bred (C) and captive-born (F). All trade reported by number, and only cleared records are included (Source: LEMIS).

| Species | Imports | | Exports | |
|--|--------------|-----------------|--------------|-----------------|
| | Wild sources | Captive sources | Wild sources | Captive sources |
| <i>Phrynosoma</i> spp. (reported at genus level) | | 4 | 221 | 2 |
| <i>Phrynosoma platyrhinos</i> | 124 | 16 | 19,565 | 857 |

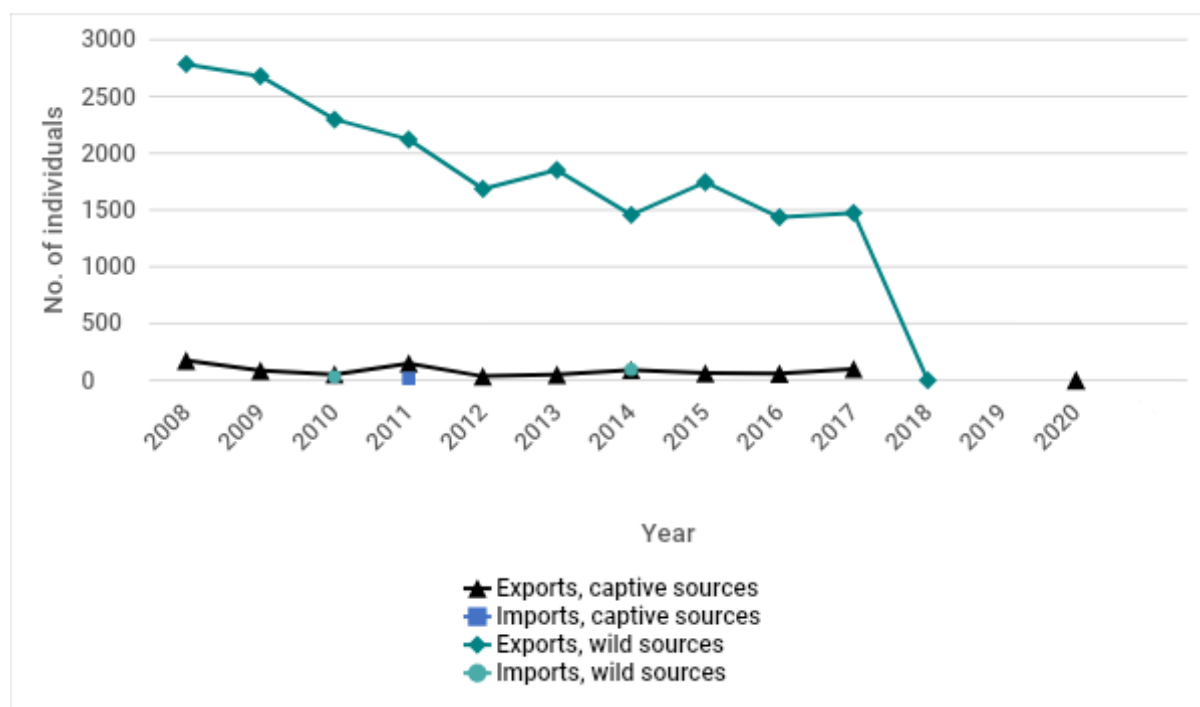


Figure 1. Imports and exports of live individuals of *Phrynosoma platyrhinos* reported by the USA between 2008 and 2020. Wild sources include trade reported as wild (W) and unknown (U), and captive sources include captive-bred (C) and captive-born (F). All trade was reported by number, and only cleared records are included (Source: LEMIS).

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

Four *Phrynosoma* taxa are currently included in CITES Appendix II. *Phrynosoma coronatum blainvilliei* was listed in Appendix II in 1975. At CoP6, *P. coronatum blainvilliei* was proposed for deletion from Appendix II, with the proposal stating there was no evidence of trade (CoP6 Prop. 28), but this was not accepted. At CoP8 (1992), *P. coronatum* was listed in CITES Appendix II following a ten-year review proposal by the USA, replacing the previous listing of the subspecies. The species was listed on the basis of controlling and monitoring trade, and to alleviate issues with subspecies identification (CoP8 Prop. 10). Since their listing, the four species currently included in Appendix II have been traded in low volumes (CITES Trade Database, 2022). Between 1991 and 2020, commercial trade in live specimens totalled 90 individuals reported by exporters and 18 reported by importers; trade peaked in 1995 and 2011 with exporters reporting 30 individuals in both years (CITES Trade Database, 2022; Table 2).

Phrynosoma blainvilliei and *P. cerroense* were split from *P. coronatum*, and *P. wigginsi* was split from *P. coronatum jamesi* in 2010, following taxonomic changes at CoP15 (CoP15 Doc. 35 [Rev. 3]) which adopted Montanucci (2004) as the Standard Reference for the four species. *P. wigginsi* is not currently considered an accepted species by Leaché and Linkem (2015). International trade in these species between 2011 and 2020 has been low (Table 2), with trade primarily occurring from the wild for scientific purposes. *P. cerroense* has not been assessed on the IUCN Red List, and *P. coronatum* and *P. blainvilliei* were assessed as Least Concern in 2007 and 2016 respectively.

Experts consider that *Phrynosoma* are generally easily distinguished (Barrows, in litt., 2022; Newbold, in litt., 2022; Young, in litt., 2022). Some species within the genus are closely related and can be difficult to distinguish, such as *P. platyrhinos* and *P. goodei* (Young, in litt., 2022). *P. goodei* was considered a subspecies of *P. platyrhinos* until 2006 (Mulcahy et al., 2006).

Table 2. CITES trade in *Phrynosoma* reported by exporters and importers between 2011 and 2020. Source indicates trade in wild (W), captive-bred (C), and previously seized or confiscated (I) specimens. Captive sources include captive-bred (C) and captive-born (F), and purpose indicates commercial (T), personal (P), and scientific (S). All trade was reported by number. No data indicates that no trade was reported (CITES Trade Database, 2022).

| Species | Term | Source | Purpose | Reported by | |
|--------------------------------|-----------|--------|---------|-------------|----------|
| | | | | Exporter | Importer |
| <i>Phrynosoma blainvilliei</i> | specimens | W | S | 35 | 35 |
| <i>Phrynosoma cerroense</i> | live | C | P | 30 | |
| <i>Phrynosoma coronatum</i> | live | C | T | | 6 |
| | | C | T | 6 | |
| | specimens | I | S | | 16 |
| | | W | S | 44 | 74 |
| <i>Phrynosoma wigginsi</i> | bodies | I | P | | 1 |

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Additional Information

Threats

Threats to the species include habitat loss and fragmentation due to urbanisation, agriculture and energy developments (e.g., solar farms), recreational off-road vehicle use, livestock grazing, fires, and invasive non-native species making habitat unsuitable. Loss and degradation of habitat displaces individuals and populations as well as creating barriers, thus hampering dispersal and immigration that may lead to lowered population viability. Drought is additionally thought to reduce fitness, survival, and reproduction. Additionally, non-native ant species displace the species' preferred food source, and is thought to be an issue particularly in California and Texas (CoP Prop. 18). The impacts of collection for trade are magnified by other concurrent threats to the species. The primary threat to *P. platyrhinos* is thought to be habitat destruction and degradation (Young, in litt., 2022), as well as rapid shifts in its habitat due to climate change causing the species to move to higher elevations (Barrows, in litt., 2022). In addition, anthropogenic development is encroaching into parts of its range, causing localised extinctions (Barrows, in litt., 2022).

Conservation, management and legislation

Phrynosoma platyrhinos is not protected under the United States Endangered Species Act. States within the USA where this species occurs have individual regulations that provide some protection to the species, but the level of protection varies. There are currently no Federal regulations that provide protection for the species at a national

level. The Supporting Statement notes that a CITES listing can complement the state regulations and management efforts to ensure that, at a national level, trade is legal and use is sustainable.

No states list the species as threatened or endangered. However, it is listed as a “Species of Greatest Conservation Need” in Nevada and its status is currently being reviewed in Idaho. In Oregon, the species is listed as “Protected Wildlife”.

Individual states within the USA have set bag and possession limits for *Phrynosoma platyrhinos* within the state or for reptiles and state-listed species more broadly (Table 3). *It is unclear how well regulated the bag and possession limits are.* Although the species is poorly studied, state resource managers and researchers have identified habitat loss and fragmentation as having a negative impact on numbers.

Table 3. Bag and possession limits, and requirements for permits or licences to collect and sell *Phrynosoma platyrhinos* in six US states.

| US state | Bag and possession limit | Permit or licence required for collection and sale |
|------------|--|--|
| Arizona | Bag and possession limit of four per person per annum (of any reptile species) | Yes |
| California | Bag limit of two live <i>P. platyrhinos</i> per day, no limit for dead specimens | Yes |
| Idaho | Bag and possession limit of four <i>P. platyrhinos</i> per person | Yes |
| Nevada | Bag limit of two <i>P. platyrhinos</i> per annum | No |
| Oregon | Listed as Protected Wildlife, collection is unlawful | Yes |
| Utah | Bag limit of three <i>P. platyrhinos</i> per day, possession limit of nine | Yes |

Mexico’s General Wildlife Law (2000) Article 84 states that any harvest of terrestrial and native wildlife species must prove that harvest rates are lower than the natural renewal of the populations subject to use and that harvest will not have a detrimental effect on populations.

Artificial Propagation/captive breeding

Most *Phrynosoma platyrhinos* specimens in trade appear to be obtained from the wild, with limited captive breeding occurring. This species is reportedly difficult to keep alive in captivity due to its specialised diet consisting primarily of ants. Due to this, many *P. platyrhinos* die in captivity. The Supporting Statement speculates that this may lead to increased levels of harvesting to replace deceased pets.

Implementation challenges (including similar species)

Phrynosoma platyrhinos can be distinguished from similar species within the genus, however, this may be difficult at juvenile stages. Body size and colouration may vary slightly regionally.

The USA and Mexico have reported exports of other species of Phrynosoma. If traded as juveniles, this may pose an implementation issue.

There are currently up to 21 accepted species of *Phrynosoma*, although some authors recognise only 12. *They are not considered to look alike, with some exceptions (see section “Inclusion in Appendix II to improve control of other listed species”). Due to taxonomic uncertainty, implementation likely requires adequate identification materials for customs officials to avoid misidentification.*

Other comments

Another proposal, CoP19 Prop. 18 to list the genus Phrynosoma in Appendix II and, if accepted, would cover Phrynosoma platyrhinos.

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Inclusion of Horned Lizards *Phrynosoma* spp. in Appendix II

Proponent: Mexico

Summary: Horned lizards *Phrynosoma* spp. are small insectivorous desert-dwelling lizards that occur in southern Canada, the United States of America (USA), and Mexico. There are up to 21 species currently recognised, however, taxonomic uncertainty remains with some authors recognising between 12 and 17 species. Four species are currently listed in Appendix II (*P. blainvillii*, *P. cerroense*, *P. coronatum* and *P. wigginsi*), although *P. coronatum* is no longer considered an accepted species by the CITES Standard Reference proposed. The USA has also proposed an individual listing of *P. platyrhinos*, CoP19 Prop. 17, which occurs in the USA and Mexico, in Appendix II.

Fifteen species have been assessed by the IUCN Red List at various times between 2007 and 2016. In 2007, one species (*P. ditmarsii*) was assessed as Data Deficient and another (*P. mcallii*) as Near Threatened; all others were assessed as Least Concern. *Phrynosoma* species are primarily affected by habitat loss and degradation due to anthropogenic development, invasive non-native grasses, and climate change. Due to climate change, some species (particularly *P. platyrhinos*, *P. hernandesi*, *P. mcallii*, and *P. blainvillii*) are thought to be declining at lower elevations with their ranges shifting to higher elevations. National assessments of *Phrynosoma* species record three vulnerable species in the USA and four threatened species in Mexico; *P. hernandesi* is considered endangered in Canada.

Thirteen of the species have been recorded in trade, most at a low or very low level. The most highly traded species is the Desert Horned Lizard *P. platyrhinos*; between 2008 and 2017 just over 2,000 live individuals were reported exported annually from the USA. Trade since then has been at a negligible level (see analysis of Prop. 17 for further discussion). Of the other species, Mexico has reported exports of just under 700 live wild-sourced *P. asio* individuals (see below). Of the four species listed in Appendix II, 90 live specimens in total were recorded in trade in the period 1991–2020. The only species currently classified as of conservation concern, (*P. mcallii* – Near Threatened) has been the subject of limited export for scientific specimens (51 imports and 53 exports reported by the US between 2011 and 2016).

The proponent seeks to include the genus *Phrynosoma* in CITES Appendix II in accordance with Res. Conf. 9.24 (Rev. CoP17), with *P. asio*, *P. braconnieri*, *P. modestum*, *P. orbiculare*, *P. platyrhinos*, *P. solare*, and *P. taurus* under Criterion A of Annex 2a and the remainder of the genus under Criterion A of Annex 2b.

- ***Phrynosoma asio*:** Endemic to Mexico, with an unknown population. Assessed as Least Concern on the IUCN Red List in 2012 with the population trend estimated to be stable. The USA reported importing 41 live captive-bred individuals between 2010 and 2014 and exporting 23 live captive-bred and 12 wild-sourced individuals (two originating from Mexico, and 10 from the USA) between 2011 and 2020. Mexico reported exporting 578 live wild-sourced individuals for commercial purposes in 2019–2021, with numbers increasing nearly 8-fold during this time, and 24 live individuals for scientific purposes in 2000–2017.
- ***Phrynosoma braconnieri*:** Endemic to Mexico and found in two states (Puebla and Oaxaca). Assessed as Least Concern on the IUCN Red List in 2007 with a stable population trend, but no population estimate available. Under special protection in Mexico. Negligible trade reported.
- ***Phrynosoma modestum*:** Occurs in Mexico and USA. Assessed as Least Concern on the IUCN Red List in 2007, with a stable population trend. No population information available. Just under 200 live wild-sourced individuals reported as exported by the USA between 2008 and 2020, most in 2014.
- ***Phrynosoma orbiculare*:** Endemic to Mexico. Assessed as Least Concern on the IUCN Red List in 2007 with a stable population trend and considered common in parts of its range. Nationally threatened in Mexico. Negligible trade reported.

- ***Phrynosoma platyrhinos***: (see analysis of CoP19 Prop. 17).
- ***Phrynosoma solare***: Occurs in USA and Mexico. Reported as widely distributed and one of the commonest *Phrynosoma* species in the Sonoran Desert. Assessed as Least Concern on the IUCN Red List in 2007, with a stable population trend. Negligible trade reported.
- ***Phrynosoma taurus***: Endemic to Mexico and occurring in four states (Morelos, Puebla, Oaxaca, and Guerrero). Assessed as Least Concern on the IUCN Red List in 2007 with a stable population trend. Assessed as nationally threatened in Mexico. Negligible trade reported.

Four species within the genus are currently listed in CITES Appendix II. However, these species are not thought to be easily confused with other species in the genus. Additionally, with minimal training the proposed species are not likely to be confused according to experts with a few exceptions of newly split and discovered species.

Analysis: Of the species of *Phrynosoma* proposed for inclusion in the Appendices, only one (*P. platyrhinos*—the subject of Proposal 17) is known to have featured in trade in notable numbers and even in this case reported trade has been at a relatively low level (around 2,000 per year from 2008–2017) with negligible trade since then. Other species have either not been recorded in trade or are recorded at a low level. This includes those species currently listed in Appendix II, in which minimal trade has been reported since 1991. With the exception of very limited export and import of scientific specimens of the Near Threatened *P. mcallii*, all species recorded in trade are not currently considered of conservation concern. Analysis of Proposal 17 indicates that *P. platyrhinos* is unlikely to meet the criteria for inclusion in Appendix II. It seems unlikely that any other *Phrynosoma* species meets these criteria.

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 is no CITES Standard Reference covering the genus *Phrynosoma*. Montanucci (2004) is the Standard Reference for the four currently Appendix II listed species *P. blainvillii*, *P. cerroense*, *P. coronatum*, and *P. wigginsi* (CoP15 Doc. 35 [Rev.3]), when *P. blainvillii*, *P. cerroense*, and *P. wigginsi* were split from *P. coronatum* on 23rd June 2010, following taxonomic changes adopted at CoP15. *P. wigginsi* is not currently considered an accepted species by Leaché and Linkem (2015).

Currently, 21 species of the genus *Phrynosoma* are recognised by the suggested Standard Reference, the Reptile Database (Uetz et al., 2022).

Range

Canada, Mexico, the USA

IUCN Global Category

Of the 21 recognised species within the genus *Phrynosoma*, 15 have been assessed on the IUCN Red List (Table 1).

Table 1. Species, IUCN Red List global threat category, and range and national threat status of the 21 species included in the listing proposal (CoP19 Prop. 18). Asterisk (*) indicates the species being proposed under Annex 2a Criterion B, dagger (†) indicates a species already listed in CITES Appendix II. National threat/protection status was retrieved from: Mexico: NOM-059-SEMARNAT-2010, last updated 2019; USA: NatureServe and state-level wildlife, fish and game departments; Canada: COSEWIC (2018). IUCN Red List population trends are indicated using the following symbols: – stable, ▼ declining and ? unknown.

| Species and year of description ⁵ | IUCN Global Category (version 3.1) | Range and national threat status |
|--|------------------------------------|--|
| <i>Phrynosoma asio</i> (1864)* | Least Concern (2012), – | Mexico (special protection) Guatemala (anecdotal evidence only) |
| <i>Phrynosoma bauri</i> (2015) | Not assessed | USA (not assessed) |
| <i>Phrynosoma blainvillii</i> (1839)† | Least Concern (2016), ▼ | USA (vulnerable) |

⁵ Following the taxonomy by Uetz et al. (2022).

| Species and year of description ⁵ | IUCN Global Category (version 3.1) | Range and national threat status |
|--|------------------------------------|---|
| | | Mexico (not assessed) |
| <i>Phrynosoma braconnieri</i> (1870)* | Least Concern (2007), – | Mexico (special protection) |
| <i>Phrynosoma brevirostris</i> (1858) | Not assessed | Canada (not assessed) USA (not assessed) |
| <i>Phrynosoma cerroense</i> (1893)† | Not assessed | Mexico (threatened) |
| <i>Phrynosoma cornutum</i> (1825) | Least Concern (2007), – | Canada (not assessed) USA (apparently Secure; Colorado: special concern, Texas: threatened, New Mexico: full protection) Mexico (not assessed) |
| <i>Phrynosoma coronatum</i> (1835)† | Least Concern (2007), ▼ | USA (not assessed) Mexico (not assessed) |
| <i>Phrynosoma diminutum</i> (2015) | Not assessed | USA (not assessed) |
| <i>Phrynosoma ditmarsii</i> (1906) | Data Deficient (2007), ? | Mexico (not assessed, <i>previously threatened but was delisted [Aguilar-Morales et al., 2018]</i>) |
| <i>Phrynosoma douglasii</i> (1829) | Least Concern (2007) | Canada (extirpated) USA (secure) Mexico (not assessed) |
| <i>Phrynosoma goodei</i> (1893) | Least Concern (2016), ? | USA (vulnerable) Mexico (not assessed) |
| <i>Phrynosoma hernandesi</i> (1858) | Least Concern (2007), – | Canada (<i>Alberta: imperiled; Saskatchewan: critically imperiled [NatureServe, 2022]; endangered; special Concern</i>) USA (secure; New Mexico: full protection, Texas: threatened, Montana: potential risk) Mexico (not assessed) |
| <i>Phrynosoma mcallii</i> (1852) | Near Threatened (2007), ▼ | USA (vulnerable; California: threatened, Arizona: threatened) Mexico (threatened) |
| <i>Phrynosoma modestum</i> (1852)* | Least Concern (2007), – | USA (secure) Mexico (not assessed) |
| <i>Phrynosoma orbiculare</i> (1758)* | Least Concern (2007), – | Mexico (threatened) |
| <i>Phrynosoma ornatissimum</i> (1858) | Not assessed | USA (not assessed) Mexico (not assessed) |
| <i>Phrynosoma platyrhinos</i> (1852)* | Least Concern (2016), – | USA (secure; <i>not protected</i>) Mexico (not assessed) |
| <i>Phrynosoma sherbrookei</i> (2014) | Not assessed | Mexico (not assessed) |
| <i>Phrynosoma solare</i> (1845)* | Least Concern (2007), – | USA (secure) Mexico (not assessed) |
| <i>Phrynosoma taurus</i> (1870)* | Least Concern (2007), – | Mexico (threatened) |

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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 trends

According to the IUCN Red List assessments, the population trends are declining for three of the species (*P. blainvillii*, *P. coronatum*, and *P. mcallii*). The remainder are either stable, unknown, or not assessed. Table 2 summarises available information per species.

Table 2. Asterisk (*) indicates the species being proposed under Annex 2a Criterion B, dagger (†) indicates a species already listed in CITES Appendix II. Evidence of international trade includes records of CITES trade data (2010–2020), LEMIS (US reported data, 2008–2020), SEMARNAT (Mexico reported data from the Supporting Statement of the Proposal).

| Species (<i>Phrynosoma</i>) | Population / abundance estimate | Range information (from SS otherwise unless stated) | Evidence of international trade |
|----------------------------------|---|--|---|
| <i>P. asio</i> * | 1,012 individuals/ha (Guerrero, Mexico). No further information provided, reference not found in Proposal. | Likely endemic to Mexico (Oaxaca, Guerrero, Michoacán, Colima, Chiapas, Morelos, Jalisco) | <p>The USA reported imports of 41 live captive-bred individuals for commercial purposes between 2010 and 2014, and reported exports of 23 live captive-bred and 12 wild-sourced individuals for commercial purposes (two originating from Mexico, and 10 from the USA) from 2011–2020 (LEMIS).</p> <p>Mexico reported exporting 578 live wild-sourced individuals for commercial purposes in the period 2019–2021, increasing nearly 8-fold from 2019–2021, and 24 live individuals in 2000–2017 for scientific purposes (SEMARNAT).</p> <p>Mexico reported seizures of 30 individuals in 2000–2020 (SEMARNAT). [Unclear if for international trade].</p> |
| <i>P. bauri</i> | No population information available | Endemic to USA (Colorado, Nebraska, Wyoming, New Mexico) | No evidence of international trade |
| <i>P. blainvillii</i> † | Presumably greater than 10,000 and may exceed 100,000 (2016). Populations in California have experienced severe declines throughout their range, leaving some locations of previously abundant populations almost or completely absent of lizards. | California (USA) and northern Baja California (Mexico) | <p>In 2015, 35 wild-sourced scientific specimens were exported from the USA to Canada, and 35 were re-exported from Canada to the USA in 2016. In 2008, 2 bodies were reported as imports from Mexico from seized and/or confiscated sources (CITES).</p> |
| <i>P. braconnieri</i> * | No population information available | Endemic to Mexico (Puebla, Oaxaca) | <p>The USA reported imports of one captive-sourced live individual from Germany in 2012 and exports to Canada of four captive-sourced live individuals in 2019 (LEMIS).</p> |
| <i>P. brevirostris</i> | No population information available | Canada (Alberta, Saskatchewan), USA (Wyoming, North Dakota, South Dakota, Nebraska, Utah, Colorado; Uetz et al. 2022). | No evidence of international trade |

| Species (<i>Phrynosoma</i>) | Population / abundance estimate | Range information (from SS otherwise unless stated) | Evidence of international trade |
|----------------------------------|---|---|--|
| <i>P. cerroense</i> † | No population information available | Endemic to Mexico (the Cerros Island, Coyote, Bahía Concepción, Baja California) | In 2011, the USA reported exports of 30 live captive-bred individuals to Germany for personal purposes (CITES). |
| <i>P. cornutum</i> | Declined throughout its range, especially in Oklahoma, and has declined in central and eastern Texas. In Colorado, the population appears to be relatively stable. | USA (Colorado, Kansas, New Mexico, Arizona, Texas, Louisiana, Oklahoma, North Carolina, introduced to Alabama and Florida), Mexico (Chihuahua, Coahuila, Durango, Nuevo Leon, Aguascalientes, San Luis Potosí, Sonora, Tamaulipas), Canada (British Columbia; Uetz et al., 2022). | The USA reported exports of 280 wild- sourced and 33 captive-born live individuals, all originating from USA (LEMIS). Mexico reported seizures of 12 individuals in 2000–2020 (SEMARNAT). [Unclear if for international trade]. |
| <i>P. coronatum</i> † | No population information available | California (USA) and Baja California (Mexico) | The USA reported no commercial trade in this species and low levels of scientific trade 2008–2020. From 2008–2010, a total of 3 bodies and one live individual originating from Mexico were refused at import (LEMIS). In 2009–2020, a total of 60 scientific specimens were reported in trade by exporters and 90 reported by importers. The USA reported imports of one body and one live individual from Mexico from seized or confiscated sources in 2009 and 2010, respectively. In 2020 a total of six live captive-bred individuals were reported in trade from Germany to Japan for commercial purposes (CITES). Mexico reported exporting eight live individuals in 2000–2017 for scientific purposes (SEMARNAT). |
| <i>P. diminutum</i> | No population information available | Endemic to USA (Colorado, likely New Mexico) | No evidence of international trade |
| <i>P. ditmarsii</i> | No population information available | Endemic to Sonoran Desert (Mexico) | The USA reported exports of 15 bodies originating from Mexico for scientific purposes in 2011 (LEMIS) |
| <i>P. douglasii</i> | No population information available | Canada (British Columbia), USA (Washington, Oregon, California, Idaho, Montana, Wyoming, North Dakota, South Dakota, Nebraska, California, Nevada, | The USA reported imports of two live individuals from captive-bred sources from the Netherlands, as well as low levels of trade in scientific specimens (LEMIS). |

| Species (<i>Phrynosoma</i>) | Population / abundance estimate | Range information (from SS otherwise unless stated) | Evidence of international trade |
|----------------------------------|---|---|--|
| | | Utah, Colorado, Arizona, New Mexico, Texas), Mexico (Chihuahua; Uetz et al., 2022). | |
| <i>P. goodei</i> | No population information available | Restricted to southwestern Arizona (USA) and northwestern Sonora (Mexico) (Hammerson, 2019). | No evidence of international trade |
| <i>P. hernandesi</i> | No population information available | Canada (Alberta, Saskatchewan) USA (New Mexico, Texas, Utah, Colorado, Arizona, South Dakota, Idaho, Nevada, Oregon) Mexico (Sonora, Chihuahua; Uetz et al., 2022). | The USA reported exports of two wild-sourced and nine captive-sourced individuals for commercial purposes in 2012–2015, as well as 15 live individuals for personal purposes to Germany in 2011, and low levels of scientific trade in 2012 (LEMIS). |
| <i>P. mcallii</i> | Estimated at 18,494 adults and 8,685 juveniles in Yuha Basin (24,122 ha) in southern California (2002). In California, during 2002–2005 in stabilised sand fields, populations declined 50% each year with an overall decline of 90%. | Endemic to the Salton Trough (Arizona, California, Mexico) | The USA reported imports of 51 and exports of 53 scientific specimens in 2011–2016, some originating from Mexico and some from the USA (LEMIS). Mexico reported seizures of six individuals in 2000–2020 (SEMARNAT). [Unclear if for international trade]. |
| <i>P. modestum</i> * | No population information available | USA (Arizona, New Mexico, Texas, Colorado), Mexico (Chihuahua, Coahuila, Nuevo Leon, Durango, Zacatecas, San Luis Potosi, Sonora, Aguascalientes; one isolated record in Tamaulipas; Uetz et al., 2022). | The USA reported commercial exports of 171 wild-sourced live individuals and 34 captive-born individuals from 2008–2020, all originating from the USA and destined for EU Member States, Japan, Switzerland, and Canada. Exports peaked in 2014 with 113 wild-sourced individuals exported. There were additionally reports of low levels of scientific trade (LEMIS). |
| <i>P. orbiculare</i> * | It is common in the Sierra el Tigre, but less so in the Yécora area (Aguilar- Morales & Van Devender, 2018). | Endemic to Mexico | The USA reported imports of six live captive-bred individuals for commercial purposes from France in 2010 and four from Germany in 2010 and 2016. In 2017, the USA reported exporting 19 live captive-bred individuals to Thailand and Japan for commercial purposes. Additionally, there were low levels of trade in scientific specimens (LEMIS). Mexico reported exporting 12 live individuals in 2000–2017 for scientific purposes (SEMARNAT). Mexico reported seizures of 46 individuals in 2000–2020 |

| Species (<i>Phrynosoma</i>) | Population / abundance estimate | Range information (from SS otherwise unless stated) | Evidence of international trade |
|----------------------------------|---|--|--|
| | | | (SEMARNAT). [Unclear if for international trade]. |
| <i>P. ornatissimum</i> | No population information available | USA (New Mexico, Texas), Mexico (Durango, Chihuahua, Zacatecas; Uetz et al., 2022). | No evidence of international trade |
| <i>P. platyrhinos</i> * | Estimated to be greater than 100,000 individuals (2016). Stable, or slowly declining | USA (Oregon, Idaho, Nevada, Utah, California, Arizona), Mexico (Baja California Norte; Uetz et al., 2022). | Over 20,000 live individuals were exported for commercial purposes by the USA from 2008–2017, of which 96% wild-sourced, and only three reported in 2018–2020 (LEMIS). |
| <i>P. sherbrookei</i> | No population information available | Endemic to Mexico (Guerrero) | No evidence of international trade |
| <i>P. solare</i> * | Widely distributed and the most common species of the genus in the Mexican Sonoran Desert (Aguilar-Morales & Van Devender, 2018). | USA (Arizona, New Mexico), Mexico (Sonora, Sinaloa, Baja California; Uetz et al., 2022). | The USA reported exports of two wild-sourced individuals to Thailand in 2017, and one captive-bred individual for commercial purposes to Japan in 2011. Low levels of trade in scientific specimens were also reported (LEMIS). |
| <i>P. taurus</i> * | 0.28 individuals/ha (Guerrero, Mexico). No further information provided, reference not found in Proposal. | Endemic to Mexico (Morelos, Puebla, Oaxaca, Guerrero) | The USA reported imports of 13 live captive-bred individuals for commercial purposes from Germany in 2012. Low levels of trade in scientific specimens were additionally reported (LEMIS). Mexico reported exporting 24 live wild-sourced individuals in 2021 for commercial purposes (SEMARNAT). |

International trade

Phrynosoma species are traded internationally for the pet trade. *Phrynosoma* species have been reported for sale in several countries in Europe and in Japan, Malaysia, the Philippines, and Taiwan POC. The Supporting Statement notes that there is an increasing demand for horned lizards. However, a query to Reptiles Magazine noted that horned lizards had virtually vanished from the pet trade owing to their dietary requirements, occasionally being found for sale for reptile keepers that were “up to the challenge” (Love, 2017).

Ten species of *Phrynosoma* have been reported in US trade data (Table 3; LEMIS, 2022). *Phrynosoma platyrhinos* was reported in the highest volumes (96% of all trade). From 2008–2020, 20,602 live *P. platyrhinos*, 126 scientific specimens, and 18 bodies, were declared for import and export from the USA (LEMIS 2022). The majority of the live exported *P. platyrhinos* were reported as wild-sourced (96%), with the remainder reported as captive-bred (3%) and captive-born (1%; Table 3; LEMIS, 2022). The general trend of exports of live, wild-sourced individuals has decreased over this time period (Figure 1), with very low levels of trade reported from 2018–2020 (LEMIS, 2022). The main importers of live individuals for commercial purposes were the European Union (EU27; 46%, primarily the Netherlands, and Germany), UK (12%), and Japan (11%). All imports of *P. platyrhinos* reported were from the Netherlands (94), and the UK (46), all originating from the USA. The majority of all individuals of all species in trade were reported as originating in the USA (>99%). There were no records of trade reported by the USA involving individuals originating from Canada.

Table 3. Imports and exports for commercial purposes of live individuals of *Phrynosoma* reported by the USA between 2008 and 2020. Wild sources include trade reported as wild (W) and unknown (U), and captive sources include captive-bred (C) and captive-born (F). All trade reported by number, and only cleared records are included. No data indicates that no trade records were available (LEMIS, 2022). Plus (+) indicates a species that occurs in the USA, asterisk (*) indicates the species being proposed under Annex 2a Criterion B.

| Species | Imports | | Exports | |
|--|--------------|-----------------|--------------|-----------------|
| | Wild sources | Captive sources | Wild sources | Captive sources |
| <i>Phrynosoma</i> spp. (reported at Genus level) | | 4 | 221 | 2 |
| <i>Phrynosoma asio</i> * | | 41 | 12 | 23 |
| <i>Phrynosoma braconneri</i> * | | 1 | | 4 |
| <i>Phrynosoma cornutum</i> + | | | 280 | 33 |
| <i>Phrynosoma douglassii</i> + | | 2 | | |
| <i>Phrynosoma hernandesi</i> + | | | 2 | 9 |
| <i>Phrynosoma modestum</i> + | | | 171 | 34 |
| <i>Phrynosoma orbiculare</i> * | | 10 | | 19 |
| <i>Phrynosoma platyrhinos</i> + | 124 | 16 | 19,565 | 857 |
| <i>Phrynosoma solare</i> + | | | 2 | 1 |
| <i>Phrynosoma taurus</i> * | | 13 | | |

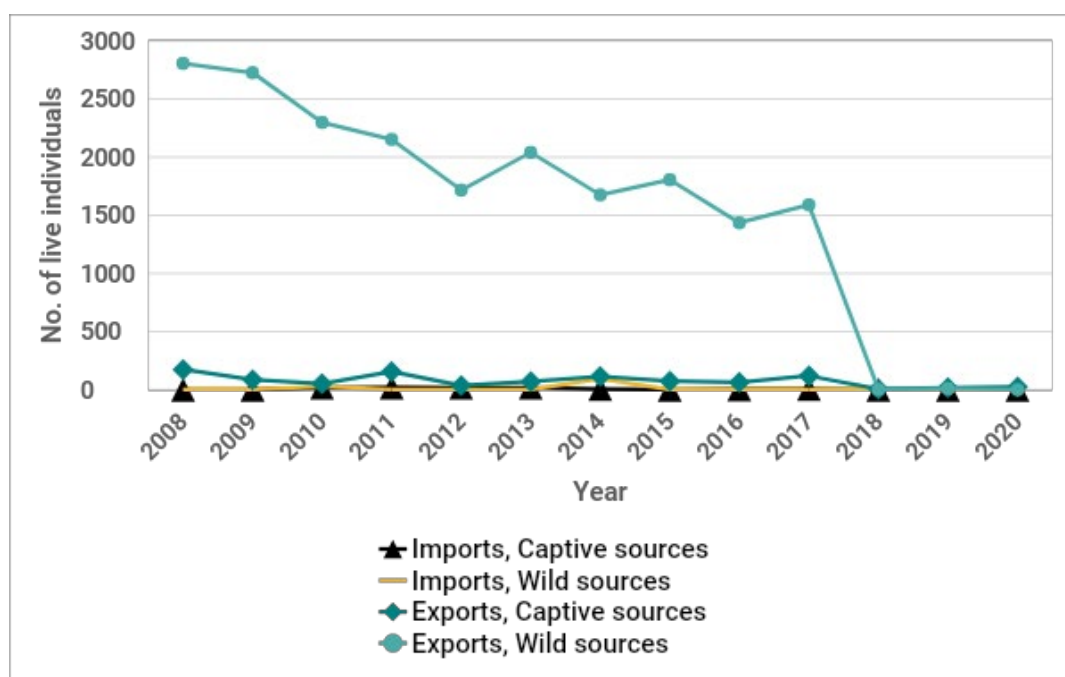


Figure 1. Imports and exports of live individuals of *Phrynosoma* species reported by the United States of America between 2008 and 2020. Wild sources include trade reported as wild (W) and unknown (U), and captive sources include captive-bred (C) and captive-born (F). All trade was reported by number, and only cleared records are included (LEMIS, 2022).

According to customs data from Mexico (SEMARNAT), Mexico reported exporting 45 live specimens of *Phrynosoma* between 2000 and 2017: 24 *P. asio*, 12 *P. orbiculare*, 8 *P. coronatum*, all for scientific and non-commercial purposes. From 2019–2021, Mexico exported a total of 602 wild-caught *Phrynosoma* species for commercial purposes with 42 *P. asio* exported in 2019; 170 *P. asio* in 2020 and 366 *P. asio* and 24 *P. taurus* in 2021. *Phrynosoma* from Mexico were reportedly exported for commercial reasons to Spain (250 specimens); Japan (174); Germany (162), and the USA (6). The Supporting Statement notes that there are some discrepancies between the records held in the CITES Trade Database, LEMIS, and Mexican data.

While there were no commercial exports reported from Mexico in 2017–2018, Mexican endemic species were found offered for sale online in Europe in the same period, including advertisements for 68 *P. orbiculare*, and two *P. taurus*. These were reportedly priced at EUR100–200 (USD100–201) and EUR500 (USD502) respectively. In the USA, online platforms advertised *P. asio* priced at USD700 for a pair (one male and one female) in 2019, and *P. platyrhinos* for USD39.99.

According to LEMIS records, in general, wild-sourced live *Phrynosoma* were reported to have lower value (in USD) than captive-sourced individuals (Table 4; LEMIS 2022). *P. asio* was reported as having the highest value (USD138 on average per individual), followed by *P. taurus* and *P. braconneri*, with an average of USD101 and USD100 respectively (LEMIS 2022). The species with the highest reported values were all species that do not naturally occur in the wild in the USA. It should be noted that these values are declared by the exporting party, and may therefore not reflect market prices.

Table 4. Average value reported per live individual in trade by source, 2008–2018, all records with values were included (including cleared, seized, and refused trade, and all purposes), all values in USD at time of reporting (LEMIS, 2022). Plus (+) indicates a species that occurs in the USA, asterisk (*) indicates the species being proposed under Annex 2a Criterion B, dagger (†) indicates a species already listed in CITES Appendix II.

| Species | Source | | |
|--|--------------------|--------------------|------------|
| | Captive-bred (USD) | Captive-born (USD) | Wild (USD) |
| <i>Phrynosoma</i> spp. (reported at Genus level) | 59 | | 7 |
| <i>Phrynosoma asio</i> * | 138 | | 49 |
| <i>Phrynosoma braconneri</i> * | 100 | | |
| <i>Phrynosoma cerroense</i> +† | 20 | | |
| <i>Phrynosoma cornutum</i> + | | 15 | 12 |
| <i>Phrynosoma coronatum</i> +† | | | 100 |
| <i>Phrynosoma douglasii</i> + | 17 | | |
| <i>Phrynosoma hernandesi</i> + | 16 | | 38 |
| <i>Phrynosoma modestum</i> +* | | 15 | 18 |
| <i>Phrynosoma orbiculare</i> * | 88 | | 1 |
| <i>Phrynosoma platyrhinos</i> +* | 10 | 10 | 11 |
| <i>Phrynosoma solare</i> + | 96 | | 15 |
| <i>Phrynosoma taurus</i> * | 101 | | |

Illegal trade and seizure records

In Mexico, *Phrynosoma* are highly valued and reportedly sold illegally on the black market and in pet stores. In Chihuahua they are collected and offered internationally in pet stores and reptile shows. From 2000–2020, Mexico seized 302 *Phrynosoma* spp. comprising *P. orbiculare* (46), *P. asio* (30), *P. cornutum* (12), *P. mcalli* (6), and 203 *Phrynosoma* reported at the genus level. *Phrynosoma* are the most frequently seized genus of lizards in Mexico. It is unclear in what circumstances these seizures were made, whether they were destined for international trade, and why they were deemed illegal.

Eight seizure records involving *Phrynosoma* are held in WiTIS. Seven were reported at the genus level involving 102 specimens in Thailand, the Philippines, Mexico, Germany, and the USA from 2013–2022, and one was reported as *P. orbiculare* involving one specimen in Mexico in 2013. Again, it is unclear whether these specimens were destined for international trade, and on what grounds they were deemed illegal.

According to LEMIS, from 2008–2020, the USA reported the refusal of 7 live individuals, 24 bodies, 50 specimens, 27 units of oil, six units of medicine, and one unit of powder at import. The one unit of powder was reported at genus level and was subsequently seized in 2015. From 2010–2011, six units of medicine and 12 units of oil were reported from wild-sourced *P. cornutum*, all of which were refused entry and then abandoned while 15 units of *Phrynosoma* spp. oil were also refused entry and then seized: all originated from Mexico. Live individuals refused at import were reported in 2010–2017 and comprised three captive-bred *P. orbiculare* originating from Germany in 2017, one wild-sourced *P. cornutum*, one wild-sourced *P. coronatum*, and two wild-sourced *P. solare* from Mexico. Also, 15 live captive-bred *P. platyrhinos* were refused export and subsequently seized in 2017.

CITES trade data

Following their listing, the four species currently included in Appendix II have been traded in low volumes (CITES Trade Database, 2022). Between 1991 and 2020, commercial trade in live specimens totalled 90 individuals reported by exporters and 18 reported by importers; trade peaked in 1995 and 2011 with exporters reporting 30 individuals in both years (CITES Trade Database, 2022; Table 5).

Table 5. CITES trade in *Phrynosoma* reported by exporters and importers between 2011 and 2020. Source indicates wild (W), captive-bred (C), and previously seized or confiscated (I). and captive sources include captive-bred (C) and captive-born (F), and purpose indicates commercial (T), personal (P), and scientific (S). All trade was reported by number (CITES Trade Database, 2022).

| Species | Term | Source | Purpose | Reported by | |
|-------------------------------|-----------|--------|---------|-------------|----------|
| | | | | Exporter | Importer |
| <i>Phrynosoma blainvillii</i> | bodies | I | P | | 2 |
| | specimens | W | S | 35 | 35 |
| <i>Phrynosoma cerroense</i> | live | C | P | 30 | |
| <i>Phrynosoma coronatum</i> | live | C | T | | 6 |
| | | C | T | 6 | |
| | specimens | I | S | | 16 |
| | | W | S | 44 | 74 |
| <i>Phrynosoma wigginsi</i> | bodies | I | P | | 1 |

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 Supporting Statement notes that “all *Phrynosoma* species look alike except to the trained eye”. They can be identified by the number of scaly dorsal rows, notch between horns, number of horns on the head, length of the tail, and stripes on the back, among other characteristics. Juveniles in particular are thought difficult to distinguish due to the lack of large horns that develop in adulthood.

Experts consider that *Phrynosoma* are generally easily distinguished (Barrows, in litt., 2022; Newbold, in litt., 2022; Young, in litt., 2022). Some species within the genus are closely related and can be difficult to distinguish, such as *P. platyrhinos* and *P. goodei* (Young, in litt., 2022), of which *P. goodei* was considered a subspecies of the former until 2006 (Mulcahy, 2006). *P. hernandesi* and *P. douglassii* are also considered difficult to distinguish (Newbold, in litt., 2022), with *P. hernandesi* previously considered a subspecies of *P. douglassii* until 1997 (Zamudio et al., 1997).

Four *Phrynosoma* species are currently included in CITES Appendix II. These are *P. blainvillii*, *P. cerroense*, *P. coronatum*, and *P. wigginsi*. *Phrynosoma coronatum blainvillei* was listed in Appendix II in 1973. At CoP6, *P. coronatum blainvillei* was proposed for deletion from Appendix II, with the proposal stating there was no evidence of trade (CoP6 Prop. 28), but this was not accepted. At CoP8 (1992), *Phrynosoma coronatum* was listed in CITES Appendix II following a ten-year review proposal by the USA, replacing the previous listing of the subspecies. The species was listed on the basis of controlling and monitoring trade and to alleviate issues with subspecies identification (CoP8 Prop. 10).

Phrynosoma blainvillii and *Phrynosoma cerroense* were split from *Phrynosoma coronatum*, and *Phrynosoma wigginsi* was split from *Phrynosoma coronatum jamesi* in 2010, following taxonomic changes adopted at CoP15 (CoP15 Doc. 35 [Rev. 3]) adopting Montanucci (2004) as the Standard Reference. *P. wigginsi* is not currently considered an accepted species by Leaché and Linkem (2015). In the past decade (2011–2020), reported international trade in these species has been low (Table 5) with trade from the wild primarily occurring for scientific purposes. *P. cerroense* has not been assessed on the IUCN Red List, and *P. coronatum* and *P. blainvillii* were assessed as Least Concern in 2007 and 2016 respectively.

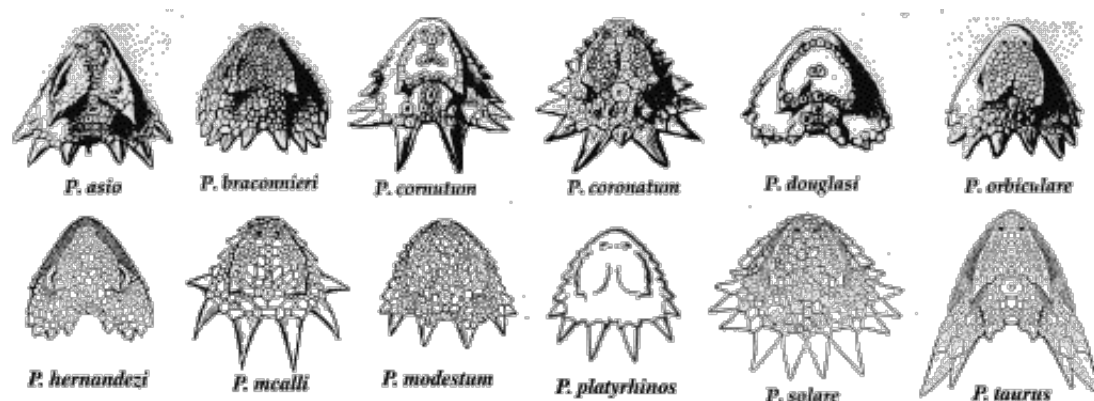


Figure 1. Ten of the *Phrynosoma* species. (Source: <http://www.zo.utexas.edu/faculty/pianka/phryno.html>, reproduced with permission from E. Pianka).

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Additional Information

Threats

The introduction of invasive ant species from Brazil and Argentina affects *Phrynosoma* in California and Texas because they do not eat them and they displace their preferred ant prey species. Other factors include agricultural pesticides, climate change, changes in land use, such as urbanisation, conversion of wildlands to agriculture, growth of woody plants, and encroachment of mixed-grass prairies. In Sonora, the introduction of non-native grass can impede the movement of some species, particularly *P. platyrhinos*.

A review of Phrynosoma species occurring in the Mexican Sonoran Desert (P. cornutum, P. ditmarsii, P. goodei, P. hernandesi, P. mcallii, P. modestum, P. orbiculare, and P. solare) concluded that "most of these horned lizards are reasonably common and not seriously threatened by habitat destruction; horned lizard populations are locally impacted by human activities including cattle grazing, buffelgrass planting, mining, and disturbance near rural towns" (Aguilar-Morales and Van Devender, 2018).

P. platyrhinos

*Threats to the species include habitat loss and fragmentation, and direct mortality due to urbanisation, agriculture and energy developments, recreational off-road vehicle use, livestock grazing, fires, and invasive non-native plant species making habitat unsuitable. Loss and degradation of habitat displaces individuals and populations as well as creating barriers hampering dispersal and immigration which may lead to lowered population viability. Drought is additionally thought to reduce fitness, survival, and reproduction. The impacts of collection for trade are magnified by other concurrent threats to the species (CoP19 Prop.17). The primary threat to *P. platyrhinos* is thought to be habitat destruction and degradation (Young, in litt., 2022), as well as rapid shifts in its habitat due to climate change, reportedly causing the species to move to higher elevations (Barrows, in litt., 2022). In addition, anthropogenic development is encroaching parts of its habitat, but due to the large range of the species some of the impacts are absorbed before leading to extinctions, although localised extinctions are likely to occur (Barrows, in litt., 2022).*

P. hernandesi

*In Canada, *P. hernandesi* faces numerous threats associated with urbanisation, tourism infrastructure and activities, agricultural activities, oil and gas drilling, and climate change. These threats contribute to habitat loss, degradation, or fragmentation and/or cause direct and indirect mortality.*

P. mcallii

*The habitat of *P. mcallii* declined by 92% in 2001–2007 in the Coachella Valley (Barrows and Allen, 2009). Barrows (in litt., 2022) considered *P. mcallii* the most at-risk species out of the three native Canadian species *P. platyrhinos*, *P. mcallii*, and *P. blainvillii*. Primary threats (in order of risk to this species) include anthropogenic habitat conversion (agriculture, energy development, urbanisation), invasive plant species (*Brassica tournefortii*), and climate change. Because of its habitat specialisation (flat, fine-textured sand deserts) it does not have an option to move to higher elevations to escape higher temperatures and increased aridity (Barrows, in litt., 2022).*

P. blainvillii

This species has a larger range but has lost much of its habitat to anthropogenic habitat conversion (one occurring throughout what is now the metropolis that includes the cities of Santa Barbara, Los Angeles, and San Diego as well as the Tijuana and Ensenada in Mexico). However, there are protected mid-elevation public lands where this

species is still relatively common (Barrows, in litt., 2022). Climate change is also a threat to *P. blainvilli* along the western edge of the Colorado and Mojave Deserts, where populations are rapidly retreating to higher elevations (Barrows, in litt., 2022). Barrows (in litt., 2022) noted that the species may be extirpated from Joshua Tree National Park within the next decade due to the park not having high enough elevations to accommodate this species' cooler temperature requirements. This species' access to cooler coastal habitats has been eliminated by total habitat conversion of that zone (Barrows, in litt., 2022).

P. coronatum

The Supporting Statement notes that *P. coronatum* is experiencing population declines and local extirpations that are more pronounced in agricultural and urban areas. A primary factor contributing to these declines is the destruction of native chaparral habitats with sandy substrates.

P. cerroense

In Mexico, *P. cerroense* is described as being affected by intensive agriculture in the Magdalena plain, potentially eliminating a significant part of its distribution area.

P. cornutum

This species has declined throughout its range, especially in Oklahoma and Texas USA due to habitat loss and alteration as a result of agriculture or urbanisation. The use of insecticides to combat the Brazilian Fire Ant may have been detrimental to *P. cornutum*, either directly or by removing its natural prey species.

Conservation, management and legislation

States in the USA where these species are found have individual regulations that provide some protection, but the level varies from state to state and there are currently no federal regulations that provide protection. Around 1994, *P. cornutum*, *P. douglassii brevirostra*, and *P. mcallii* were proposed for ESA-protection, however they were ultimately not protected (USFWS, 1994). Table 1 lists the state-level and national threatened status per species.

The Supporting Statement notes that it is unlawful to take *Phrynosoma* from the wild in Oklahoma and Texas. It is illegal to collect *P. mcallii* in Arizona and California without a state scientific collection permit (Aguilar-Morales and Van Devender, 2018). Individual states within the USA have set bag and possession limits for *P. platyrhinos* within the state or for reptiles and state-listed species more broadly (Table 6).

Table 6. Bag and possession limits, and requirements for permits or licence to collect and sell *Phrynosoma* spp. in US states (*P. platyrhinos* information: CoP19 Prop.17).

| US state | Bag and possession limit | Permit or license required for collection and sale |
|------------|--|--|
| Arizona | Bag and possession limit of four per person per annum (of any reptile species) <i>P. mcallii</i> may not be collected in Arizona. | Yes (<i>P. platyrhinos</i>) |
| California | Bag limit of two live <i>P. platyrhinos</i> per day, no limit for dead specimens | Yes (<i>P. platyrhinos</i>) |
| Idaho | Bag and possession limit of four <i>P. platyrhinos</i> per person | Yes (<i>P. platyrhinos</i>) |
| Nevada | Bag limit of two <i>P. platyrhinos</i> per annum | No (<i>P. platyrhinos</i>) |
| Oregon | Listed as Protected Wildlife, collection is unlawful | Yes (<i>P. platyrhinos</i>) |
| Texas | <i>P. cornutum</i> and <i>P. hernandesi</i> are considered threatened at the state level in Texas, meaning that these species cannot be possessed, transported, or sold without permits. | Yes |
| Utah | Bag limit of three <i>P. platyrhinos</i> per day, possession limit of 9 | Yes (<i>P. platyrhinos</i>) |

Mexico's General Wildlife Law (2000) Article 84 states that any harvest of terrestrial and native wildlife species must be based on proof that harvest rates are lower than the natural renewal of the populations subject to use and that harvest will not have a detrimental effect on populations (CoP19 Prop.17).

Phrynosoma species are found in various protected natural areas in Canada, Mexico, and the USA.

Captive breeding

Phrynosoma spp. are difficult to keep in captivity due to their highly specialised diet consisting of ants. For example, *P. orbiculare* has been reported to refuse to feed in captivity, dying of starvation within a few weeks. Nevertheless, there are records of trade in live individuals that are reported as captive-bred or captive-born (see section above on international trade).

In 2008, the Los Angeles Zoo began a captive breeding project for *P. asio* that required a very high level of care and attention in captivity due to its specific food and moisture requirements. In Mexico, there are four management units for the conservation of wildlife and two premises or facilities that manage wildlife in a confined form outside of its natural habitat that carry out intensive management of individuals of the genus *Phrynosoma*. In the USA, Fort Worth Zoo has been breeding *P. cornutum* since 2011, and Fossil Rim Wildlife Center in Texas announced the start of a breeding programme for the species in 2022 (Sawyer, in litt., 2022).

Implementation challenges (including similar species)

The Supporting Statement notes that all *Phrynosoma* species look alike except to the trained eye (see section "Inclusion in Appendix II to improve control of other listed species"). Juvenile *P. platyrhinos* are thought to be difficult to distinguish from other *Phrynosoma* species (CoP19 Prop. 17).

Due to taxonomic uncertainty, implementation would likely require adequate identification materials for customs officials to avoid mislabelling if the proposal is accepted.

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Inclusion of Pygmy Bluetongue Lizard *Tiliqua adelaidensis* in Appendix I

Proponent: Australia

Summary: The Pygmy Bluetongue Lizard *Tiliqua adelaidensis* is a medium-sized skink endemic to South Australia, where it lives in empty spider burrows in isolated remnants of native temperate grassland. The species was considered extinct until rediscovered in 1992. It was classified as Endangered by IUCN in 2017. The species was included in CITES Appendix III in 2022 (effective 22 June). All known subpopulations of *Tiliqua adelaidensis* are wholly conservation dependent.

Formerly known from the southern suburbs of the city of Adelaide, north to the town of Mannanarie, *T. adelaidensis* now only survives in the northern part of its former range in 33 small (mostly less than 100 ha), broadly disjunct patches, indicating a loss of roughly 40% of its former distribution range. There is no detailed population information; estimates range from around 5,000 to under 10,000.

The species has limited dispersal ability and all known subpopulations are thought to be genetically distinct as well as physically separated with only limited gene flow between even very close subpopulations.

Ongoing habitat destruction and loss due to intensive agriculture and grazing, coupled with the unique ecological community the species is confined to (i.e., the critically endangered Iron-grass Natural Temperate Grassland of South Australia), have already resulted in a documented, marked decline of the wild population (including mature individuals) and in complete loss of certain subpopulations. Since 1992, no definite, consistent trends have been recorded across all subpopulations—while some local subpopulations have been stable, there have been marked decreases in others.

The species is protected under national legislation and there is no legal commercial trade. Instances of Pygmy Bluetongue Lizards offered for sale were first reported in late 2017. The actual volume of illegal trade remains unknown, but the species commands high prices.

Analysis: *Tiliqua adelaidensis* is believed to have a relatively small population and a restricted and fragmented area of distribution with the area and quality of its habitat in decline. Current population trends are unknown. Despite protection under Australian law, in recent years the species has been identified within the European and Japanese pet markets and has evidently been the subject of illegal export, although the extent of this trade and its impact on wild populations remains unknown. The species may therefore meet criteria for inclusion in Appendix I set out in Annex 1 of Res. Conf. 9.24 (Rev. CoP17). Inclusion of the species in Appendix I would be in line with existing national legislation. The 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

Tiliqua adelaidensis (Peters, 1864).

Synonym: *Cyclodus adelaidensis* (Peters, 1864).

Range

Australia

IUCN Global Category

Endangered B2ab (ii,iii,iv,v) (assessed 2017, ver 3.1)

Biological criteria for inclusion in Appendix I

A) Small wild population

Overall population size has proven difficult to estimate—estimates are of under 10,000 individuals, probably around 5,000 individuals left in the wild. According to the IUCN Red List Assessment (2017), multiple subpopulations have recently been lost as they were too small to persist or due to human disturbance such as residential development.

B) Restricted area of distribution

Endemic to South Australia, *T. adalaidensis* currently extends from Peterborough south to Bagot Well. The area of occupancy amounts to 33 fragmented sites that altogether extend to less than 500 km². These remnants of patches are separated by areas of unsuitable habitat (roads, ploughed pasture and agricultural matrix) and only a small number of them reach 100 ha in size. Both the area and quality of habitat are undergoing continual decline, as the vast majority of the areas occupied by *Tiliqua adalaidensis* are now used for livestock grazing (Nielsen and Bull, 2017). Overgrazing can lead to reduced local abundances of the species's most common prey, i.e. grasshoppers, and therefore to *T. adalaidensis* individuals with lower body conditions (Nielsen and Bull, 2017). Additionally, a study by Delean et al. (2013) forecast further future habitat degradation across the northern two-thirds of the current species' range as a result of the construction of wind farms and other infrastructures.

C) Decline in number of wild individuals

Prior to 1992, there was no information on the subpopulations of *Tiliqua adalaidensis* or its habitat, as there had been no record of the species in the wild for the previous 33 years. From 1992, studies on *T. adalaidensis* have not shown consistent trends across subpopulations. In fact, while some local subpopulations have been stable, others have notably decreased. Overall, the population trend is considered to be decreasing, with certain subpopulations known to have been lost as a result of unviability.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Since national legislation prohibits export of live native reptiles for commercial purposes, there is no legal commercial trade in this species or any parts of this species. Although the volume of illegal trade is currently unknown, *Tiliqua* skinks are highly sought throughout the northern hemisphere and sale advertisements in Europe have been reported to the South Australian Government since 2017. In 2018, Pygmy Bluetongue Lizards were detected in a reptile trade outlet in the UK, where they were sold for EUR6,000 each. One other instance of sale in Europe recorded one individual being sold for EUR9,000. Gravid adults have been purchased in Germany for EUR5,000 a pair. A report on the trade in nationally protected lizards from Australia, Cuba, and Mexico reported a total of 17 specimens of *T. adalaidensis* being traded at surveyed online platforms and in social media groups in Europe over a period of six months (September 2017 to March 2018) involving traders from the UK, Germany, and Russian Federation. The report recorded individuals being sold for a minimum of EUR150 (Altherr et al., 2019). Australian wildlife enforcement authorities were informed of Pygmy Bluetongue Lizard specimens being advertised for sale in Japan in 2021.

All trade in this species from the wild for commercial purposes is prohibited under Australian law. There are no records of species-specific instances of illegal trade of *T. adalaidensis* on either the CITES Trade Database nor LEMIS. WiTIS reports a total of 20 seizures of at least 180 live individuals of the genus *Tiliqua* from 2011 to 2021 (these likely include trade instances of other *Tiliqua* spp.). Only one seizure recorded the source of the specimens (i.e., taken from the wild). Trade originated from Australia (50%), Indonesia (35%), Philippines (10%), and Thailand (5%).

The relatively low instances of illegal trafficking of the species are likely linked to limited opportunities for illegal collecting caused by international borders being closed between March 2020 and early 2022 (as well as closed state borders within Australia itself during the same period) and should not be considered as realistic for the future now that international borders have reopened (Chapple, in litt., 2022). At present, poaching to supply the illegal trade has been identified as an increasing threat, and recent indications hint that this trade is now more substantial and is likely to be growing due to raised demand. Since the species is restricted to fragmented subpopulations, any depletion through either collection or habitat damage as a result of poaching risks bringing the subpopulation down to unviable numbers. The inclusion of *T. adalaidensis* in Appendix I will help emphasise that commercial exploitation of this species would be a very serious threatening process in case it becomes regular or widespread (Hutchinson, in litt., 2022).

Additional information

Threats

According to the IUCN Red List Assessment (2017), the primary threat to *Tiliqua adelaidensis* has been identified as the intensification of agriculture practices, and particularly the conversion of traditionally sheep-grazed land to cropland, which leads to the fragmentation of the subpopulations. The resulting large areas of unsuitable habitat prevent movement of individuals between subpopulations, which in turn results in inbreeding and loss of genetic diversity. Additional impacts come from pesticides and herbicides, which affect prey species, and land-use changes related to infrastructure (including road building).

Conservation, management and legislation

This species was listed in CITES Appendix III, effective 22 June 2022. *Tiliqua adelaidensis* is classified as endangered by both Australian national (Environment Protection and Biodiversity Conservation Act (EPBC) 1999) and South Australian state (Schedule 7 of the National Parks and Wildlife (NPW) Act 1972) legislation. This Act provides for fines for illegal possession and molestation of individuals. Specimens cannot be kept or traded as pets. Export of live individuals is only permitted for non-commercial purposes (e.g., exhibition, scientific research as outlined in the legislation) and prior to issuing of any export permit, the proposed export must demonstrate this purpose among other requirements. No such permits have been issued for the export of live specimens from Australia. At present, no examples of this species are held in private collections in Australia.

The species has been subject to a recovery programme since its rediscovery in 1992, *which included research efforts into the species' natural history* (Bull et al., 2015). An additional recovery plan that came into force under national environmental legislation in 2012 has aimed to identify management actions and research that would help stop the decline of the species. Furthermore, campaigns to raise awareness on conserving the species (including the related issue of habitat protection) have been carried out in local communities over the years.

The subpopulations of *Tiliqua adelaidensis* are wholly conservation dependent—in fact, all known subpopulations require active land management with appropriate grazing regimes to ensure their survival. To a vast extent, this currently occurs as an incidental result of sheep grazing, which maintains suitable habitat.

Captive breeding

The first captive breeding of *T. adelaidensis* occurred in early 2015 at Monarto Safari Park. The captive population, still viable, is used for research to understand further the biology and captive husbandry requirements of this species. No captive breeding for commercial purposes (i.e., with the aim to trade captive-bred specimens) takes place as it is prohibited under Australian legislation.

Implementation challenges (including similar species)

There are no similar species to *Tiliqua adelaidensis*. Australia has listed a number of other *Tiliqua* species and subspecies in Appendix III. Other species of *Tiliqua* are much larger and have markedly different colour patterns. Morphological similarity to some species of *Cyclodomorphus* would not be an identification issue for CITES export regulators at the Australian border since live specimens of native reptiles cannot be exported for commercial purposes and other non-commercial international trade in native reptiles requires export permits under Australia's national environmental legislation.

Potential risk(s) of a listing

No potential risks linked to the proposed amendment are identified.

Potential benefit(s) of listing for trade regulation

Tiliqua adelaidensis has only been found in natural grasslands that are dominated by native grasses and shrubs—this habitat supports populations of mygalomorph (trapdoor) and lycosid (wolf) spiders that dig vertical burrows with a single entrance (Bull et al., 2015). Once abandoned by the spider, the holes last for some years and become important refuges for numerous small creatures, amongst which *T. adelaidensis* is one of the largest and most specialised (Hutchinson, in litt., 2022). The burrows function both as a refuge from predators, extreme weather conditions, and grass fires, and as an ambush point from where the lizards can prey on passing insects (Nielsen and Bull, 2017). For burrowing spider populations to be sustained within native grasslands, the soil needs to have no recent or long-term soil disturbance (Delean et al., 2013). Moreover, the spiders themselves take several years to mature, so that new holes only accumulate relatively slowly (Hutchinson, in litt., 2022). The illegal trade of *T. adelaidensis* would almost certainly involve destruction of the species' spider hole hiding places (Hutchinson, in litt., 2022). Therefore, *T. adelaidensis* subpopulations are critically linked to the populations of spiders that build the burrows, and since no protective measures are currently in place for these species, the uplisting of the *T. adelaidensis* to Appendix I will likely constitute a potential benefit for the spider populations.

Other comments

Collection from the wild is likely to involve digging the animals out of the spider holes which the species uses as refuge. This would result in the permanent removal of the hole itself and therefore it becomes unusable in the

future. Loss of the essential refuge, coupled with relatively small litter size and long generation time (i.e., six years), mean that impacted subpopulations are likely to be slow to recover. Therefore, destruction of habitat linked to illegal harvesting represents a further threat to the species, and since most subpopulations register low numbers of isolated individuals, even low levels of collection from the wild can have significant effects on the survival of the species. An Appendix I listing would reinforce the national legislation and ensure the co-operation of other Parties.

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Transfer of Puerto Rican Boa *Epicrates inornatus* from Appendix I to Appendix II

Proponent: United States of America

Summary: *Epicrates inornatus* (now more widely known as *Chilabothrus inornatus*) is an adaptable semi-arboreal, non-venomous snake endemic to Puerto Rico, where it has a wide distribution. The population, currently considered stable at around 30,000 individuals, has recovered from historical declines attributed in part to deforestation in the early 20th century. This is thought to be due in part to the increase in forested areas in Puerto Rico. The species was assessed by IUCN (as *C. inornatus*) in 2015 as Least Concern on the basis of a large distribution, a lack of widespread threats, and an ability to inhabit altered environments. It is legally protected in Puerto Rico.

The species has been listed in CITES Appendix I since 1977. No exports of the species have been reported from Puerto Rico since the listing. Limited trade (192 items) in *Epicrates inornatus* has been reported by other Parties in the period 1975–2014, but none of these was reported to be wild-sourced or to have originated in Puerto Rico. No exports of the species have been reported in the CITES Trade Database by any Party since 2014.

The Virgin Islands Boa *Chilabothrus granti* (IUCN Endangered, assessed 2015) and Red-tailed Boa *Boa constrictor* (IUCN Least Concern, assessed 2014) also occur on Puerto Rico, as does an introduced population of reticulated python *Malayopython reticulatus* (IUCN Least Concern, assessed 2011). All three species are included in the CITES appendices, with *C. granti* in Appendix I and *B. constrictor* and *M. reticulatus* in Appendix II. There are no reported exports in the CITES trade database of these species from Puerto Rico in the most recent years of reported trade (2008–2018) from the USA.

This proposal is the outcome of a review conducted by the United States of America and considered by the 27th meeting of the Animals Committee (2014) as part of the *Periodic Review of the Appendices* under Res. Conf. 14.8 (Rev. CoP17). The Committee agreed that the species did not meet either the biological or trade criteria for inclusion in Appendix I and asked the United States to submit a proposal for consideration at CoP19 to transfer the species to Appendix II.

Analysis: The Puerto Rican Boa is classified as Least Concern on the IUCN Red List in 2015. It is widely distributed within Puerto Rico and has a stable population, with a recent estimate of 30,000 individuals. It does not have a small, or declining population, nor does it have a restricted distribution. No trade has been recorded from Puerto Rico since its listing in 1977 and no international trade in the species from any CITES Party has been reported since 2014, indicating low demand for the species, which remains legally protected in Puerto Rico. It does not meet the biological or 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 Puerto Rican boa. Transfer of the species to Appendix II is in line with the Precautionary measures in Annex 4 of Res. Conf. 9.24 (Rev. CoP17). This proposal is supported by the Animals Committee.

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 has been listed in Appendix I since 1975.

Taxonomy

Epicrates inornatus has the scientific synonyms *Boa inornata* and *Chilabothrus inornatus*. The IUCN Red List assessment utilises the synonym *C. inornatus*.

Range

Puerto Rico, USA

IUCN Global Category

Least Concern (assessed 2015, ver 3.1) as *Chilabothrus inornatus*

Biological criteria for inclusion in Appendix I**A) Small wild population**

The species is considered more abundant than previously thought at the time of listing in 1970. *Numbers have declined in the past, but this boa is still relatively common in many areas and occurs in several protected areas (Rodriguez et al., 2018).* A recent population model for the Puerto Rican Boa suggests a current island-wide estimated population size of more than 30,000 individuals.

B) Restricted area of distribution

This species is considered a habitat generalist with a wide distribution in Puerto Rico. *The species can inhabit moist forests, dry forests, karst landscapes, caves, and additionally altered environments including plantations, rural gardens, and urban areas (Rodriguez et al., 2018).*

In general, the Puerto Rican Boa is considered to have recovered from the historical deforestation in the early 20th century as lands once used for agriculture have reverted to secondary forest, with a subsequent increase in forested areas in Puerto Rico. *The species's most recent classification as Least Concern by IUCN is in part due to its large distribution range (Rodriguez et al., 2018).*

C) Decline in number of wild individuals

In general, there is a lack of long-term population studies to compare past versus present population estimates. The Puerto Rican Boa is probably more abundant today than previously thought at the time of listing in 1970, in part due to the increase in forested areas in Puerto Rico. Although the actual population trend of the Puerto Rican Boa is unknown, the 2011 Puerto Rican Boa five-year status review described this species' status as stable based on its broad distribution and apparent abundance.

Trade criteria for inclusion in Appendix I**The species is or may be affected by trade**

During the period 1975–2014, there was no record of international trade of *this species* out of Puerto Rico reported in the CITES Trade Database.

Reports from the CITES Trade Database show there was international trade reported from other non-range States in this time frame, with 195 items reported by exporters. The majority of these items (74%) were live individuals, with 24% bodies and 2% specimens.

Just over half of exporter reported items were associated with an unknown source code. Out of 87 exporter reported items associated with a source code, 90% were reported as captive-bred, with the remaining either captive born or pre-convention. The majority (85%) of exporter reported trade was prior to 2000. Between 2014 and 2021 no further trade in this species has been recorded.

The US Fish and Wildlife Service is aware of a reptile breeder in Florida that has offered Puerto Rican Boas for sale on his website, however, this breeder only sold these boas within Florida and not interstate or internationally. There have also been a few reported cases of locals collecting Puerto Rican Boas for sale within the island through the online classified reptile section. One person was caught and fined accordingly. *A preliminary search for online advertisements using common names for the species in English and Spanish found three advertisements for the species with a total of eight individuals. Two were offered for sale in 2008 by one seller on a platform hosted in Canada and one was offered for sale in 2017 from a seller stated to be in California. The seller in California stated that sales were possible only within California and not interstate. All three advertisements stated that the individuals were captive-bred.*

There is no additional information to suggest that the Puerto Rican Boa has been or is being significantly impacted by trade. *There are no records of seizures in this species in either EU-TWIX or WiTIS.*

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**

There is no evidence of an unsatisfied or unreported demand for subsistence uses or commercial trade in this species and a low volume of international trade. *Future exploitation is not expected to increase, given the difficulty*

in accessing areas where the species occurs and a lack of commercial incentives for international trade in the species (AC27 Doc 24.3.7). Trade in other similar species is non-existent.

With extremely low trade volumes and demand for this species and similar species in Puerto Rico, it is unlikely that the transfer of this species from Appendix I to Appendix II will stimulate trade, or cause enforcement problems for any other CITES-listed species.

Additional information

Threats

The major impacts on this species are habitat loss and fragmentation from human development, predation from exotic mammals (namely domestic cats *Felis catus*), and poaching and intentional killings.

It is also adversely affected by inappropriate management practices when translocating and handling Puerto Rican Boas, emergent diseases (i.e., snake fungal disease), hurricanes, and climate change. These involve a variety of impacts, which reduce or degrade available habitat and may have direct impacts on the species, for example, mortality from roadkill and human persecution.

The last IUCN Red List assessment, conducted in 2015, mentioned capture for the pet trade as a threat (Rodriguez et al., 2018). The species has also been said to be locally threatened by illegal hunting for meat for human consumption and oil as a medicinal remedy (AC27 Doc 24.3.7). The practice of hunting or capturing Puerto Rican Boas may still occur, but probably to a lesser degree. The extent or effect of illegal hunting is uncertain.

Conservation, management and legislation

Within Puerto Rico, the species is included in the list of protected species classified as vulnerable by the Puerto Rico Department of Natural and Environmental Resources (PRDNER), whose criteria are based on the IUCN Red List criteria (AC27 Doc 24.3.7). It is currently listed on the United States Endangered Species Act as endangered (AC27 Doc 24.3.7) but the US Fish and Wildlife Service has proposed that it be removed from the Endangered Species Act as it no longer meets the definition of a threatened species (USFWS, 2022).

Safeguards in the form of national laws and regulations regarding wildlife conservation are already in place in Puerto Rico for *E. inornatus*; the boa is protected under Puerto Rican Law, the US Endangered Species Act, and CITES. In 2004 PRDNER approved Regulation 6766 to Regulate the Management of Threatened and Endangered Species in Puerto Rico. This regulation explicitly prohibits the possession, transportation, taking, destruction, hunting, and killing, of any wildlife species listed as threatened or endangered, which includes *E. inornatus*.

The species occurs within several protected areas in Puerto Rico.

Captive breeding

According to the International Species Information Systems-Zoological Information Management System, there are *E. inornatus* individuals in one zoo in Europe and nine zoos in the USA. D. Barber, Curator of Ectotherms at Fort Worth Zoo, commented that the species is fairly easy to breed, but institutions do not breed them regularly. Apparently in captivity individuals tend to be aggressive and are probably not a priority for captive breeding. In Puerto Rico, there is at least one organised group (Puerto Rico Reptiles, Inc.) that promotes keeping reptiles as pets and may be an interested stakeholder for Puerto Rican Boa trade.

Implementation challenges (including similar species)

There are two native Boidae species on the island, the Puerto Rican Boa and the Virgin Islands Boa *Chilabothrus granti*. The Virgin Islands Boa in the main island of Puerto Rico has a very limited range where they co-occur with the Puerto Rican Boa and could be mistaken by the general public. There is an established population of Red-tailed Boa *Boa constrictor* and a relatively recent invasion of Reticulated Python *Malayopython reticulatus* in Puerto Rico, and although these species are distinct in size and colour from the Puerto Rican Boa, they could still be mistaken by the general public.

All three of these similar species are listed in the CITES appendices: *Chilabothrus granti* is listed under the synonym *Epicrates monensis* in Appendix I and *Boa constrictor* and *Malayopython reticulatus* (under the synonym *Python reticulatus*) are in Appendix II. According to the CITES Trade Database, there are no reported exports in these species from Puerto Rico in the most recent years of reported trade from 2008–2018. Given that there is low demand and trade in all three similar species from Puerto Rico, it is unlikely that the transfer of this species from Appendix I to Appendix II will lead to CITES implementation challenges.

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

It appears that transfer from CITES Appendix I to II protection would not have any conservation impact on this species and would not be expected to affect the nature of the trade.

The national and Puerto Rican legal protections for this species discourage any commercial incentives to engage in international trade in this species.

Potential benefit(s) for trade regulation of a transfer from Appendix I to II

CITES Appendix II listing would still allow the United States and other Parties to regulate and monitor any trade in *Epicrates inornatus*.

References

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Inclusion of Timber Rattlesnake *Crotalus horridus* in Appendix II

Proponent: United States of America

Summary: The Timber Rattlesnake *Crotalus horridus* is a terrestrial long-lived ectotherm that inhabits a variety of habitats including temperate forest, inland wetlands, pastureland and rocky areas. The species is morphologically distinct from other rattlesnake species due to the presence of dark zigzag patterns on its back. The species is native to the USA where it is known to be extant in 21 states and is now extinct from its marginal range in southern Canada. The proponent makes a case for Appendix II listing to combat unsustainable use and illegal trade. The same proposal was submitted to both CoP10 and CoP11 in 1997 and 1999 respectively and withdrawn on both occasions.

The species was last assessed for inclusion on the IUCN Red List in 2007 and classified as Least Concern due to a wide distribution and presumed large population. The population size is not currently known but an assessment by NatureServe in 2014 stated that it was presumed to be at least 100,000 individuals and “Apparently Secure” across its range and common in some areas, despite the species being assessed as threatened in 23 (74%) states. The species has been extirpated from the states of Maine and Rhode Island in the USA. Prominent threats to the species identified by NatureServe, the IUCN Red List assessment, and a recent survey conducted across the USA are habitat loss and fragmentation, mortality from roadkill, hunting, and persecution. The species is known to be hunted domestically in unknown volumes during recreational rattlesnake roundups, and is also used in the tradition of serpent handling in Appalachian churches. The species is sold within the USA live and in the form of skins, venom, and novelty items including taxidermy specimens.

No global trade data exist, but the US has recorded exports of only 47 individuals between 2010 and 2015, the majority as captive sourced. No trade has been reported since 2015. Imports to the US from Germany of captive-bred specimens (mainly liquid forms of medicinal parts and products) between 2010 and 2020, suggest that there may be some captive breeding of the species for commercial purposes outside its range. Very few records of online advertisements have been found outside the USA, but included some for live individuals and for homoeopathic remedies containing extremely diluted quantities of snake venom in India. There is anecdotal evidence that live specimens of the species may be sold at reptile shows in Europe for over USD800, but this is based on information from one seizure in Florida in 2013. There are also reports of offers for sale at a reptile show in South Africa, but it is not clear how recently this was observed and whether the animals were wild-sourced.

International trade was not identified as a prominent threat to the species by the IUCN Red List nor the NatureServe 2014 assessments. According to the NatureServe 2014 assessment, the species was in demand for use as pets and in private collections, but it is not clear if this was internationally or domestically.

Harvest of the species is now prohibited in 18 out of 31 states in which it is extant, and some populations of the species inhabit protected areas.

Analysis: This species was most recently assessed as Least Concern on the IUCN Red List in 2007 and “Apparently Secure” across its range by NatureServe in 2014. The most recent status information available suggests that the population size was at least 100,000 and that a short-term decline of over 10% over three generations was possible. The species has been subject to considerable human induced mortality. Any trade appears to be predominantly domestic. Available evidence shows that international trade is minimal and is unlikely to be having a significant impact on the species’ population in comparison to other effects. There is no indication that regulation of international trade is necessary and therefore the species does not appear to meet the criteria for listing in Appendix 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

Crotalus horridus (Linnaeus, 1758).

The CITES Standard reference is McDiarmid, R.W., Campbell, J.A., and Touré, T.A. (1999). Snake species of the world: a taxonomic and geographic reference. Volume 1. Herpetologists' League. Washington, DC.

Range

Extant (resident): USA

Extinct: Canada (Hammerson, 2007)

IUCN Global Category

Least Concern (assessed 2007, ver. 3.1)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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 size

The species was assessed as Least Concern on the IUCN Red List in 2007 due to its wide distribution, presumed large population, and because it was unlikely to be declining fast enough to qualify as threatened. *In 2014, NatureServe stated the population size is unknown but was presumed to be greater than 100,000 and classified the species as "Apparently Secure" across its range (NatureServe, 2014). The NatureServe assessment stated that the species is still fairly common in parts of its range (NatureServe, 2014). Updated estimates for C. horridus populations could not be identified and updated population monitoring research for this species is needed.*

A total of 23 out of 31 states in the USA with extant populations consider them at population numbers that constitute a threatened status. *Based on the information presented by the proponent from a combination of the NatureServe assessment in 2014 and updated sources, out of the states in which populations are considered threatened, 22% are classified as endangered and 74% vulnerable. In Virginia, the species is assessed as two populations, with the mountain populations assessed as apparently secure and the coastal populations as endangered. The species is extirpated in the USA states of Maine, Rhode Island and from the entirety of its historical range in Canada.*

Available state-level population trend details are provided by five states: Virginia, New York, Massachusetts, New Hampshire, and Connecticut. NatureServe (2014) reported an observed long-term global population decline of 30 to 50% and a *short-term decline of possibly more than 10% over 3 generations (20 to 30 years).*

Crotalus horridus was once one of the most wide-ranging North American rattlesnake species; however, current populations are now fragmented.

International trade

The USA submitted proposals at CITES CoP10 in 1997 and CoP11 in 1999 to include the species in Appendix II (see CoP10 Prop. 10.63 and CoP11 Prop. 11.44). These proposals were subsequently withdrawn on both occasions. At CoP10, the Netherlands (on behalf of EU Member States) stated that conservation problems for C. horridus did not seem to be caused by international trade and recommended listing the species in Appendix III (CoP10 Summary Report of the Committee I Meeting).

The Proponent states that "It is inferred that C. horridus is affected by trade." The IUCN Red List assessment in 2007 and the NatureServe assessment in 2014 state that commercial collection for the pet trade is a threat but do not specify if this is for domestic or international use (Hammerson, 2007; NatureServe, 2014). The NatureServe assessment also refers to a decline in commercial collections for the pet trade in recent years (NatureServe, 2014).

The behavioural characteristic of communal denning, *where entire populations undergo communal hibernation during winter months (NatureServe, 2014), reportedly makes it particularly easy for large numbers of around 40 to 550 individuals to be collected at one time. One expert commented that this behaviour occurs only in northern populations, which are therefore more vulnerable to collection (Spear, in litt., 2022). Collection from the wild to supply the pet trade is reportedly a threat in most states (74%) in which the species is extant in the USA. Domestic commercial use of C. horridus is observed in the live pet trade, skin trade, venom trade, in recreational rattlesnake roundups, and for sale as "novelty" items (e.g., for taxidermy and jewellery). A study in Texas in 1994 concluded that the impact of rattlesnake roundups on populations of the species could not be quantified due to its unregulated nature (Adams et al., 1994), and a more recent study in 2000 was also unable to identify harvest*

volumes of the species for the sport (Fitzgerald and Painter, 2000). Though largely considered an antiquated practice, rattlesnake hunts and roundups in which it is not required to return the snakes to the wild are still practised in Texas, Oklahoma, and Alabama. One source stated that roundups no longer occur in Georgia and that in Pennsylvania they still occur but with regulations, including a requirement for permits to harvest rattlesnakes, and for the event to have educational merit (Sawyer, in litt., 2022). The same source stated that across the southern range of the species, roundups are likely mostly unregulated. Live *C. horridus* are also still used by Christian “serpent-handling” churches in Appalachia.

USA export data for 2010–2020 from LEMIS show that a total of 47 live individuals have been reported in cleared direct exports from the USA to other countries for commercial purposes, with all of these occurring between 2010 and 2015. The majority of live individuals were reported in exports in 2010 (25) and 2013 (10) with fewer than five individuals exported per year in 2011–2012 and 2015–2015. Close to half (43%) of these exports were to Mexico between 2010 and 2012, with Germany and Thailand the next most prominent importers (15% of individuals each) followed by Japan (11% of individuals). Most of the live individuals exported (79%) for commercial purposes were reported to be captive-bred and no wild-sourced individuals were reported in exports since 2014. Two bodies, 200 skin pieces, six shoes, and 45 specimens were also reported in cleared exports for commercial purposes between 2010 and 2020.

The proponent states that live animals, dead animals, museum and research specimens, and derivatives (e.g., venom extracts, medicinal products, skeletons, skins, and trophies) are known to be in international trade. An analysis of LEMIS data for 2010–2020 shows that only small volumes of commodities including bodies, live individuals, shoes, skin pieces, and specimens of the species have been exported for commercial purposes.

There is anecdotal evidence that the species is for sale online and in physical markets. The species is said to be readily available for purchase online for as much as USD250 with no information regarding the origin of the snake. A rapid search of online advertisements conducted by TRAFFIC in August 2022, found 24 online advertisements for commodities from the species; most of these advertisements (75%) were located in the USA with four in India, one in the Czech Republic, and one in Germany. One advertisement did not have a location. Advertisements located in India were all for homoeopathic remedies containing extremely diluted forms of the species’ venom. Stated uses of homoeopathic products in advertisements ranged from vision defects and insect bites to relief from nosebleeds and weakness in the elderly and prices ranged from USD1–USD11 per bottle. It was not stated in any advertisement if the snake venom originated from wild-sourced or captive-bred individuals. Both advertisements in Europe were for live individuals, with three individuals offered for sale in Germany and one in the Czech Republic. Both stated the individuals to be captive-bred and did not mention prices. Most of the advertisements (65%) located in the USA were for skins or products from skins with only two for live individuals and no advertisement specified the source of the product. Prices in US advertisements ranged from USD35 for homoeopathic medicine to USD795 for a taxidermy mount. Prices of live individuals offered for sale were lower than in the example stated by the proponent, with both live individuals offered for sale for USD95.

Demand for *C. horridus* has also been shown in the South African pet trade where Timber Rattlesnakes were reported to be offered for sale at a local reptile show. In 2013 a man in Florida was convicted for illegally purchasing and transporting 20 protected, wild-caught north-eastern *C. horridus* across state lines. Evidence from the trial showed the snakes were destined for the European pet trade where a single Timber Rattlesnake can sell for over USD800 at reptile shows. There are no records of seizures of this species in the WiTIS database from 2008–2020, or in EU-TWIX in 2022.

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
- B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Additional information

Threats

One expert stated that trade is not a threat to the species, with roadkill and persecution prominent threats instead (Fitzgerald, in litt., 2022). Another expert commented there is not sufficient evidence that poaching and trade were driving populations to extinction but also commented that illegal snake trade is difficult to track (Spear, in litt., 2022). The same expert commented that if there was undocumented trade occurring for the species, it was unlikely to have the potential to drive declines in most southern populations but could impact upon northern populations.

The primary threat identified in the 2014 NatureServe assessment was land development and habitat destruction (NatureServe, 2014). The IUCN Red List assessment in 2007 also stated that primary threats were loss and fragmentation of habitats, which can lead to isolated and nonviable populations, in addition to mortality arising from illegal snake hunting and roadkill. Habitats are increasingly becoming fragmented by roadways and residential development, as well as agricultural development.

A survey of the primary threats to the species, completed by all states within the USA in the species' range, revealed roadways and road mortality as the largest threat (27 out of 31 states) to the species' survival. Poaching and illegal collection were the fourth after human development and persecution and were perceived to be a threat in 23 out of 31 states.

Conservation, management and legislation

Crotalus horridus is not listed or afforded direct national protection; however, 18 of the extant 31 USA range states directly prohibit harvest. Unfortunately, such statutes are often not enforced.

Translocated individuals and head-start wild-caught individuals continue to be studied to measure their ability to augment endangered northern populations. Population monitoring in the majority of the southern range states of *C. horridus* in the USA is lacking when compared to monitoring efforts in the midwest and northeast.

Some *C. horridus* populations are found in protected areas and state lands. For example, Pennsylvania has 900,000 ha of Forest State land, with the largest continuous blocks of *C. horridus* habitat in the northeast. National wetland protection laws, in accordance with the U.S. Environmental Protection Act (1969), indirectly protects portions of the rattlesnake's southern wetland habitat range. This helps to offset the lack of conservation measures in certain states of the USA, such as Florida.

Captive breeding

Captive breeding programmes for the conservation of *C. horridus* have proven difficult. Although the species is not considered "easy" to breed, it has been reproduced in a captive setting and a genetically managed *ex situ* population may be a reasonable conservation action. A Rhode Island zoo has one of the only known captive breeding programmes for the state-extirpated reptile and is the augmentation source (using neonates) for north-eastern head-starting efforts. According to the ZIMS 360 database for zoos, there are *C. horridus* individuals in 20 institutions (18 in the USA, 1 in Russia, and 1 in Cyprus) with no births in the previous year.

Although no commercial captive breeding centres could be found for this species, LEMIS reports cleared direct exports of captive-bred C. horridus specimens into the USA from Germany for commercial purposes between 2010 and 2020. This was mostly in medicinal parts or products (around 2,800 l), followed by extracts (10 l) and 12 live individuals. The largest annual volume of medicinal parts or products (over 1,000 l) was reported in direct exports in 2019.

Implementation challenges (including similar species)

Under most circumstances, *C. horridus* parts and derivatives are distinguishable from other similar species in trade. *An expert also commented that the species is not morphologically similar to any other rattlesnake (Spear, in litt., 2022).*

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Inclusion of Amazon Matamata *Chelus fimbriata* and Orinoco Matamata *Chelus orinocensis* in Appendix II

Proponents: Brazil, Colombia, Costa Rica, Peru

Summary: *Chelus* is a genus of distinctive, relatively large (30–50 cm) highly aquatic freshwater turtles found in South America, formerly considered to comprise a single species, the Matamata *Chelus fimbriata*. Recently, the splitting into two separate species, the Amazon Matamata *Chelus fimbriata* and the Orinoco Matamata *Chelus orinocensis*, has been proposed. The proponents include discussion of the implications of this split in their Supporting Statement. This split is not recognised by current CITES standard nomenclature and there is no proposal to update the standard nomenclature for *Chelus* species submitted for CoP19. In this analysis, the current CITES standard nomenclature for this genus is followed, which recognises only one species *Chelus fimbriata* in the genus. Were the proposal to be accepted, standard CITES nomenclature would apply to the listing.

The Matamata *Chelus fimbriata* occurs in aquatic habitats in water systems including rivers, lagoons and flooded forests in nine range States in South America (Bolivarian Republic of Venezuela (henceforth Venezuela), Brazil, Colombia, Ecuador, French Guiana, Plurinational State of Bolivia (henceforth Bolivia), Suriname, Guyana, and Peru)). The species has an overall range calculated at close to 7 million km² but is likely to be restricted to only certain habitats within this area.

The species has not been assessed on the IUCN Red List. Its global conservation status was assessed as least concern by the IUCN SSC Tortoise and Freshwater Turtle Specialist Group (TFTSG) in 2011. The species is reported to be affected by pollution, habitat loss and fragmentation as well as disturbance of nesting sites through developments. Quantitative population data are not available for any range State, although in Venezuela it has been noted as locally common in some areas and one study in a protected area in Colombia found a density of 2.3 per 10 m² indicating that the species may be at least locally abundant.

The species is in some international demand from turtle and aquarium enthusiasts. There is very little information on the scale of this demand, although prices in online fora are relatively high compared with other turtle species (an average of USD340 for sites in the USA and UK, and USD60–285 in China). Information from online platforms publishing advice on keeping reptiles as pets indicates that this species is not suitable for beginners due to factors including its large size and carnivorous diet. Data from the US and Peruvian authorities show that most legal reported exports originate from Peru, with no reported trade or seizure reports from four range States (Brazil, Bolivia, Ecuador, and France - French Guiana) that together make up more than 70% of the estimated global distribution. USA data show over 2,000 individuals in direct exports from Peru with most (95%) between 2015 and 2020. The majority (60%) were reported captive-bred with 520 wild-sourced individuals and the remaining ranched. Commercial export of wild-collected individuals of this species is illegal in Peru. National trade data from Peruvian authorities show exports of nearly 64,000 live individuals of unknown sources for commercial purposes between 2010 and 2020 with China (~40,000) and the USA (fewer than 15,000) as prominent importers. There is a discrepancy between these data and those of the USA, with Peruvian national authority data reporting 12,000 more individuals in exports to the USA than are reported as imported by the USA for the same time period. The reason for this discrepancy is unknown. The USA has also reported some imports from Guyana (around 700 of wild origin between 2010 and 2020) and Venezuela (around 600 reported captive-bred between 2008 and 2012).

Most seizures reported occur within Colombia with over 7,000 individuals reported between 2019 and 2020; in 2013 and 2014 nearly 500 wild-sourced live individuals from Colombia were refused entry to the US and seized. There are reports of illicit trade routes operating from Colombia, Brazil, and Venezuela (via Colombia) into Peru.

Export of the species is reported to be prohibited from Brazil, Colombia, and Venezuela despite records of import from these countries into the USA. There are reported to be numerous Matamatas in captivity outside range States. Although there are no authorisations for captive breeding of the species in Colombia, the proponents indicate that individuals seized in Colombia appear captive-bred due to their uniformity in size and physical characteristics.

Analysis: The Matamata is a very widely distributed species, classified in 2011 as least concern by the IUCN TFTSG, with a global distribution of several million square kilometres. It is said to be at least locally common within this range.

The species is in some demand in the exotic pet trade, chiefly on account of its distinctive appearance (although maintaining large adults in captivity apparently presents challenges) and it has been reportedly exported in some numbers (low thousands per year) from range States (mainly Peru, but also Guyana, Venezuela and possibly Colombia) in the past decade. The origin of the specimens in trade is generally unclear. Some are reported as wild-collected, while others are reported as ranched or captive sourced. There is no information on the impact of wild collection on populations. However, given its very extensive range and predominance of juveniles in trade. The species would therefore appear not to meet the criteria for inclusion in Appendix II set out in Res. Conf. 9.24 (Rev. CoP17).

Other Considerations: Given the likely reclassification of *Chelus fimbriata* as two separate species, if Parties were to consider it necessary to list the taxon, it may be preferable to list the genus for any future taxonomic changes to be accommodated.

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

In 2020, the description of a new species, *Chelus orinocensis*, which divides *C. fimbriata* into two species (Vargas-Ramírez *et al.*, 2020), implied that the wide distribution area considered for *C. fimbriata* in fact corresponds to two species with independent distributions and inhabiting different territories: *C. fimbriata* in the Amazon basin and Mahury River drainage, and *C. orinocensis* in the Orinoco basin, Rio Negro and Essequibo. These taxonomic changes were proposed following molecular studies.

The proponents have not submitted a proposal to CoP19 for an updated standard nomenclature for this species. In this analysis, the current CITES standard nomenclature for this genus is followed, which recognises one species Chelus fimbriata (Frost, 2007).

Range

Bolivia, Brazil, Colombia, Ecuador, French Guiana, Guyana, Peru, Suriname, Venezuela

IUCN Global Category

This species has not been assessed on the IUCN Red List

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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 size

No quantitative data are available on the population size of this species, however, in the areas where it is distributed its sightings are said to be very rare, and when they do occur, it is often observed alone. The total distribution of the species is 6,900,551 km² which is one of the largest distributions of any turtle species in the Amazon basin.

In 2011 the IUCN TFTSG assessed the global conservation status of *Chelus fimbriata* and considered the species to be in the category of least concern, however no justification for this assessment was provided. Based on its large distribution range, the species is not considered to be under immediate threat.

A study in 2021 reported 182 new occurrence records for the species in five out of nine range States, with most of these restricted to elevations of less than 200 m and in water systems less than 1 m in depth. Half of all new occurrence records were in Brazil (50%), followed by Colombia (25%), and Venezuela (24%) (Cunha et al., 2021).

In the absence of population estimates, all available evidence on the distribution or population densities of the species is presented below.

Bolivia: The most recent distribution area calculated from all known occurrence records is 768,661 km² (Cunha et al., 2021).

Brazil: The most recent distribution area calculated from all known occurrence records is 3,790,347 km² (Cunha et al., 2021).

Colombia: Research conducted between 2013 and 2020 in the Bojonawi Nature Reserve, Colombia, estimated a population density of *C. fimbriata* (recognised as *Chelus orinocensis* by the authors) of 2.3 turtles per 10 m², with evidence of an increase in the population by 2020 (Lasso et al., 2020). The species was the second most abundant out of seven species involved in the population study. The authors also noted *C. fimbriata* is categorised as least concern in the Colombian National Red list. The most recent distribution area calculated from all known occurrence records is 677,644 km² (Cunha et al., 2021).

Ecuador: The most recent distribution area calculated from all known occurrence records is 66,233 km² (Cunha et al., 2021).

French Guiana: The most recent distribution area calculated from all known occurrence records is 55,407 km² (Cunha et al., 2021).

Guyana: The most recent distribution area calculated from all known occurrence records is 155,835 km² (Cunha et al., 2021).

Peru: The most recent distribution area calculated from all known occurrence records is 690,735 km² (Cunha et al., 2021).

Suriname: There are no occurrence records for this country (Cunha et al., 2021).

Venezuela: Populations in Venezuela appear to be stable. In some localities on the plains of Venezuela, the species is believed to be common. The most recent distribution area calculated from all known occurrence records is 680,732 km² (Cunha et al., 2021).

International trade

The species is said to be collected mainly at the hatchling and juvenile stages for international trade as pets amongst turtle and aquarium enthusiasts. Key demand regions are Europe, the USA, and China. The species can reach up to 50 cm in length. One online platform in the USA providing information on keeping reptiles as pets advises these are “not an advisable pet for beginners” due to their height and weight (Reptiles Cove, 2022). An online platform in China lists the species amongst the “top ten most difficult to raise water turtles” due to factors including a need for a dark environment, strict pH and temperature requirements, and a diet of live food. (Maigoo, 2022). Another platform providing advice on keeping turtles classifies the experience level for this species as intermediate, stating the species may reach weights of up to 15 kg and is carnivorous with a diet of fish and aquatic creatures. This platform states the average price range for an individual is USD250–USD400 (All Turtles, 2022).

The extent of collection and its impact is unknown, with illegal trade also said to be poorly documented in Neotropical countries. Two experts stated that it is believed females are caught for the purposes of establishing captive breeding centres (Cunha and Vargas-Ramírez, in litt., 2022). One expert stated that they believed trade was a threat to the species but that further research on populations was needed to determine the magnitude of the threat (Lasso, in litt., 2022). Another stated that reported seizures in Colombia may represent only a fraction of the volume of illegal trade in the species and that a lack of knowledge regarding the total magnitude of trade was a barrier to assessing if trade is a threat to the species (Vargas-Ramírez, in litt., 2022).

The species is said to be collected for the pet trade in Colombia and Venezuela (Cunha et al., 2021). There is thought to be a trade network that operates by transporting specimens from Colombia to Peru where they can be legally exported for commercial purposes. An expert also stated that it is suspected that the species also enters Peru from the Brazilian Amazon, and from Venezuela, via Colombia (Lasso, in litt., 2022).

A brief preliminary search found seven online advertisements for at least 18 individuals (TRAFFIC, 2022). Prices for individuals offered for sale ranged from USD225–USD500 with an average of USD340. Six of the advertisements were from sellers located in the USA, with one from a seller located in the UK. Three advertisements specified the source of the individuals offered for sale, with two stating them to be “captive hatched” in the US and one “captive raised” in Peru. An additional preliminary search for advertisements in China found the species for sale on two online platforms (Xu, in litt., 2022). One platform had 28 traders, with two of them claiming monthly sale volumes of 33–34 individuals and prices ranging from USD60–USD285. The origin and source of the individuals were not stated, although most of the turtles in the images provided appeared to be juveniles.

Available information on legal trade from LEMIS between 2010 and 2020 and illegal trade through seizures for each range State is presented below.

Colombia

In Colombia the export of Matamatas is said to be prohibited. According to LEMIS, a total of 475 wild-sourced live individuals imported into the USA for commercial purposes in a direct export from Colombia were refused entry and seized. Most (450) of these were reported in 2013, with the remaining 25 reported in 2014.

There is evidence from the Supporting Statement of a total of 3,295 individuals seized in two incidents in Colombia. In 2019, the Environmental and Ecological Police and the Airport Police seized 1,359 individuals in Bogotá that were to be transported by parcel from Bogotá to Leticia. In 2021, 1,936 individuals, which were intended to be transported to the south of the country, were seized in El Dorado airport in Bogotá. Harvest of the species in Colombia has been reported to occur in the Orinoco basin. According to seizure data from the environmental authority in the Colombian Amazon (Corpoamazonia), between 2013 and 2018 there have been significant volumes of hatchlings arriving by air to the city of Leticia, close to the border with Peru. These seizures suggest the existence of an exit route for individuals using a route from Bogotá to Leticia. From Leticia, it is thought that individuals would be transported to Peru where the species can be legally exported (Lasso et al. 2018). Four seizures of the species in Colombia totalling 3,810 individuals are reported in TRAFFIC’s wildlife trade information system (WiTIS) between 2010 and 2020, with most (81%) of these seized between 2019 and 2020, although it is not clear what their intended destination was.

Guyana

According to reports in LEMIS, a total of 695 live wild-sourced individuals were cleared for import into the USA for commercial purposes originating from Guyana, with almost all of these direct exports. There was an average of 62 individuals per year originating from Guyana, with the highest annual volume of 120 individuals reported in 2016.

Peru

According to evidence presented by the proponents based on export databases from the Peruvian National Forestry and Wildlife Service SERFOR and Regional Government of Loreto, there were legal exports of 63,612 individuals for commercial purposes between 2010 and 2020. Most (83%) of these were exported in the five years 2016–2020. The exports increased from 749 live individuals in 2010 to 18,355 in 2018. The data also show that there has been a reduction in the number of individuals exported since 2018, with 8,366 in 2019 and 4,472 in 2020. These individuals are reported to be sourced from the wild in the form of ranching and from production in authorised zoos but the source codes for the individuals exported are not available in the data.

According to the export databases from Peru, the main export destinations for *C. fimbriata* are China and the USA, which together account for the majority (86%) of exports since 2010. China dominated the market between 2012 and 2018, and in 2019 the USA imported 58% of all *C. fimbriata* exports from Peru.

Data sourced from cleared imports in LEMIS for the same time period 2010–2020 show direct exports of 2,077 live individuals for commercial purposes originating from Peru, of which the majority (58%) were captive-bred, with 25% wild-sourced and the remaining ranched. It is illegal to export wild-sourced individuals of this species for commercial purposes in Peru and it is not clear if the reported wild-sourced individuals were incorrectly declared in documentation, or illegally exported. The volume of imports reported in LEMIS is around 12,500 less than the reported 14,660 exports of live individuals for commercial purposes reported in data from the Peruvian authorities (SERFOR database and the Government of Loreto) in the Supporting Statement. The volume of imports per year is relatively low, with 42 individuals in 2010, none between 2011 and 2013, and 72 in 2014; from 2015 onwards, there has been an average of 330 individuals a year with the highest volume of 401 reported in 2016.

In the period 2001–2020, there have been 46 seizures in Peru, in which a total of 1,000 live individuals have been seized.

Suriname

According to reports in LEMIS, a total of three wild-sourced live individuals cleared for import into the USA for commercial purposes originated in Suriname, all of which were reported in 2018.

Venezuela

According to LEMIS data, a total of 621 captive-bred individuals cleared for import into the USA for commercial purposes were directly exported from Venezuela. Most of these (585) were reported before 2012. There were no imports from Venezuela between 2012 and 2019 and 36 reported in 2020.

Table 1 presents a summary of trade data for each range State above; evidence of legal international trade or seizures is available for five out of eight range States with evidence of wild-sourced international trade available for three range States.

Table 1. Summary of trade data from range States for the Matamata. Sources: * SS = proponents supporting statement, T = Records in TRAFFIC's wildlife trade information system (WiTIS), P = Data presented by proponents from the Peruvian National Forestry and Wildlife Service SERFOR and Regional Government of Loreto, L = LEMIS

| Range State | Evidence from seizures (P/W*) | The number of exports of live individuals for commercial purposes 2010–2020 originating from the range State (source, P/L*) |
|---------------|-------------------------------|---|
| Brazil | | |
| Bolivia | | |
| Colombia | 3,295 (SS) 3,810 (T), 475 (L) | |
| Ecuador | | |
| Guyana | | 695 (wild-sourced, L) |
| French Guyana | | |
| Peru | 1,000 (SS) | 63,612 ⁺ (unknown, P) 1,204 (captive-bred, L) 343 (rancher, L) 530 (wild-sourced, L) |
| Suriname | | 3 (wild-sourced, L) |
| Venezuela | | 621 (captive-bred, L) |

+ Data from the Peruvian National Forestry and Wildlife Service SERFOR and Regional Government of Loreto does not report if individuals originated from Peru or other range States.

Non-range States

Small volumes of reported imports in LEMIS were from direct exports from non-range States: Hong Kong SAR (2018) and Malaysia (2017) each reported one export of a captive-bred individual, whilst the Netherlands reported exports of five wild-sourced individuals in 2018. Hong Kong SAR additionally reported the re-export of three wild individuals that originated in Guyana in 2013.

A survey of turtle sales in Hong Kong SAR on social media, online forums, and in physical markets from 2017–2018 found 339 individuals for sale, with most (89%) of these in physical markets (Sung et al., 2021). The origin or source of the individuals was not stated.

Six seizures of the species were recorded in WiTIS between 2010 and 2020 for non-range States in Asia and Central America: Malaysia (1), Philippines (2), Thailand (1), and Mexico (2). A total of 37 individuals were seized with the origin of the species not known.

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**
- B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved**

Additional information**Threats**

Threats identified include the construction of the Meta and Amazonian waterways on the Orinoco and Amazon rivers, which could lead to reductions in the availability of nesting beaches. Other threats include habitat loss and fragmentation, and human activities leading to environmental pollution.

The species is used for human consumption as food by local communities in range States including Colombia, Venezuela, and occasionally Peru, where consumption of the shell for medicinal purposes by one community has also been recorded. Both meat and eggs from the species are consumed. *The volume of individuals harvested for local consumption is not clear.*

Conservation, management and legislation

Brazil

The export of the species is prohibited.

Ecuador

The species inhabits two protected areas.

Peru

The species inhabits six protected areas. Commercial trade is permitted for ranched and captive-bred individuals. Until 2015, commercial exports required a permit issued by the National Forestry and Wildlife Authority. Since the introduction of the new Forestry and Wildlife Act in October 2015, species not listed in the CITES appendices are exempted from needing a permit. Those breeding the species for commercial purposes have to submit a management plan for in situ (e.g., in managed wild areas) or ex situ (e.g., in zoos) production of offspring to the Regional Forestry and Wildlife Authority. There are no requirements to report on factors such as genetic stock, collection rates and health care. There are no national management plans for the species or established quotas for its export from Peru.

Venezuela

The species inhabits three protected areas. The export of the species is prohibited.

Colombia

In Colombia, where the species is consumed locally, a project funded by the Global Environmental Facility aims to generate information that can be used to design management measures to ensure sustainable use of the species. Indigenous communities monitor both the use of the species and the size of populations of *C. fimbriata* (recognised by the proponents as *C. orinocensis*) along the Guaviare River. *In 2015, the Alexander von Humboldt Biological Resources Research Institute published a document with strategies for the conservation of continental turtles of Colombia between 2015 and 2020, inclusive of this species. The strategies were developed in a workshop with 27 researchers from institutions including NGOs, Governmental organisations, Universities and research institutes (Morales et al., 2015).* The export of the species is prohibited.

No information is available on programmes for monitoring the status of wild populations in any range State, or for assessing the feasibility of harvesting species from the wild. The species is not currently listed in any category of global or national threat in any of the range States. Information is not available for all range States on other protection measures and it is not clear whether the export of the species for commercial purposes is legal in Bolivia, Ecuador, French Guiana, Guyana, and Suriname.

Captive breeding

Outside of the range States, there are said to be numerous Matamatas in captivity. In 1982 several hundred juveniles were illegally exported from Guyana to the USA, where they were seized and deposited at the Los Angeles Zoo and redistributed to other collections.

Anecdotal evidence from the proponents indicates that although seized species from Colombia may be taken from the wild, there is also a high probability that they are being illegally bred in captivity within Colombia, where there are currently no authorisations for captive breeding or ranching for commercial purposes. This is based on evidence that seized individuals are of a similar size with similar morphological characteristics which are unlikely for wild-sourced specimens.

Further information on captive breeding is available from three range States as follows:

Brazil

A local expert stated that there are no specialised centres for captive breeding (Cunha, in litt., 2022)

Colombia

There are no authorisations for captive breeding or ranching for commercial purposes.

Peru

There are two approved *in-situ* management areas and four approved zoos, which are the source of production of the species for commercial export. Data on the size of the breeding stock are not currently available. It is not possible to know or differentiate with certainty the volume of Matamatas that come from *ex-situ* management or

from *in-situ* management, and although the proponents state that exports from Peru originate from ranched individuals and production in authorised zoos, it is not clear what method of production is used in each management facility. The proponents state that at least three people or companies that currently manage farms for this species have been found to be in illegal possession of Matamatas in the seizure database managed by the National Forestry and Wildlife Service.

Implementation challenges (including similar species)

There are no similar species.

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Inclusion of Alligator Snapping Turtle *Macrochelys temminckii* and Common Snapping Turtle *Chelydra serpentina* in Appendix II

Proponent: United States of America

Summary: *Macrochelys temminckii* and *Chelydra serpentina*, commonly referred to as snapping turtles, are large aquatic freshwater turtles, native to North America, with *M. temminckii* endemic to the USA. They inhabit a wide range of waterbodies including large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows. Currently, both species are listed in Appendix III and both are proposed for inclusion in Appendix II with *M. temminckii* proposed for inclusion under Article II, paragraph 2(a) and *C. serpentina* proposed as a lookalike under Annex 2b of Res. Conf. 9.24 (Rev. CoP17).

Macrochelys temminckii was estimated to have a total population of 361,213 individuals in 2021, with population estimates ranging from 68,154–1,435,825 individuals. Regional *M. temminckii* population estimates vary across its range from an estimated 200,000 individuals in a southern region to 213 individuals in a northern region. The northern populations of *M. temminckii* are also reported to be experiencing a greater level of range contraction and local extirpations than southern populations. The most recent IUCN Red List assessment for *M. temminckii* was conducted in 1996 and classified the species as Vulnerable. *Chelydra serpentina* was classified as Least Concern in 2010, with an estimated total population of between 10,000 and one million individuals.

Extensive historic commercial harvest for human food of *M. temminckii* in the USA led to population decreases. Due to the species' slow life history, populations in areas where harvesting occurred have either remained stable or decreased. Direct international trade in *M. temminckii* in the most recent ten years (2011–2020), was above 350,000 (averaging ~38,000 annually) with virtually all reported as live, wild-sourced individuals exported by the USA. However, most individuals in trade are immatures hatched (and presumed bred) in captivity, which are reported as wild-sourced because the legal acquisition status of founder stock remains unknown. Virtually all this trade was imported by Hong Kong SAR, Macao SAR, and mainland China. *Macrochelys temminckii* is affected across its range by a variety of factors including nest predation, mortality and injury associated with freshwater fishing and other recreational activities and adult harvest. It is unknown to what extent the international trade drives harvest in *M. temminckii* and what impact it has on the wild population, and expert opinions on this differ. The commercial harvest of wild *M. temminckii* is prohibited in all states within its range and restricted personal harvest is only permitted in Louisiana and Mississippi. Some illegal harvest of *M. temminckii* was reported still to occur in the USA, but seizures involving the species were infrequent and the scale and impact on wild populations unknown.

Extensive trade has also been reported in *Chelydra serpentina* (~570,000 for the period 2011–2020); this is almost entirely (98%) in individuals reported as captive-bred or captive-born. While adult *C. serpentina* can be readily distinguished from *M. temminckii*, immature individuals are similar in appearance.

Analysis: *Macrochelys temminckii* and *Chelydra serpentina* are relatively widespread freshwater turtles occurring in North America. Neither species has a small population, with *M. temminckii* the least abundant of the two, estimated to have a global population of over 300,000. The two species are both in international trade, with an average annual export of around 38,000 live *M. temminckii* individuals from the USA reported as wild-sourced. However, most of these comprise immature individuals hatched (and presumed bred) in captivity. *Macrochelys temminckii* populations had decreased due to commercial harvesting in the past, however, commercial trade in specimens harvested from the wild is now banned in all states in the USA with only limited personal harvest permitted in two states. Based on the information available it does not appear that *M. temminckii* meets the criteria for inclusion in Appendix II under Annex 2a of Res. Conf. 9.24 (Rev CoP17). This being the case *C. serpentina* would not meet the lookalike criteria set out in Annex 2b of the

Resolution, despite the difficulty in distinguishing between hatchlings, these being the principal specimens in trade.

Other Considerations: In November 2021, the U.S. Fish and Wildlife Service (USFWS) proposed including *M. temminckii* as a threatened species on the List of Endangered and Threatened Wildlife. This would extend the Endangered Species Act of 1973 (ESA) to this species. If adopted it would be illegal to import or export, take, possess, transport across states or USA borders or conduct activities with this species without an authorisation permit.

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 Checklist of Chelonians of the World (Fritz and Havaš, 2007), the CITES Standard Reference, includes the two proposed species *Macrochelys temminckii* and *Chelydra serpentina*. Morphological and generic analyses of *M. temminckii* has indicated the genus comprised three species, *M. temminckii*, *M. apalachicola*, and *M. suwanniensis* (Thomas *et al.*, 2014). A recent unpublished study assessed *Macrochelys* using next-generation sequencing and supported the designation of the three species (Apodaca *et al.*, 2022). CITES currently follows Fritz and Havaš (2007) and as such *M. suwanniensis* and *M. apalachicola* are treated as synonyms of *M. temminckii*.

Range

Macrochelys temminckii: USA and South Korea (introduced).

Chelydra serpentina: Canada, USA, mainland China (introduced), Japan (introduced), and Taiwan POC (introduced).

IUCN Global Category

Macrochelys temminckii: Vulnerable A1cd (assessed 1996, ver. 2.3)

Chelydra serpentina: Least Concern (assessed 2010, 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

Macrochelys temminckii is the largest species of freshwater turtle in North America and is characterised by a large head, long tail, and a strongly hooked upper jaw. It occurs in 12 states: Alabama, Arkansas, Florida, Georgia, Illinois, Kentucky, Louisiana, Mississippi, Missouri, Oklahoma, Tennessee, and Texas. While its presence in its historical range states of Indiana and Kansas is unknown. The species is among the most aquatic freshwater turtles and inhabits a range of deep waterbodies including large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows. *Macrochelys temminckii* has a relatively long generation length with males reaching sexual maturity at 11–21 years and females at 13–21 years. It also has a relatively low reproductive output compared to *Chelydra serpentina*, with females laying only a single clutch per year with an average of 27.8 eggs per clutch.

The global population of *M. temminckii* was estimated to total 361,213 individuals according to a 2020 and 2021 national species assessment, with 55% of these individuals occurring in a southern region encompassing eastern Mississippi, western Alabama, and small parts of Louisiana and Florida (i.e., Alabama Unit). This population estimate was derived from abundance values ranging from 68,154–1,435,825 individuals, highlighting the high degree of uncertainty in population estimates. The abundance of this species across its range was estimated to vary from 200,000 individuals in the Alabama Unit to 213 individuals in a northern region encompassing Missouri, Illinois, Indiana, Kentucky, and Tennessee. The northern populations were also reported to experience greater range contractions and local extirpations than southern populations with range contractions reported in Illinois, Kansas, Kentucky, Missouri, Oklahoma, and Tennessee. *A long-term study on the M. temminckii population in Texas reported a stable population trend (Fitzgerald, in litt., 2022).* The last IUCN Red List assessment for *M. temminckii* was conducted in 1996, 26 years ago, and classified the species as Vulnerable. *However, no justification for the listing was provided.*

Chelydra serpentina has a widespread range including 42 states in eastern and central United States, where the species is both native and present, and extends north into southern Canada (including the six provinces of Manitoba, New Brunswick, Nova Scotia, Ontario, Quebec, and Saskatchewan). *C. serpentina* has also been introduced to areas in western USA, as well as to mainland China, Japan, and Taiwan POC. *Chelydra serpentina* is

highly aquatic and found in a variety of freshwater habitats such as rivers, lakes, reservoirs, ponds and marshes. *Chelydra serpentina* has a relatively slow life history that varies across its range. In southern areas males are reported to reach sexual maturity around four to six years and females around four to 20 years. However, northern populations of males reached sexual maturity around 15–20 years and females around 17–19 years. Female *C. serpentina* generally produce one clutch per year, with clutch size averaging 35 eggs (ranging from four to 109 eggs).

The total population size for *C. serpentina* is unknown. A global abundance estimate for the species was reported to range between 10,000 and >1,000,000 individuals. *Chelydra serpentina* was last assessed on the IUCN Red List in 2010 and classified as Least Concern. In Canada, *C. serpentina* is a species of special concern (reassessment is in progress). The size of Canada's population (10% of the global population), while unknown, is estimated to be in the thousands.

National use

Historical extensive commercial harvest of this species for meat consumption, peaking in the late 1960s and 1970s, led to marked population decreases in the species' population. The commercial and recreational harvest of *M. temminckii* is now banned in all states, except for Louisiana and Mississippi where recreational harvest is still permitted. In Louisiana, harvest is restricted to one *M. temminckii* individual a day per person, per vehicle/vessel with a fishing licence. No tagging requirements or reporting is required so the quantity of *M. temminckii* is unknown. In Mississippi harvest is restricted to one individual per year, prohibited from April 1 to June 30 and restricted to individuals with a straight-line carapace length of ≥ 24 inches (≥ 61 cm). Male *M. temminckii* mature at 11–21 years and females at 13–21 years. Females lay only one clutch per year with an average clutch size of 28 eggs (ranging from nine to 61 eggs). The slow reproductive rate of the species means it is highly susceptible to overharvesting, particularly of females, whereby an annual survivorship of <98% is likely to be unsustainable. While harvest restrictions have reduced the number of turtles being collected from the wild, demographic studies in areas where harvesting occurred indicate populations have either stayed constant or decreased. *Macrochelys temminckii* hatchlings have been posted online for USD100 per turtle, with a notification on the legality in trading in these species in the USA.

Chelydra serpentina is harvested in the USA for domestic meat consumption and to a lesser degree for the pet trade. The personal harvest of this species is permitted throughout almost all of its range in the USA with variable levels of regulation. The harvest of *C. serpentina* in Canada is now prohibited in all six provinces. There is no evidence of a wide-scale pet trade for this species in Canada, but some evidence demand is increasing in cosmopolitan areas.

Legal trade

International trade in *M. temminckii* has been subject to CITES regulation since 2006 when the species was included in Appendix III. According to the CITES Trade Database, all direct trade in *M. temminckii* in the most recent ten years, 2011–2020, for commercial and personal purposes was exported by the USA and virtually all reported as of live, wild-sourced individuals (>99%, 303,706). As the USA CITES annual reports for 2019 and 2020 were not yet submitted and included the CITES Trade Database at the time of writing, this number likely represents an underestimate. Direct exports of live *M. temminckii* from the USA for commercial and personal purposes remained relatively constant between 2011 and 2018, averaging an annual export of 37,963 individuals. The highest quantity of this trade was imported by Hong Kong SAR (46%; 138,937), Macao SAR (32%; 96,440) and mainland China (21%; 64,343), representing a combined total of 99% of all trade (Figure 1).

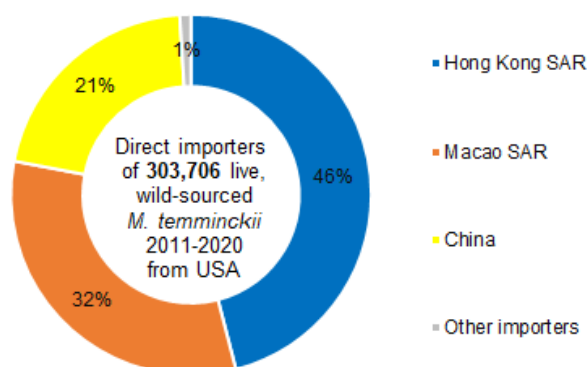


Figure 1. Direct imports of live *M. temminckii* individuals exported by the USA for commercial and personal purposes, 2011–2020, as reported by the USA (CITES Trade Database).

According to trade data from LEMIS, direct trade of *M. temminckii* exported by the USA in the most recent ten years, 2011–2020, for commercial and personal purposes almost exclusively comprised live individuals (>99%; 364,832). This live trade was virtually all reported as involving wild-sourced individuals (Figure 2.; 98%; 357,875; source W and U). The average annual trade in live, wild-sourced individuals totalled 35,688, and decreased by 28% during this period, while trade in captive-born individuals increased between 2018 and 2020. Corresponding with the CITES trade data, the highest quantity of imports was by Hong Kong SAR (54%; 192,905), Macao SAR (27%; 96,440) and mainland China (18%; 63,343), accounting for a combined total of 99% of all trade.

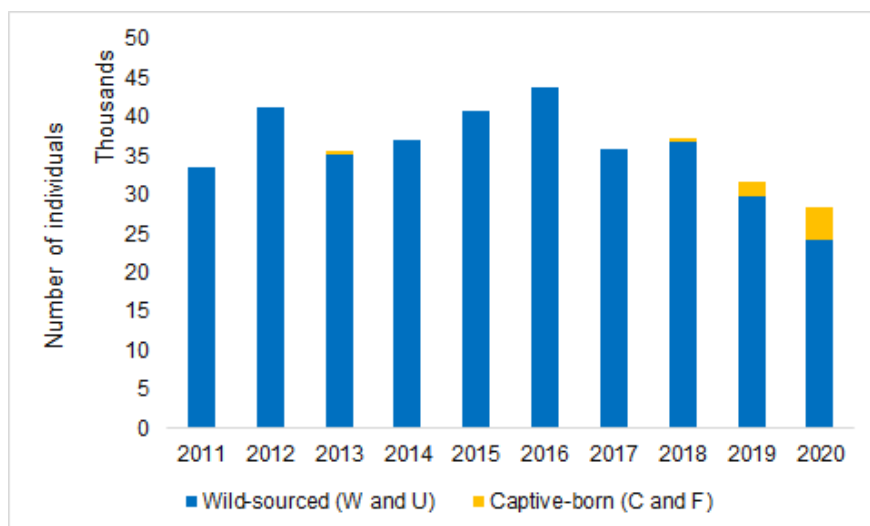


Figure 2. Direct exports of live *M. temminckii* individuals exported by the USA for commercial and personal purposes, 2011–2020 (LEMIS).

While virtually all live *M. temminckii* exports were reported as sourced from the wild, most individuals in trade comprise immature animals hatched in captive breeding facilities. As such, this trade is not documented in accordance with Resolution Conf. 12.10 (Rev. CoP15), and the legal acquisition of founder stock remains unknown.

Chelydra serpentina was included by the USA in CITES Appendix III in 2016. According to an analysis of CITES trade data presented here, direct trade in *C. serpentina* from 2011–2020, for commercial, personal and hunting purposes was all in live individuals (570,168) and almost entirely exported by USA (>99%), as reported by the USA. Almost all individuals were captive-bred or captive-born (98%; 558,308; source C and F) and imported by mainland China (95%). The remainder of the trade was in wild-sourced individuals (11,510) and imported almost exclusively by Hong Kong SAR (99%).

According to a USFWS (2021) report all *M. temminckii* individuals exported by the USA originated from 12 CITES permitted farms in Arkansas, Louisiana, Missouri, and Mississippi. Furthermore, these facilities do not explicitly label these individuals as captive-bred or captive-born as it is unknown whether these facilities meet the requirement of Resolution Conf. 10.16 (Rev.) as self-sustaining populations and bred in a “controlled environment”. The legal acquisition of founder stock also remains unknown.

Illegal trade

There is evidence that illegal harvest of *M. temminckii* still occurs in the USA, but its prevalence varies across its range. The highest estimates of illegal trade were reported in areas where legal harvest also occurs. *M. temminckii* was reported to be infrequently seized during investigations with confiscations typically involving the possession of individual turtles without a permit (O’Hanlon, in litt., 2022). In 2017, 60 adult *M. temminckii* were seized from three men convicted of collecting them in a single year and transporting them across state lines in violation of the Lacey Act (Eastern District of Texas Department of Justice, 2017). In 2017, 21 adult, and six juvenile *M. temminckii* were released into eastern Texas after being seized in an illegal trafficking attempt in 2016 (Texas Parks and Wildlife, 2021). No major seizures of *M. temminckii* have been reported in the state of Florida (O’Hanlon in litt., 2022).

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

Macrochelys temminckii and *C. serpentina* are the only North American species of the Chelydridae family. There are two additional species in the genus *Chelydra*, namely *C. acutirostris* and *C. rossignonii* both occurring in southern Central America and northwestern South America. Both taxa were once considered subspecies of *C. serpentina* and morphological differences among the *Chelydra* species are subtle. As such, they may be hard to differentiate and present an implementation challenge for enforcement officers, particularly when dealing with immature individuals. However, *C. acutirostris* and *C. rossignonii* are geographically distinct from the North American Chelydridae species and are unlikely to be encountered in trade from the USA.

Additional information

Threats

Macrochelys temminckii is impacted across its range by habitat loss and modification, nest predation, mortality and injury associated with freshwater fishing and other recreational activities such as incidental hooking, hook ingestion and bycatch from commercial fishing, as well as legal and illegal adult harvesting. The major effects on this species include nest predation, human-activity related mortality and injury and adult harvest. These were estimated to reduce the survival rate of more than half of the *M. temminckii* individuals in broad regional units across its range (USFWS, 2021). *In Florida, the primary threat to this species was accidental mortality from interactions with fishing gear, and while illegal trade does occur it was not considered a primary threat to the species* (O'Hanlon, in litt., 2022). *Domestic trade and poaching were reported to have a large impact on this species* (Gordon, in litt., 2022) with small-scale poaching events reported to be not uncommon (Weissgold, in litt., 2022). *However, it is unknown how the acquisition of adult breeding stock from wild populations, collection of eggs from the wild, or supplementation of wild individuals for exports, is impacting this species.* To a lesser degree this species is also impacted by disease, nest parasites, and climate change.

Conflicting expert opinion exists on whether international trade is impacting the wild population of M. temminckii. Weissgold (in litt., 2022) reported that the international trade in this species has been, and is still, a threat to its wild population. In contrast, Fitzgerald (in litt., 2022) reported that international trade in M. temminckii was not considered a threat to its wild population.

Chelydra serpentina is considered to be an adaptable species but ongoing large-scale legal harvest of the species for international trade was noted to be causing local population decreases. Populations of this species are most impacted by habitat loss, degradation and conversion, and road mortality, particularly in Canada. Other impacts across its range include nest predation, and mortality and injury associated with vehicle collisions and recreational or commercial fishing activities, illegal harvest and persecution. *Chelydra serpentina meat was reported to have notable value in commercial trade due to varying state-level protection* (Weissgold, in litt., 2022). *Illegal harvest of the species is likely to be continuing* (Weissgold, in litt., 2022). *Furthermore, Weissgold (in litt., 2022) reported that canned turtle meat soup is not identified distinctly under the customs regime, and that significant exports may occur that are not being declared to USFWS. It has been suggested that the high international demand for C. serpentina cannot be met by the captive turtle industry. Fitzgerald (in litt., 2022) reported that international trade in C. serpentina is not considered a threat to the species' wild population. International trade in C. serpentina had been a problem in Texas, however, the state banned commercial harvest in 2007, with the exception of C. serpentina individuals in private waters* (Fitzgerald, in litt., 2022).

Conservation, management and legislation

Commercial harvest of *M. temminckii* is prohibited in all states within its range, while personal harvest is permitted with restrictions in Louisiana and Mississippi (Table 1). The USFWS proposed to list *M. temminckii* as a threatened species on the List of Endangered and Threatened Wildlife in November 2021, extending the ESA (1973) to this species. However, the decision on the listing is still pending. *C. serpentina* is not protected under the ESA and has not been proposed to be listed as threatened or endangered. Harvesting of this species for commercial purposes from the wild is permitted in about half the states in which it occurs, and personal harvest is also permitted in nearly all states within its range in the USA.

Table 1. Protected status and harvest regulations for *Macrochelys temminckii* within the USA.

| US state | State status | Year commercial harvest prohibited | Year personal harvest prohibited | Notes |
|----------|--------------------|------------------------------------|----------------------------------|-------|
| Alabama | Species of concern | 2012 | 2012 | |
| Arkansas | None | 1994 | 1994 | |
| Florida | Threatened | 2009 | 2009 | |
| Georgia | Threatened | 1992 | 1992 | |
| Illinois | Endangered | 1994 | 1994 | |

| US state | State status | Year commercial harvest prohibited | Year personal harvest prohibited | Notes |
|-------------|---|------------------------------------|----------------------------------|--|
| Indiana | Endangered (Persistence in state unknown) | 1994 | 1994 | |
| Kansas | Species of greatest conservation need (Persistence in state unknown) | Unsure | Unsure | |
| Kentucky | Threatened | 1975 | 2012 | |
| Louisiana | Species of greatest conservation need | 2004 | Still allowed | Licence required for personal harvest; harvest limits: one turtle per day, per person, per vehicle/vessel; no restrictions on size of turtle |
| Mississippi | Species of greatest conservation need | 1991 | Still allowed | Licence required for personal harvest; harvest limits: one turtle per year; carapace length of turtle must be 24 inches or greater |
| Missouri | Species of conservation concern | 1980 | 1980 | |
| Oklahoma | Species of greatest conservation need | Never allowed | 1992 | |
| Tennessee | In Need of Management; considered rare to very rare and imperilled | 1991 | 1991 | |
| Texas | Threatened | 1993 | 1993 | |

Captive breeding

Conservation measures that have been implemented for *M. temminckii* in the USA include captive rearing, head-starting, reintroductions and habitat restoration and improvement projects. A captive breeding programme was established in Oklahoma in 1999, and head-started releases and reintroductions have been carried out across the states of Illinois, Kansas, Oklahoma, and Tennessee.

Implementation challenges (including similar species)

The adults of these two species exhibit distinct morphological features. *Most game wardens and USFWS enforcement officers claim to be able to distinguish between the two species (Gordon, in litt., 2022). Furthermore, a range of online identification resources for these two species is also available (Fitzgerald, in litt., 2022).* An implementation challenge is differentiating immature individuals of the two species as immatures of both species are characterised by a rough dark-coloured carapace with three distinct keels.

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Inclusion of Map turtles *Graptemys barbouri*, *G. ernsti*, *G. gibbonsi*, *G. pearlensis*, and *G. pulchra* in Appendix II

Proponent: United States of America

Summary: Broad-headed map turtles *Graptemys* spp. are a clade of five species within a genus of 14 species of medium-sized freshwater turtles with characteristic map-like markings on the carapace. The five species proposed for inclusion in Appendix II, *G. barbouri*, *G. ernsti*, *G. gibbonsi*, *G. pearlensis*, and *G. pulchra*, commonly referred to as broad-headed map turtles, are all endemic to southeastern USA. They were included in Appendix III in 2006.

Graptemys barbouri occurs in several river systems across the states of Florida, Georgia, and Alabama. In 2014–2015 its population in Florida was estimated at between 11,000 and 28,000 individuals and considered secure, with evidence the species had expanded its range in the state. *G. barbouri* was classified as Vulnerable on the IUCN Red List in 2010 due to an inferred population decline caused by habitat degradation, overharvesting, predation and disease. Very little international trade is reported in CITES trade data (56 live captive-bred and captive-born exported from the European Union (EU27) to mainland China and Hong Kong SAR, 2011–2020). Commercial harvest is banned throughout its range.

Graptemys ernsti has a restricted range in western Florida and southern Alabama. No population estimates are available. The species was classified as Near Threatened on the IUCN Red List in 2010 as it was perceived to have been experiencing population declines caused by habitat degradation. No international trade was reported between 2011 and 2020. *Graptemys ernsti* is protected from commercial exploitation in Alabama; in Florida collection is prohibited and possession limited to two individuals.

Graptemys gibbonsi has a restricted range in the Pascagoula River system in the state of Mississippi. Its population was recently estimated at just over 34,000 individuals. *Graptemys gibbonsi* was classified as Endangered on the IUCN Red List in 2010 due to a restricted range and an inferred population decline attributed to threats including water pollution, habitat loss and degradation, collection for the pet trade, direct human persecution and predation. Very little international trade is reported in CITES trade data (101 captive-bred individuals exported by the EU to China and Hong Kong SAR, between 2011 and 2020). The species is protected from commercial exploitation in Mississippi, with licensed collection limited to four per year.

Graptemys pearlensis has a restricted range along a combined 940 km section of the Pearl and Bogue Chitto rivers in the states of Louisiana and Mississippi. Its global population was estimated at 22,000 in 2020. *Graptemys pearlensis* was assessed as Endangered on the IUCN Red List in 2010 due to an estimated population decline of 80–90% between 1950 and 2010 caused by habitat degradation and loss, pollution, overharvesting, persecution and predation. Minimal international trade in *G. pearlensis* has been reported. Harvest of *G. pearlensis* is regulated in the state of Mississippi, with licensed collection limited to four individuals per year.

Graptemys pulchra occurs in river systems in the states of Alabama, northeastern Mississippi and northwestern Georgia. No population estimates are available and, while not particularly abundant in its range, no marked declines have been observed. *Graptemys pulchra* was classified as Near Threatened on the IUCN Red List in 2010 as the species was experiencing unquantified levels of threat from habitat degradation and predation. No international trade in *G. pulchra* has been reported between 2011 and 2020. *Graptemys pulchra* is prohibited from commercial harvest in Alabama and Georgia, with harvesting regulated in the states of Tennessee and Mississippi where licensed collection is limited to four individuals per year during certain periods.

There was no available evidence of illegal trade or seizures for these species. These five *Graptemys* species all resemble one another, particularly as juveniles, and may be difficult to distinguish when in trade.

Analysis: These five species of *Graptemys* turtles are all endemic to the USA. Three of the species (*G. barbouri*, *G. gibbonsi*, and *G. pearlensis*) have populations numbering in the low tens of thousands. Abundance observations for *G. ernsti* and *G. pulchra* indicate their populations are not of conservation concern. Unlike some other *Graptemys* species not covered by this proposal, minimal international trade in wild-sourced individuals of any of these five species has been reported between 2011 and 2020, and only small numbers of captive-born individuals have been reported in trade. The proposed *Graptemys* species occurring in the US states of Alabama, Florida, and Georgia are protected and thus prohibited from harvesting. Those occurring in the states of Mississippi and Tennessee are subject to harvest regulations but harvesting is not regulated in Louisiana. Because of existing national trade and harvesting regulations in most states and a lack of international trade in wild-sourced individuals, it is unlikely that any of the species meet the criteria for inclusion in Appendix II in Annex 2a of Res. Conf. 9.24 (Rev. CoP17).

Other Considerations: In 2021, the U.S. Fish and Wildlife Service announced that it intends to list *Graptemys pearlensis* as Threatened under the Endangered Species Act, and the four additional broad-headed map turtle species (*G. barbouri*, *G. ernsti*, *G. gibbonsi*, *G. pulchra*) as Threatened due to the similarity of their appearance. If adopted, this would prohibit these species from being harvested, offered for sale or sold and imported or exported via interstate or 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 Checklist of Chelonians of the World (Fritz and Havaš 2007), the CITES Standard Reference, includes the proposed species *G. barbouri*, *G. ernsti*, *G. gibbonsi* and *G. pulchra*, with the exception of *G. pearlensis*, described in 2010 (Ennen et al., 2010).

Range

United States of America

IUCN Global Category

Graptemys barbouri: Vulnerable A2bcde (2010) ver. 3.1. Year of description 1942.

Graptemys ernsti: Near Threatened (2010) ver. 3.1. Year of description 1992.

Graptemys gibbonsi: Endangered A2bce+4ce (2010) ver. 3.1. Year of description 1992.

Graptemys pearlensis: Endangered A1bcde+4bcde (2010) ver. 3.1. Year of description 2010.

Graptemys pulchra: Near Threatened (2010) ver. 3.1. Year of description 1893.

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

Graptemys barbouri, *G. ernsti*, *G. gibbonsi*, *G. pearlensis*, and *G. pulchra* are all endemic to the southeastern USA.

Graptemys barbouri occurs in several river systems across the states of Florida, Georgia, and Alabama.

Graptemys ernsti has a restricted range along sections of the Escambia, Yellow, and Shoal rivers and tributaries in western Florida and southern Alabama.

Graptemys gibbonsi has a restricted range in the Pascagoula River system in the state of Mississippi.

Graptemys pearlensis has a restricted range along a combined 940 km section of the Pearl and Bogue Chitto rivers in Louisiana and Mississippi states.

Graptemys pulchra occurs in river systems in the states of Alabama, northeastern Mississippi and northwestern Georgia.

Population estimates, abundance, and trend data vary across the five *Graptemys* species.

Graptemys barbouri

The total population of this species was estimated to be between 1,000 and 10,000 individuals in 2010. This species was reported to consist of between 1 and 20 subpopulations and purported to vary in abundance across its range. More recent population abundance estimates from river systems in Florida reported a total abundance across the state of between 11,601 and 28,176 individuals. *Basking density estimates across this species' range, provided in Lindeman et al., (2020), varied between 0.1 and 21.8 individuals per river km (ind./km).* This species was reported to be abundant in parts of its range, and while a declining population trend was reported in 2010, its population in Florida in 2014–2015 was noted as secure with evidence of range expansion.

Graptemys ernsti

No recent population estimates or abundance data was available for this species. Past estimates in the Escambia River in Florida record 5 ind./km and 5.9 ind./km in the Conecuh River in Alabama. *Basking density estimates across this species' range varied from 2.6–24.9 ind./km (Lindeman et al., 2020).*

Graptemys gibbonsi

The global population was estimated to total 34,081 individuals. A density estimate of 34–44 ind./km was reported in 2009 and *basking density estimates across this species' range varied from 1.8–8.9 ind./km.*

Graptemys pearlensis

The global population of *G. pearlensis* was estimated to total 21,841 individuals in 2020. *Studies in the Pearl River Drainage estimated a total population of around 110 in 1999 (Devros in litt., 2022).* This species' population trend was reported to be declining in 2010, supported by recent studies reporting declines and rapid declines across parts of its range. *This was supported by long-term trend observations of the G. pearlensis population in the Pearl River Drainage, where 105 individuals (listed previously as G. pulchra) were observed in 1953, 36 in 1971, 38 in 2017 and 72 in 2022 (Devros in litt., 2022).* The relative abundance of the Pearl River Drainage population was estimated to be 2.4 ind./km in 1998, 16.1 ind./km in 2017, and 9.99 ind./km in 2022 (Devros in litt., 2022). In Louisiana, a study in 2020–2021 observed *G. pearlensis* basking densities of 3.1 ind./km with mark-resight estimates averaging 8 ind./km (ranging from 1–23 ind./km; Selman, unpublished data, in litt., 2022). *Graptemys pearlensis* densities in the main stem of the Pearl River were reported to be low to very low compared with *G. oculifera* (Selman, unpublished data, in litt., 2022).

Graptemys pulchra

No recent population estimates were available for this species. *Basking density estimates across this species' range, provided in Lindeman et al. (2020), varied between 0.1 and 4.8 ind./km. Preliminary data on G. pulchra in Mississippi report basking densities of 1.3 ind./km and mark-resight population estimates of 16 ind./km (Selman, unpublished data, in litt., 2022).*

According to IUCN Red List assessments four of the five species (*G. barbouri*, *G. ernsti*, *G. gibbonsi*, and *G. pearlensis*) were assessed as having decreasing populations, while the population trend for *G. pulchra* was unknown.

National use

In the USA, the scale of Graptemys turtle harvesting from the wild is unknown (Selman in litt., 2022). Informal interviews with people utilising the rivers, such as fishermen, report Graptemys turtles are still being collected as there is insufficient law enforcement to prevent their harvest and sale (Selman, in litt., 2022). In the USA, legal collection of these five *Graptemys* species is permitted in the states of Mississippi and Louisiana and the possession of these species by private persons or entities is allowed in compliance with applicable Federal permits and regulations. Online sales in broad-headed map turtles indicated they have been nationally traded, including one purported intention to breed *G. barbouri* for commercial purposes. Online advertisements of broad-headed map turtles mostly state the individuals to be captive bred, however, the source of these individuals can be hard to verify. *Graptemys pulchra* was reported not to be frequently observed or highly desired in the pet trade.

International trade

According to CITES trade data, direct exports of Graptemys barbouri, G. ernsti, G. gibbonsi, G. pearlensis, and G. pulchra over the most recent ten-years of trade, 2011–2020, for commercial and personal purposes totalled 160 live, captive-bred and captive-born (source C and F) individuals, as reported by exporters. This trade was exported by Italy (119) and Germany (31) to mainland China (150), and Switzerland (10) to Hong Kong SAR (10). In 2018, the USA also exported 18 live individuals of G. pulchra, G. barbouri, and G. pearlensis to Austria for captive-breeding purposes. Six of these individuals (all G. pearlensis) were reported as wild-sourced.

In context with all Graptemys turtle trade, direct exports of other Graptemys species (not covered by this proposal) for commercial and personal purposes between 2011 and 2020 almost entirely comprised live individuals (1,330,728 mostly G. pseudogeographica (false map turtle) which is not included in this proposal), as reported by exporters. Only one individual was exported at the genus level (i.e., Graptemys spp.). The majority of this trade was exported by the USA (828,613) comprised almost entirely of wild-sourced individuals (96%, 792,656) imported by

the EU27 countries and the UK (72%, 595,719); and mainland China (501,509) all of which were captive-bred and virtually all imported by EU27 countries (over 99%, 500,509).

According to LEMIS, no direct commercial trade in *G. barbouri*, *G. ernsti*, *G. gibbonsi*, *G. pearlensis*, and *G. pulchra* was reported to be exported by the USA, 2011–2020. Non-commercial trade for captive breeding purposes during this period included two live *G. barbouri* individuals, six live *G. pearlensis* individuals and ten live *G. pulchra* individuals exported from the USA to Austria in 2018.

Illegal trade

No recent seizures of *Graptemys* in the USA have been recorded (Lindeman in litt., 2022). The extent of enforcement actions relating to these species in the USA is unknown. There are concerns that these five species may be incorrectly declared to law enforcement authorities and thus bypassing CITES regulations due to the difficulty in identifying species within the broad-headed map turtle group. However, no evidence for this was identified.

A short online survey on the European online platform Terraristik found one seller located in Germany offering *G. gibbonsi* and *G. barbouri* as well as one advert from an individual in Italy seeking to purchase *G. gibbonsi*. The source and price of the turtles were not indicated. A total of 22 adverts were discovered for all *Graptemys* species.

Additional information

Threats

The five broad-headed map turtle species are threatened by habitat loss and degradation (including declining water quality from agricultural run-off, development, mining, hydrological modifications and impoundments and removal or loss of deadwood), drought, flooding, sea-level-rise and turtle harvesting. *Conflicting expert opinion exists on whether international trade is impacting wild populations of these species. Selman (in litt., 2022) reported that the international trade in these species was a threat and that they were still being harvested, but the level of off-take was unknown. Weissgold (in litt., 2022) reported that poaching levels in these species had varied over the decades primarily driven by demand in East Asia, and international trade presents a threat to these species, especially isolated subpopulations. In contrast, Fitzgerald (in litt., 2022) considered international trade was not a threat to these species due to the low levels of non-wild sourced individuals in trade.*

Species-specific threats:

Graptemys barbouri

Habitat degradation, overharvesting and predation, as well as hybridisation.

Graptemys ernsti

Water pollution, habitat degradation and possibly the international pet trade.

Graptemys gibbonsi

Water pollution, habitat degradation, pet trade, persecution and predation.

Graptemys pearlensis

Water pollution, habitat degradation, pet trade, persecution and predation. *Anecdotal observations from the Pearl River Drainage found no evidence of commercial harvesting of G. pearlensis, with pollution reported to be the most significant threat process in this location (Devros in litt., 2022). Direct persecution of turtles by humans in the form of turtles being shot for recreation, a noted historical threat, was also not observed in recent studies in the Pearl River Drainage (Devros in litt., 2022).*

Graptemys pulchra

Predation and pollution may be threatening this species.

Conservation, management and legislation

All species of the genus *Graptemys* have been included in Appendix III since 2006. A summary of national regulations and management according to US state is included in Table 1.

Table 1. Protection status, legislation, management, and conservation for *Graptemys* species according to US state

| US state | Species occurring in state and level of protection (protected, regulated or not regulated) | Legislation, management and conservation |
|----------|--|---|
| Alabama | <i>G. barbouri</i> (protected) <i>G. ernsti</i> (protected) | According to Alabama state law it is illegal to take, capture, kill; attempt to take, capture or kill; possess or offer for sale or |

| US state | Species occurring in state and level of protection (protected, regulated or not regulated) | Legislation, management and conservation |
|-------------|---|---|
| | <i>G. pulchra</i> (protected) | trade any specimens, parts or reproductive products of any <i>Graptemys</i> species. Activities involving these species is only permitted with a scientific collection or written permit from the State's Department of Conservation and Natural Resources. <i>G. barbouri</i> and <i>G. ernsti</i> were identified as species of greatest conservation need in the Alabama State Wildlife Action Plan (Alabama Department of Conservation and Natural Resources, 2015). |
| Florida | <i>G. barbouri</i> (protected) <i>G. ernsti</i> (protected) | The state of Florida lists <i>G. barbouri</i> as threatened under the state's Endangered and Threatened Species Rule. According to this Rule threatened species are prohibited from being possessed or taken from the wild. Additionally, species resembling threatened species including <i>G. ernsti</i> are also prohibited from being taken from the wild. <i>G. barbouri</i> and <i>G. ernsti</i> were identified as species of greatest conservation need in the Florida State Wildlife Action Plan (Florida Fish and Wildlife Conservation Commission, 2019). |
| Georgia | <i>G. barbouri</i> (protected) <i>G. pulchra</i> (protected) | The state of Georgia lists <i>G. barbouri</i> as threatened and <i>G. pulchra</i> as rare in its Protected Species list, and the species are thus subject to Federal and State laws. <i>G. barbouri</i> and <i>G. pulchra</i> as species of greatest conservation need in the Georgia State Wildlife Action Plan (Georgia Department of Natural Resources, 2015). |
| Louisiana | <i>G. pearlensis</i> (not regulated) <i>G. pulchra</i> (not regulated) | The state of Louisiana does not have any specific regulations regarding <i>Graptemys</i> species and species not listed as prohibited or with specific regulations (including <i>G. pearlensis</i>) can be harvested with no limits. <i>G. pearlensis</i> is listed as a species of greatest conservation need in the Louisiana State Wildlife Plan (Holcomb et al., 2015). |
| Mississippi | <i>G. gibbonsi</i> (regulated) <i>G. pearlensis</i> (regulated) <i>G. pulchra</i> (regulated) | The state of Mississippi regulates the harvesting of all <i>Graptemys</i> species through licences, stipulating that a person may possess and harvest from the wild no more than 10 non-game turtles per licence year. No more than four can be taken of the same species or subspecies and none may be collected between 1st April and 30th June. <i>G. gibbonsi</i> , <i>G. pearlensis</i> , and <i>G. pulchra</i> are listed as species of greatest conservation need in the Mississippi State Wildlife Action Plan (Mississippi Museum of Natural Science, 2015). |
| Tennessee | <i>G. pulchra</i> (regulated) | The state of Tennessee regulates the harvesting of <i>Graptemys</i> species through licences. |

All *Graptemys* spp. were listed in Annex C of the EU Wildlife Trade Regulations in 2008.

In China, the domestic management of wild-sourced *Graptemys* spp. is reported to be equal to Class II protected wild animals in the list of National Key Protected Wild animals since 2018 (Ling, in litt., 2022). This law stipulates captive breeding of wildlife under special state protection "shall benefit the protection of the species and scientific research and may not damage wild populations" (Ling, in litt., 2022).

Captive breeding

The extent of captive breeding in the five species of broad-headed map turtles for commercial purposes is unknown. The majority of *Graptemys* species were reported to do well in captivity. Captive-bred specimens of *G. gibbonsi* were reported in Europe and the USA. Large-scale commercial breeding in slow-reproducing turtle species was reportedly unprofitable. One population of *G. gibbonsi* is reportedly being maintained in a USA zoological institute as an assurance colony.

Implementation challenges (including similar species)

All species of the genus *Graptemys* are difficult to differentiate due to their similarity of appearance, particularly among hatchlings, and identification poses a particular enforcement challenge to the trade and regulation of these species.

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Transfer of Red-crowned Roofed Turtle *Batagur kachuga* from Appendix II to Appendix I

Proponent: India

Summary: *Batagur kachuga*, one of six species in the genus *Batagur*, is a large freshwater turtle, extant in India and thought to be extinct in Bangladesh. The most recent IUCN Red List assessment for the species, conducted in 2019, noted that the only reliable records of the species in the previous 12–13 years were in the National Chambal Sanctuary in northern India, where there were estimated to be around 500 mature individuals. The species was assessed as Critically Endangered on the basis of suspected historical decline and continuing adverse impacts of pollution, hydrological projects and local subsistence consumption. It was included in Appendix II in 2002. Currently *Batagur affinis* and *B. baska* are included in Appendix I and the remaining members of the genus in Appendix II.

The species is said to be in demand internationally for consumption as food and as pets, with the males more popular as pets due to their bright breeding colouration. Since 2016, the species has been reported in seizures within and outside India and in offers for sale on social media platforms. The CITES trade database reports only 14 live captive-bred specimens in commercial trade since the Appendix II listing in 2002 and no commercial exports since 2006. Although the species can be bred in captivity there are no known commercial captive breeding centres.

Analysis: The Critically Endangered *Batagur kachuga* has a small wild population with a highly restricted distribution in India. It is believed to have undergone a marked historical decline that is thought likely to be continuing. Despite legal protection there is international demand for the species and ongoing illegal trade is suspected. The species therefore appears to meet the criteria for inclusion in Appendix I in Res. Conf. 9.24 (Rev. CoP17).

All commercial exports of wild specimens of the species have been illegal in India since 1999. The 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.

CITES background

The species has been listed in CITES Appendix II since 2002, with the initial listing under the name Kachuga kachuga (UNEP, 2022). The 2002 listing was based on a Critically Endangered IUCN Red Listing assessment for the species in 2000, a declining population trend of more than 80% within 20 years and the threat of national and international trade for consumption of eggs, meat, and shells (Praschag et al., 2019).

Taxonomy

This species has several scientific synonyms, including Batagur bakeri, Batagur ellioti, Emys kachuga, Emys lineata, Emys lineata, Kachuga fusca, and Kachuga kachuga.

Range

India (extant), Bangladesh (thought to be extinct)

IUCN Global Category

Critically Endangered A2cd+4cd (assessed 2019, ver 3.1)

Biological criteria for inclusion in Appendix I

A) Small wild population

There are estimates of 500 mature individuals in the wild (Praschag et al., 2019). *This estimate from the IUCN Red List assessment published in 2019 is said to be consistent with recent extrapolations of nest surveys in a 50 km² area of the National Chambal Sanctuary between 2019–2022, the only area where the species is known to occur (Singh, in litt., 2022).*

B) Restricted area of distribution

The range of this species has declined and it is thought now to be extinct in Bangladesh, *with no records in the last ten years (Praschag, in litt., 2022)*. Available habitat for the species is decreasing due to water pollution and hydrological projects. *The only reliable records of the species in the 12–13 years prior to the 2019 Red List assessment were reported to be in the National Chambal Sanctuary, where 50 nests were observed over 100 km (Praschag et al., 2019).*

C) Decline in number of wild individuals

According to the IUCN Red List assessment conducted in 2019, there was an inferred population reduction of at least 80% in the past 50 years.

Trade criteria for inclusion in Appendix I**The species is or may be affected by trade**

Batagur kachuga is said to be illegally collected for the international pet trade and for consumption both locally and internationally. The males have a bright coloration and are hence preferred by pet traders. In the recent past, the major importers of *Kachuga* species were in South-East and East Asia, including China, Thailand, South Korea, Hong Kong SAR, Singapore, Japan, and Malaysia, and smaller numbers of specimens were sent to pet markets in Germany, Italy, the UK, and the USA.

Batagur kachuga is included in CITES Appendix II, allowing international commercial trade in the species provided such trade is not detrimental to the species. The CITES Trade Database shows some trade in the species since 2000, including the import of live captive-bred turtles for commercial trade purposes in 2005 and 2006 (six and eight turtles, respectively) into Japan from Lebanon, which reportedly originated in Kazakhstan. More recently, in 2012 one live turtle (from an unknown source) was reportedly exported from Singapore to Austria for a zoo, and in 2018, two live turtles were reportedly exported from Hong Kong SAR to the United States (sourced from confiscated or seized specimens) for educational purposes.

Although the species has been protected from collection for commercial purposes in India since 1986, there is evidence that the species may be illegally traded.

WiTIS has records of eight live individuals of the species reported in five seizures between 2010–2022, all of which occurred in 2017. There was evidence of international trade in four seizures. In one seizure, an unknown quantity of individuals was being smuggled from India to Malaysia and in another, two individuals were being smuggled from India to Nepal. Another seizure involved three individuals originating in India with the intended destination not clear. The remaining two seizures involved individuals of the species being smuggled from Bangladesh to Hong Kong SAR.

In 2017, 23 male *B. kachuga* were confiscated in Uttar Pradesh; at least five animals were confiscated in Hong Kong SAR and several were recorded in the Chinese pet trade. *Four individuals were also reported in seizures by the Directorate of Revenue Intelligence in India in 2020 (DRI, 2020). Some of these seizures may be duplicates of the five seizures reported in WiTIS.*

*Between 2016 and 2018, the Wildlife Justice Commission identified 378 individuals of *B. kachuga* either sighted by investigators or offered for sale in physical and online markets (Stoner, 2018a). There were 28 individuals reported in seizures in the same time frame (Stoner, 2018a). It was thought that most individuals offered for sale originated in Uttar Pradesh before being transported to Bangladesh for onward distribution. The relatively small number of individuals seized in comparison to the volume offered for sale could be indicative of law enforcement efforts not being sufficient to detect species being smuggled (Stoner, 2018b). One seller of the species in Malaysia, sourcing individuals from India, commented on the rarity of the species being a factor in them being in high demand (Stoner, 2018a). The median price for one individual was USD1,700, in comparison to USD110 for the more commonly traded Black Spotted Turtle *Geoclemys hamiltonii* (Stoner, 2018b).*

Due to the ongoing decline in the species' population, which is expected to continue into the future, any trade in the species will likely have a detrimental impact on its status.

Additional information**Threats**

The species is subject to overharvest for illegal consumption locally. Other threats include habitat loss from both hydrological projects impacting upon nesting beaches and water pollution, and entanglement in fishing nets.

International trade is not considered a major threat to the species, with numbers harvested for this purpose extremely low and the threat from local overharvesting of adults and eggs a greater issue, according to one expert (Praschag, in litt., 2022).

Conservation, management and legislation

The species was listed on Schedule I of the Wild Life (Protection) Act in India in 1986. Hunting and collection of the species is prohibited (Section 9 of the Act), and all commercial trade of the species and its derivatives is prohibited (Sections 40 and Chapter VI-A of the Act). First offences with respect to the species are punishable with imprisonment between 3–7 years and a fine of at least USD125. Since at least 1999, India has additionally banned the export for commercial purposes of wild-taken specimens of all CITES-listed species, including *B. kachuga* (see CITES No.1999/39).

In India, part of the distribution of *B. kachuga* lies within protected areas. *B. kachuga* is part of the river turtle head-starting programme in northern India and monitoring of the population in the National Chambal Sanctuary is being undertaken.

Captive breeding

Although one study has shown the species can be captive-bred (Whitaker, 2009), no commercial captive breeding centres are known either within or outside India (Praschag, in litt., 2022).

Implementation challenges (including similar species)

*Within this genus, there are currently two species in Appendix I, and three, including this species, in Appendix II. The species is distinguishable from other species in the genus. Juveniles and females may be easily confused with *Batagur baska* and *Batagur affinis*, both species listed in CITES Appendix I (Praschag, in litt., 2022).*

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Transfer of the Indochinese Box Turtle *Cuora galbinifrons* from Appendix II to Appendix I

Proponents: European Union, Viet Nam

Summary: The Indochinese Box Turtle *Cuora galbinifrons* is a medium-sized terrestrial turtle occurring in forested areas between 300 and 1700 m altitude in southern China, Lao People's Democratic Republic (PDR), and Viet Nam. It is slow to mature (10–15 years) and has low fecundity, with a single clutch of 1–3 eggs produced annually.

Apparently once common, field encounters with *C. galbinifrons* are now rare even during dedicated surveys, indicating that populations have suffered severe declines, with calculated population densities of less than one per square km (km²) within protected areas and suitable habitat. The species is estimated to have undergone declines of over 90% over the past 60 years (three generations, at 20 years per generation) and was classified as Critically Endangered by IUCN Red List in 2018, with collection for food and the international pet trade identified as primary threats. The species reportedly continues to be in high demand in the international pet trade and the local and regional food markets.

Cuora galbinifrons was included in Appendix II at CoP11 in 2000 under a genus-level listing for all *Cuora* spp. A proposal to transfer the species to Appendix I was submitted at CoP16 (Prop 33), but an alternative proposal (CoP16 Prop. 32) was adopted resulting in a zero quota for wild specimens traded for commercial purposes (effective June 2013). At the time of the original listing, *C. galbinifrons* included three distinct subspecies. At CoP17 a standard reference adopted for this taxon recognised the subspecies as full valid species (*C. galbinifrons*, *C. bourreti*, and *C. picturata*). *Cuora bourreti* (CoP18 Prop. 33) and *C. picturata* (CoP18 Prop. 34) were transferred to Appendix I at CoP18.

CITES trade reports have been limited to a few thousand individuals since *C. galbinifrons* was originally listed, with some reported trade in earlier years likely to have been in *C. bourreti* and *C. picturata*. Most of the trade was reported to be from captive sources. Observations from markets indicate that actual volumes in trade may be several orders of magnitude greater than reported legal trade volumes, suggesting a high level of illegal, unreported and unregulated trade. Since the zero quota for wild commercial trade was adopted, no trade in wild specimens has been reported.

Analysis: *Cuora galbinifrons* is a relatively widespread species that has been harvested extensively and is now rarely found in the wild. The species has been assessed as Critically Endangered and appears to meet the criteria for inclusion in Appendix I. The population is likely to be small and individuals are slow growing, with limited annual reproductive output, making the species intrinsically vulnerable to exploitation and having experienced a marked decline of 90% in three generations as a result of collection for trade.

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. galbinifrons* 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 galbinifrons was traditionally considered to include two non-typical subspecies, i.e., *Cuora galbinifrons bourreti* and *Cuora galbinifrons picturata* (e.g., Fritz and Havas, 2007). However, recent research has treated both

subspecies as full species *C. bourreti* and *C. picturata*, a status that was recognised for CITES purposes by the adoption at CoP17 of Spinks *et al.* (2012) as the nomenclature standard reference for the *Cuora galbinifrons* group.

Range

China, Lao PDR, Viet Nam

IUCN Global Category

Critically Endangered A2bd+4db (assessed 2018. ver.3)

Biological criteria for inclusion in Appendix I

A) Small wild population

No population data are available for this species. Recent anecdotal evidence suggests this species is uncommon and rarely encountered. Field surveys in Lao PDR 1993–1999 had an encounter rate of one individual per day when working with a trained turtle hunting dog in prime turtle habitat.

Wang *et al.* (2011) calculated a population density of 0.786 individuals per km² in a small area at Diaoluoshan Nature Reserve in Hainan, based on surveying six sampling areas with 424 baited traps over 6,360 trap-days (Li *et al.*, 2020).

In northern Viet Nam, a 2019 survey of the bamboo and hilly secondary forests of the Tam Thanh Commune and Quan Son District encountered three individuals during time-constrained walks on random transects of 1–4.5 km in length, covering a total of 65 km over a total of 362 survey hours. The overall density of *C. galbinifrons* was calculated as 0.057 individuals per km.

Cuora galbinifrons does not reach sexual maturity until 10–15 years old, and breeding records from captive animals suggest they produce one clutch per year comprising one to three eggs. This slow growth rate and low fecundity makes this species vulnerable to high rates of decline.

B) Restricted area of distribution

Cuora galbinifrons occurs in Hainan and Guangxi in China, in north-eastern Lao PDR, and in northern Viet Nam and at least as far south as Nghe An Province. *Cuora bourreti* occupies a smaller range from central Viet Nam (Nghe An, Ha Tinh, Quang Binh, Thua Thien-Hue, Da Nang, Quang Nam, and Kon Tum Provinces), as well as from adjoining Savannakhet Province in Lao PDR. *Cuora picturata* is restricted to the eastern slopes of the Langbian Plateau in an area of less than 25,000 km².

Forest cover in Viet Nam decreased from 14.3 million ha (43% land area) in 1943 to 9.5 million ha (29%) in 1973, and since then forest cover has appeared relatively stable from assessments in 1979–1981 and 1995 (FAO, 1997; FSIV, 2009). Recent increases in forest cover since early 2000 have largely been due to the 1998–2010 goal to reforest five million ha. However, reforestation has mostly been monoculture, while the natural forests that this species depends upon continue to be lost or degraded. Quantitative data have not been available on primary forest areas or trends in the areas of occurrence of this species in China and Lao PDR.

C) Decline in number of wild individuals

Cuora galbinifrons has suffered dramatic population declines due to harvesting for the international pet trade and the Asian food consumption over the last 15–20 years, and this has likely resulted in the depletion or extirpation of each population that has been surveyed. This species was assessed as Critically Endangered on the IUCN Red List of Threatened Species in 2000 when it included the *C. bourreti* and *C. picturata* subspecies. An updated Red List assessment for *Cuora galbinifrons* was published in 2020, with an estimated population collapse of over 90% in the past 60 years (three generations, at 20 years per generation), predicted to continue.

Much of the information on the population trends of this species is anecdotal and comes from Viet Nam. It was reported that C. galbinifrons is collected intensively throughout its range, and hunters report that this once common species is now increasingly difficult to find. For instance, hunters claimed that they used to be able to collect 20 individuals a day in the 1990s, but by 2006 they could only find a few animals a week. In 2011, the Conservation of Asian Tortoises and Freshwater Turtles Workshop noted that habitat destruction and intensive hunting have caused rapid declines in C. galbinifrons (Horne et al., 2012).

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

The C. galbinifrons group represents the second most valuable type of turtle in trade in Viet Nam and Lao PDR after the C. trifasciata complex. Cuora galbinifrons is in decline due to the collection of wild individuals for the international pet trade and Asian food consumption. It is believed that most Cuora galbinifrons traded in Viet Nam are exported to Chinese markets.

Increasing economic value incentivises traders and has ensured that hunting pressure is sustained despite the increasing rarity of the species. In the last decade, due to the high economic value, consumption has largely ceased with most animals now sold into trade. According to the CITES Trade Database, *Cuora galbinifrons* was predominantly traded internationally as live specimens. Since the species was listed in 2000, there has been limited trade recorded. Table 1 summarises the direct trade of live individuals reported in the CITES Trade Database for *Cuora galbinifrons* (until 2013 this will have included trade in the taxa now recognised as full species). The majority of trade has been reported in captive-bred specimens. There are significant discrepancies between the quantities reported by importers and exporters; exporters have not reported any trade in wild-caught specimens, although importers have reported over 300 individuals. No trade in wild-caught specimens has been reported since the zero quota came into force in 2013.

Table 1. Summary of CITES trade reported by importers and exporters for live *Cuora galbinifrons* reported between 2000–2020 (CITES Trade Database, 2022).

| Source | Importer reported quantity | | Exporter reported quantity | |
|-----------------------------|----------------------------|-----------|----------------------------|-----------|
| | 2000–2013 | 2014–2020 | 2000–2013 | 2014–2020 |
| Captive-bred (C) | 695 | 8 | 791 | 20 |
| Captive-born (F) | 6 | 2 | 2 | 2 |
| Seized/confiscated (I) | 10 | | | |
| Ranched (R) | 1,500 | | | |
| Unknown (U) | 3 | | 10 | |
| Wild-sourced (W) | 333 | | | |
| Source not reported (blank) | 6 | | | |
| Total | 2,553 | 10 | 803 | 22 |

Illegal trade is a key threat to the species. Visible illegal trade in *C. galbinifrons* at Hanoi's principal wildlife market, Dong Xuan, stopped by about 2006 due to better market enforcement. Prior to this, dozens of *C. galbinifrons* were regularly available each week, often as juveniles intended for the pet market, not for food.

Over 15,000 *C. galbinifrons* were recorded during 2000–2003 in Hong Kong SAR markets alone. During this same period 905 animals were recorded as exported worldwide, indicating a high volume of illegal and unrecorded trade.

A study by Sung et al. (2021) found that between 2017 and 2018, 68 individual *C. galbinifrons* were recorded for sale; 20 advertised on social media (Hong Kong SAR based Facebook group specialising in turtle trade), 13 on an internet forum (publicly accessible Hong Kong SAR based forum containing advertisements and husbandry advice) and 35 sold in a physical market (Goldfish Market the largest pet market in Hong Kong SAR).

Additional information

Threats

The primary threat to *Cuora galbinifrons* has been collection for trade. Collection efforts include both targeted searches for turtles involving trained dogs, occasionally pitfall traps, as well as capitalising on casual turtle encounters when collecting other forest products. Turtles, of any species, are collected whenever and wherever encountered, 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.

All age classes except hatchlings are seen in trade; juveniles are normally kept at villages as traders prefer not to buy very small individuals due to high mortality. This species was present in nearly every reported turtle trade market survey in China and Hong Kong SAR since recording began in 1993. All these animals appeared wild-caught and most were offered in the food markets. There is a demand from commercial turtle farms for wild-caught turtles for founder stock, which is driving the collection of wild individuals through increased trade prices.

Hybrid turtles represent a small portion of trade but are highly priced. Hybrid turtles seem to be attractive to hobbyists due to their high variability in appearance as "designer" turtles. Hybrids of *Cuora galbinifrons* and *mouhoutii* were over nine times more expensive than the price for the parent species. Under CITES, the trade of hybrids is regulated if either of the parents is listed in the Appendices and the more restrictive Appendix applies if parents are listed in different Appendices (Sung and Fong, 2018).

Habitat loss and degradation are considered a significant but more localised threat to the species (Stuart and Timmins, 2000).

Conservation, management and legislation

Populations of *Cuora galbinifrons* are not known to be managed.

China: *Cuora galbinifrons* is included in the list of National Protected Terrestrial Wild Animals that are Beneficial, or with Important Economic and Scientific Research Value. The collecting of state major protected species is only allowed for scientific research, captive breeding, exhibition and other special reasons.

Lao PDR: *Cuora galbinifrons* is listed under Prohibited Category I of the Wildlife and Aquatic Species Law (No07/NA 24 December 2007), the highest protective category, banning hunting and collecting year-round.

Viet Nam: *Cuora galbinifrons* is banned from exploitation or use for a commercial purpose, listed in Group 1B under the Decree 06/2019/ND-CP issued 22 January 2019 on the management of endangered, precious, and rare species of forest fauna and flora and observation of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, and DECREE 84/2021/ND-CP issued on 22 September 2021 amendments to the government's decree No. 06/2019/ND-CP dated 22 January 2019 on the management of endangered, rare, and precious species of forest fauna and flora and observation of Convention on International Trade in Endangered Species of Wild Fauna and Flora (Nguyen, in litt., 2022).

Cuora galbinifrons has been assessed on the IUCN Red List as Critically Endangered since 2000. The initial assessment prepared included *C. bourreti* and *C. picturata* as subspecies of *C. galbinifrons*. An updated Red List assessment of *C. galbinifrons* (excluding *C. bourreti* and *C. picturata* as subspecies) as Critically Endangered under Red List Criteria A2bd+4bd was published in 2020 (Li et al., 2020). *Cuora galbinifrons* was not included in the 1982 Amphibia-Reptilia Red Data Book; it was listed as Insufficiently Known (suspected to be threatened) from 1988 to 1994, before being assessed as Lower Risk: near threatened in 1996. The reclassification of the Red List threatened status reflects the rapid escalation of its exploitation combined with its intrinsic biological vulnerability to exploitation. Since 2000, *Cuora galbinifrons* has consistently been included in the list of 50 species of tortoises and freshwater turtles at the highest risk of extinction (Turtle Conservation Coalition, 2011, 2018).

Captive breeding

Cuora galbinifrons is maintained in modest numbers in captivity by zoos, institutions and private hobbyists in Asia, Europe, North America, and there is a European studbook with over 150 registered animals. This species 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. It is slow to mature, produces small clutches, and there is high mortality in eggs and juveniles. The Turtle Conservation Centre at Cuc Phuong National Park (Viet Nam) maintains around 30 *Cuora galbinifrons*, *C. bourreti* and *C. picturata*. They have been breeding them with limited success, with low survival in eggs and lower long-term survival of hatchlings.

Implementation challenges (including similar species)

Cuora galbinifrons is easily distinguished from *C. bourreti* and *C. picturata* by its plastron colouration, which is solid black in *C. galbinifrons*, while bony yellow with a large black blotch on each scute in *C. bourreti* and *C. picturata*.

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Inclusion of Neotropical Wood Turtles *Rhinoclemmys* spp. in Appendix II

Proponents: Brazil, Colombia, Costa Rica, Panama

Summary: Neotropical Wood Turtles *Rhinoclemmys* spp. occur in central and northern South America and include nine recognised species characterised by colourful patterns on their limbs, head, and carapace. The species can be found in riparian forests, streams, and Neotropical forests, with some species being more aquatic and others more terrestrial. *Rhinoclemmys* are slow-growing species and have a low reproductive output although they are reported to be relatively easily bred in captivity. The major identified threats to *Rhinoclemmys* include human development, contamination of waterways and fires, leading to habitat degradation and loss. *Rhinoclemmys* species are in demand for the pet trade, with juveniles and sub-adults thought to be more desirable, and human food. None of the species is currently included in the Appendices.

Five of the nine species have been assessed as Near Threatened by IUCN, however these assessments date from between 1996 and 2007. Population decreases have been inferred for *R. areolata* and *R. rubida*. No complete population estimates exist for any species, although some localised estimates have been made, often from protected areas, with indications of local abundance in several cases (*R. areolata*, *R. nasuta*, *R. pulcherrima*, *R. punctularia* and *R. rubida*). All species have relatively extensive presumed original distributions, ranging from ca 44,000 km² (*R. diademata*) to over 2 million km² (*R. punctularia*). Available trade information is largely restricted to records of imports and exports to the USA (LEMIS data) with some limited export data from range States. Online surveys have found several of the species offered for sale in Europe, apparently in small numbers. There is also some trade to East Asia.

- ***R. annulata*:** Occurs from Costa Rica south to Ecuador. Assessed as Near Threatened on the IUCN Red List in 1996, provisionally assessed as data deficient by the Tortoise and Freshwater Turtle Specialist Group (TFTSG) in 2011 and 2018. Assessed as nationally endangered in Ecuador. Minimal trade reported.
- ***R. areolata*:** Occurs from Mexico to Honduras. Assessed as Near Threatened on the IUCN Red List in 2007 with an inferred decreasing population trend. Assessed as nationally threatened in Mexico. Minimal trade reported (just under 1,000 exported from Mexico between 2013 and 2021).
- ***R. diademata*:** Occurs in Colombia and Venezuela. Not assessed on the IUCN Red List, provisionally assessed as vulnerable by the TFTSG in 2011. Has the smallest presumed range of all *Rhinoclemmys* species (44,000 km²) and is endemic to the Maracaibo River basin. Its habitat is considered threatened. Assessed as nationally endangered in Colombia and threatened in Venezuela. Limited trade has been reported into the USA (just over 700 captive-sourced individuals imported between 2008 and 2017, most from Nicaragua (not a range State)).
- ***R. funerea*:** Occurs from Honduras to Panama. Assessed as Near Threatened on the IUCN Red List in 1996. Limited trade to or from the USA reported in 2008–2020, including just under 700 exported as captive-sourced.
- ***R. melanosterna*:** Occurs in Colombia, Ecuador, and Panama. Not assessed on the IUCN Red List, provisionally assessed as least concern by the TFTSG in 2011. Nationally assessed as near threatened in Colombia and endangered in Ecuador. Minimal trade reported with the USA.
- ***R. nasuta*:** Occurs in Colombia and Ecuador. Assessed as Near Threatened on the IUCN Red List in 1996, and provisionally assessed as near threatened by the TFTSG in 2010. Assessed nationally as near threatened in Colombia and endangered in Ecuador.
- ***R. pulcherrima*:** Occurs from Mexico to Costa Rica. Not assessed on the IUCN Red List. Assessed as nationally threatened in Mexico. Reported to be the most abundant *Rhinoclemmys* sp. in trade with one subspecies, *R. p. manni* occurring in Nicaragua and

Costa Rica, being the most colourful and desirable. US trade data indicate imports of approximately 8,000 wild-sourced and about 64,000 captive-sourced live individuals, and re-exports of about 28,000 wild, and 34,000 captive-sourced live individuals in the period 2008–2020.

- ***R. punctularia***: Occurs from Venezuela to Brazil. Not assessed on the IUCN Red List, provisionally assessed as least concern by the TFTSG in 2011. Has a large presumed range of over 2 million km². Between 2008 and 2020 the USA reported imports of around 7,000 wild-sourced and 450 captive-bred animals, and exports of approximately 3,000 wild-sourced and 440 captive-bred animals. The majority was exported to China and Hong Kong SAR, and imports were mainly from Guyana and Suriname.
- ***R. rubida***: Endemic to Mexico. Assessed as Near Threatened on the IUCN Red List in 2007 with an inferred decreasing population trend. Nationally under special protection in Mexico. Endemic to Mexico with a presumed range of ca 80,000 km². Reported trade volumes are small (fewer than 300 in total, including export of 280 from Mexico between 2019 and 2021).

Experts note that the species are easily distinguished with minimal training, however, customs officials may need special training or guidance. Some records of wild-sourced individuals reported in direct trade are from non-range States. The species are considered to be easily bred in captivity.

Analysis: There is very little information on current population levels of any *Rhinoclemmys* species. Most are believed to have relatively extensive ranges and at least some to be locally common. Available trade data are limited and very largely confined to US import and export data. Drawing on this source, the only species for which there is any indication of extensive trade are *R. pulcherrima* and *R. punctularia*. The very extensive distribution of *R. punctularia* indicates that this species is also unlikely to meet the criteria for inclusion in Appendix II. Regarding *R. pulcherrima* there is insufficient information on the scale and impact of trade to determine whether or not the species meets these criteria. Given the low levels of known trade and their generally extensive distributions, it also seems unlikely that any of the other seven species currently meet the criteria for inclusion in Appendix II set out in Res. Conf. 9.24 (Rev. CoP17).

Experts note that the species are easily distinguished with training; there would therefore be little justification for listing species for lookalike reasons (Annex 2b A in 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

A total of nine species are recognised (TTWG, 2021): *Rhinoclemmys annulata* (Gray, 1860), *Rhinoclemmys areolata* (Duméril & Bibron, 1851), *Rhinoclemmys diademata* (Mertens, 1954), *Rhinoclemmys funerea* (Cope, 1876), *Rhinoclemmys melanosterna* (Gray, 1861), *Rhinoclemmys nasuta* (Boulenger, 1902), *Rhinoclemmys pulcherrima* (Gray, 1856), *Rhinoclemmys punctularia* (Daudin, 1801), and *Rhinoclemmys rubida* (Cope, 1870).

There are additional subspecies of R. pulcherrima, R. punctularia, and R. rubida of which some are recorded in trade by their subspecies name instead of their binomial name (TTWG, 2021; LEMIS, 2022): R. pulcherrima pulcherrima (nominate subspecies), R. pulcherrima incisa, R. pulcherrima manni R. pulcherrima rogerbarbouri, R. punctularia punctularia (nominate subspecies), R. punctularia flammingeri, R. rubida rubida (nominate subspecies), R. rubida perixantha.

A taxonomic and phylogenetic revision of the genus is needed as some Rhinoclemmys subspecies may require elevating to species level and subsequently possess very small distributions (Macip Ríos, in litt., 2022).

Range and IUCN Global Category

See Table 1.

Table 1. Range, IUCN Global Category and year of assessment and population trend (▼ indicates decreasing), provisional Red List assessment and year of assessment (TFTSG), and presumed historic indigenous range (TTWG, 2021). IUCN Red List assessments are version 3.1 for those assessed in 2007, and version 2.3 for those from 1996. National status as reported in the Supporting Statement and from Páez *et al.* (2022).

| Species | Range States | IUCN Red List and population trend | TFTSG provisional Red List | Presumed historic indigenous range (km ²) | National status |
|------------------------|---|------------------------------------|----------------------------|--|--|
| <i>R. annulata</i> | Colombia, Costa Rica, Ecuador, Honduras, Nicaragua, Panama | Near Threatened (1996) | data deficient (2018) | 310,269 | Colombia: least concern Ecuador: endangered |
| <i>R. areolata</i> | Belize, Guatemala, Honduras, Mexico | Near Threatened (2007) ▼ | | 288,385 | Mexico: threatened |
| <i>R. diademata</i> | Colombia, Venezuela | | vulnerable (2011) | 44,327 | Colombia: endangered Venezuela: vulnerable |
| <i>R. funerea</i> | Costa Rica, Honduras, Nicaragua, Panama | Near Threatened (1996) | | 103,734 | |
| <i>R. melanosterna</i> | Colombia, Ecuador, Panama | | least concern (2011) | 225,817 | Colombia: near threatened Ecuador: endangered |
| <i>R. nasuta</i> | Colombia, Ecuador | Near Threatened (1996) | near threatened (2010) | 84,784 | Colombia: near threatened Ecuador: endangered |
| <i>R. pulcherrima</i> | Costa Rica, El Salvador, Guatemala, Honduras, Mexico, Nicaragua Introduced: USA (Florida) | | least concern (1996) | 259,172 | Mexico: threatened |
| <i>R. punctularia</i> | Brazil, France (French Guiana), Guyana, Suriname, Trinidad and Tobago, Venezuela Introduced: USA (Florida) | | least concern (2011) | 2,201,321 (in the top 50 of turtle taxa with largest ranges globally) | |
| <i>R. rubida</i> | Mexico | Near Threatened (2007) ▼ | data deficient (2018) | 78,865 | Mexico: under special protection |

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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

Few data exist on *Rhinoclemmys* population sizes, trends, and structures. The Supporting Statement notes that *Rhinoclemmys* habitats are threatened in many parts of their range due to increased development. In addition, local populations of some species are likely to have decreased, due to rising commercial interest in the genus over the last decade. River contamination was additionally mentioned as a possible driver of current and future decreases in local populations of aquatic species.

Rhinoclemmys turtles are slow-growing species with a low reproductive rate and have been described as having a great risk of exploitation for the pet trade. *Rhinoclemmys* turtles show a pattern of several clutches of a few eggs per year. Most species lay one or two large, elongated eggs per clutch. The species are sexually dimorphic, with females larger than males. *It is unclear whether larger specimens are favoured in trade, and whether this leads to more offtake of females.*

Available information is summarised below in Table 2.

Table 2. Summary of available population trends for *Rhinoclemmys* spp.

| Species | Biological data | Population information |
|------------------------|--|--|
| <i>R. annulata</i> | | No population density data available. Species reported to be vulnerable to overexploitation and habitat loss due to low population density, low fertility and slow growth rate. |
| <i>R. areolata</i> | In <u>Belize</u> , <i>R. areolata</i> normally lays one egg per clutch (rarely two to five eggs per clutch), with up to four clutches laid during the nesting season from May to July. | <p>The species was globally assessed as Near Threatened on the IUCN Red List in 2007 with a population decline estimated at around 30% (Van Dijk et al., 2007a).</p> <p>Based on surveys conducted in northern <u>Belize</u> between 1992 and 2000, <i>R. areolata</i> is widespread and abundant, particularly in lowland pine habitats where preliminary estimates suggest densities of 5–6 individuals/ha. Survey data from other parts of its range are lacking, but based on the presence of suitable undisturbed habitat, it may be common in southern Belize. Lowland pine in Belize is under pressure from logging and land use changes.</p> <p>In <u>Mexico</u> the species is frequently seen in Tabasco and Quintana Roo, with the largest and most dense population on Isla Cozumel, but may be found only sporadically elsewhere. It is seen more frequently in the rainy season and may aestivate at other times. Additionally, it is thought that the species is abundant on the Yucatan Peninsula (Macip Ríos, in litt., 2022), with a survey observing 25 individuals in Kaxil Kiuic, including juveniles indicating good recruitment (Enríquez-Mercado, 2022).</p> |
| <i>R. diademata</i> | No biological information available. | No population information available. |
| <i>R. funerea</i> | No biological information available. | No population information available. |
| <i>R. melanosterna</i> | Normally lays a single egg per clutch (up to five eggs recorded) and nests throughout the year. | A study of <i>R. melanosterna</i> in <u>Colombia</u> suggested that populations in freshwater ecoregions are isolated from each other and experiencing limited gene flow. |
| <i>R. nasuta</i> | <i>R. nasuta</i> lays clutches of one or two eggs, usually between January and March. Males estimated to reach sexual maturity at 12 years and females at 14 years. | <p><i>R. nasuta</i> was described as “abundant” in the rivers of the department of Chocó in <u>Colombia</u> in 2009. A 2007 study of a protected island site on the Pacific coast of Colombia, Isla Palma, and a site on the mainland, Playa Chucheros, estimated the insular population at 624 (1,560 individuals/ha) and the mainland site at 99 (248 individuals/ha; Garcés-Restrepo et al., 2013). A later study in 2011–2012 estimated the population density on Isla Palma at 1,467 individuals/ha and 325 individuals/ha at Playa Chucheros, and 372 individuals/ha at San Pedro, a location on the mainland located to the south of Playa Chucheros (Garcés-Restrepo, 2014). The population of Isla Palma showed signs of inbreeding in 2014, and its high population density may reflect the protected status of the island and the absence of predators.</p> <p>In <u>Ecuador</u>, population densities appear much lower, five individuals/ha within the Canandé Reserve, with almost 70% of the sampled population being juveniles, and the sex ratio dominated by females. Torrential rains and</p> |

| Species | Biological data | Population information |
|-----------------------|--|---|
| | | floods can expel individuals of <i>R. nasuta</i> from streams and alter their habitats. Changes in land use and logging have contributed to the loss of streams where populations of these species once existed. |
| <i>R. pulcherrima</i> | In <u>Guatemala</u> <i>R. p. incisa</i> were reported to lay three to five eggs per clutch, with up to four clutches laid per year. In <u>Costa Rica</u> <i>R. p. manni</i> lay between one and six eggs per clutch. In captivity, <i>R. p. manni</i> were observed to lay one egg every six weeks from October to April. | <i>A community monitoring programme in Mexico at the Centro Mexicano de la Tortuga in Mazunte, Oaxaca, considered the population to be healthy at the surveyed sites (Harfush, in litt., 2022).</i> |
| <i>R. punctularia</i> | <i>R. punctularia</i> nests in different months in different parts of its range, and clutches of one to three eggs have been reported. | <i>R. punctularia</i> studied in <u>Algodoal-Maiandeuá Island</u> in <u>Pará, Brazil</u> exhibited female dominated sex ratios in 2008–2009 (Wariss et al., 2011). The same study recorded 155 individuals on the island in 40 days across eight different sites (Wariss et al., 2011). |
| <i>R. rubida</i> | <i>Immature individuals have low survival rates (Harfush, in litt., 2022).</i> | <i>A mark-recapture study of a population of <i>R. r. perixantha</i> in the Chamela-Cuixmala Biosphere Reserve in Jalisco, Mexico, found this subspecies was abundant in suitable habitats (hills in dry forest), with an estimated population density of about 43 individuals/ha. The sex ratio in the population of <i>R. r. perixantha</i> was biased in favour of males (1.5 males: 1 female). However, the captured turtles were almost entirely adults, suggesting that the population may not be stable or that the juveniles use a different type of habitat, although a similar dominance of adults in other populations of <i>Rhinoclemmys</i> has been recorded. <i>A community monitoring programme in Mexico at the Centro Mexicano de la Tortuga in Mazunte, Oaxaca, is in place to monitor <i>R. rubida</i> (Harfush, in litt., 2022). Based on this monitoring programme, the population was considered healthy at the surveyed sites (Harfush, in litt., 2022).</i></i> |

International trade

The Supporting Statement asserts that legal and illegal international trade in *Rhinoclemmys* is increasing and that interest in species of the genus appears to have increased in recent years as other species within the Geoemydidae family become increasingly difficult to obtain. In 2005, imports of *Rhinoclemmys* spp. by European countries were reported to increase, possibly as a cheaper alternative to protected turtle species from other parts of the world with more regulations. International trade in *Rhinoclemmys* is primarily in live individuals for the pet trade, although there may be cross-border traffic in parts and derivatives for sale to tourists, and possibly for meat for sale in local markets. *Juveniles and sub-adults are thought to be more attractive for the pet trade (Páez, in litt., 2022).*

The Supporting Statement notes that trade to supply markets with turtles for human food in Asia have increasingly relied upon species from Africa and the Americas. Commercial impacts on *Rhinoclemmys* spp. have not been studied but could be expected to increase as overharvesting and habitat loss reduce the availability and affordability of Asian Geoemydidae. *For species impacted by habitat loss and degradation due to anthropogenic development as well as climate change, trade at any level may further exacerbate the negative impacts on the species, particularly those with restricted ranges (Harfush, in litt., 2022).* *Rhinoclemmys* species are reported to be impacted by trade *due to their attractive colours, ease of being kept in captivity, and ease with which they can be collected from the wild (Macip Ríos, in litt., 2022).*

Seven species of *Rhinoclemmys* (all except *R. melanosterna* and *R. nasuta*) were recorded for sale on European online platforms in a survey from September 2017 to September 2018 (Table 3).

Table 3. Trade summary for each of the *Rhinoclemmys* species.

| Species | Range States | International trade summary |
|------------------------|--|---|
| <i>R. annulata</i> | Colombia, Costa Rica, Ecuador, Honduras, Nicaragua, Panama | <p>The USA reported imports of 10 live captive-bred individuals for commercial purposes from Nicaragua in 2020 (LEMIS, 2022).</p> <p>An online survey of European platforms in 2017–2018 found advertisements for six captive-bred individuals.</p> |
| <i>R. areolata</i> | Belize, Guatemala, Honduras, Mexico | <p>The USA reported commercial trade in captive-bred live individuals in 2015, with one imported from Singapore and 13 exported to Hong Kong SAR (LEMIS, 2022).</p> <p>Mexico reportedly exported a total of 914 individuals of unknown source between 2013 and 2021.</p> <p>An online survey of European platforms in 2017–2018 found advertisements for five captive-bred individuals.</p> |
| <i>R. diademata</i> | Colombia, Venezuela | <p>The USA reported commercial trade in captive-sourced live individuals in 2008–2017, reporting imports of 702 (14% originating from Venezuela and 85% from Nicaragua), all captive-bred, and exports of 94 (53% originating in Venezuela) to Japan (54%), Hong Kong SAR (35%), Republic of Korea (6%), and Taiwan POC (4%), of which 27 were captive-born (origin: USA) and the remainder captive-bred (LEMIS, 2022).</p> <p>An online survey of European platforms in 2017–2018 found advertisements for three captive-bred individuals and 11 from undisclosed sources.</p> |
| <i>R. funerea</i> | Costa Rica, Honduras, Nicaragua, Panama | <p>The USA reported commercial trade, including imports of 255 live captive-bred individuals (87% from Nicaragua and 13% from Costa Rica), and exports of 84 wild-sourced (77% from the USA, 14% from Guatemala, 5% from Suriname, and 4% from Nicaragua) and 778 captive-sourced live individuals (86% from the USA, 14% from Nicaragua). Main reported importers were Hong Kong SAR (40%), followed by Japan (15%) and the EU27 (14%; LEMIS, 2022). The USA is not a range State.</p> <p>An online survey of European platforms in 2017–2018 found advertisements for two captive-bred individuals and 14 from undisclosed sources.</p> |
| <i>R. melanosterna</i> | Colombia, Ecuador, Panama | <p>The USA reported imports of eight live, captive-bred individuals from Hong Kong SAR (origin reported as Japan) in 2015, and reported exports of three live captive-bred individuals in 2020 destined for the Netherlands and originating in the USA (LEMIS, 2022).</p> |
| <i>R. nasuta</i> | Colombia, Ecuador | <p>The USA did not report commercial trade in this species (LEMIS, 2022).</p> <p>One anecdotal report exists of an import of <i>R. nasuta</i> into the USA via Peru, a non-range State, prior to 2009.</p> |
| <i>R. pulcherrima</i> | Costa Rica, El Salvador, Guatemala, Honduras, Mexico, Nicaragua Introduced: USA (Florida) | <p><i>R. pulcherrima</i> has been described as the most common species in trade, specifically <i>R. p. manni</i>, as the most colourful of the <i>Rhinoclemmys</i> turtles.</p> <p>In 2008–2020, the USA reported exports of 27,587 wild-sourced live individuals, primarily originating from El Salvador (30%) and Nicaragua (22%), as well as from non-range States including the USA (40%), Cabo Verde (7%), and low levels (<1%) from Canada, Guyana, Nigeria, and Suriname. The vast majority was exported to China (68%) and Hong Kong SAR (28%). The USA additionally reported commercial exports of 34,046 captive-sourced live individuals, with 50% originating in the USA, 25% in Nicaragua, 19% in El Salvador, 6% in Cabo Verde, and >1% in Guyana and Panama. The majority was exported to China (73%) and Hong Kong SAR (19%; LEMIS, 2022).</p> |

| Species | Range States | International trade summary |
|-----------------------|--|--|
| | | <p>The USA additionally reported commercial imports of 7.765 wild-sourced live individuals, primarily from El Salvador (75%) and Guatemala (14%), and Nicaragua (6%), as well as lower levels from non-range States with 4% from Guyana and <1% from Suriname. Imports of 63,661 captive-bred live individuals, of which the majority originated from Nicaragua (67%) and El Salvador (33%). Imports (LEMIS, 2022).</p> <p>It may be that some of this trade was misdeclared <i>P. punctularia</i>, in particular those records reported as originating from Suriname and Guyana.</p> <p>Mexico reportedly exported a total of 126 individuals of unknown source between 2019 and 2021.</p> <p>An online survey of European platforms in 2017–2018 found advertisements for six wild-caught and 18 captive-bred individuals and 53 from undisclosed sources, stating a price of EUR300 (USD301) for one individual. Some merchants on European online platforms offered "<i>R. incisa</i>" as a species rather than a subspecies of <i>R. pulcherrima</i>.</p> <p>There are reports of wild-caught <i>R. p. manni</i> in the pet trade in Costa Rica. <i>R. pulcherrima</i> was reported in pet stores in the Republic of Korea and has been advertised for sale in a Facebook group in the Philippines.</p> <p><i>R. pulcherrima</i> was described as "a great turtle to keep if you can find one" in Reptiles Magazine in 2011 (Spinner, 2011).</p> |
| <i>R. punctularia</i> | <p>Brazil, France (French Guiana), Guyana, Suriname, Trinidad and Tobago, Venezuela</p> <p>Introduced: USA (Florida)</p> | <p>The USA reported commercial trade in live individuals 2008–2020. Exports totalled 442 from captive sources (68% originating from the USA and 32% from Guyana) and 2,911 from wild sources, with 54% originating from Guyana, 10% from Suriname, and <1% from Venezuela, as well as trade reported as originating from non-range States including the USA (33%), Saudi Arabia (2%), and Nicaragua (1%). Exports were primarily to China (67%) and Hong Kong SAR (15%), as well as EU27 (6%; LEMIS, 2022).</p> <p>In the same period, the USA reported imports of 450 captive-bred live individuals from Nicaragua, as well as 6,649 wild-sourced live individuals originating from Guyana (90%) and Suriname (10%), with some indirectly imported via Hong Kong SAR <0.5% (LEMIS, 2022).</p> <p>It may be that some of this trade was misdeclared <i>P. pulcherrima</i>, in particular those records reported as originating from Nicaragua.</p> <p>An online survey of European platforms in 2017–2018 found advertisements for seven individuals from undisclosed sources.</p> <p><i>R. punctularia</i> has been offered for sale in Taiwan POC in a 2004–2005 survey and has been reported for sale in pet stores in the Republic of Korea.</p> |
| <i>R. rubida</i> | Mexico | <p>The USA reported commercial trade in captive-sourced individuals, with 14 imported (from Germany [11] and the Czech Republic [3]), and 236 exported in 2013–2017. All exported individuals were reported as originating in the USA and were exported primarily to Hong Kong SAR (63%) and Japan (16%). One wild-sourced live individual was seized on import in 2018, originating from Mexico (LEMIS, 2022).</p> <p>Mexico reportedly exported a total of 280 individuals of unknown source between 2019 and 2021, increasing significantly from 2020–2021 (see Figure 2).</p> |

| Species | Range States | International trade summary |
|---------|--------------|---|
| | | An online survey of European platforms in 2017–2018 found an advertisement stating a price of EUR500 (USD502) for an undisclosed quantity of individuals. |

A total of 125 *Rhinoclemmys* individuals were observed, 34 of which were advertised as captive-bred, six as wild-sourced, and a further 85 had no source specified. The cost for an individual ranged between EUR 50–EUR 500 (*R. rubida*; USD50–USD502). A website based in California (USA) described *R. pulcherrima* as one of the most common wood tortoises seen in the pet trade noting that they have been imported in large numbers in recent years. This site also noted that the animals often arrive at importers with some scratches and cuts, the most serious problems being shell rot and lack of scales. *This may be indicative of wild-sourced individuals.*

All species except one (R. nasuta) have been recorded in USA-reported trade. One report exists of an import of *R. nasuta* to the USA via Peru, a non-range State. According to the LEMIS database (2022), a total of 79,757 live *Rhinoclemmys* were reportedly cleared and imported into the USA between 2008 and 2020, 18% of which were reported as wild-sourced⁶ (14,465 live individuals) and 82% from captive sources⁷ (65,292 live individuals; Table 4). In addition, the USA reported exports of 67,288 live individuals, of which 47% were wild-sourced and 53% were from captive sources (Table 4; LEMIS, 2022). Trade reported by the USA has declined markedly from 2015 onward (Figure 1; LEMIS, 2022). The majority (>99%) of this trade was reported for commercial purposes (LEMIS, 2022). The majority of trade reported was in *R. pulcherrima* with lower levels of trade in *R. punctulata* overall but in similar numbers for wild imports (Table 4).

Table 4. Imports, exports and transit of live individuals of *Rhinoclemmys* reported by the USA between 2008–2020.

| Species | Imports to USA | | Exports from USA | | Transit in USA | |
|--|----------------|-----------------|------------------|-----------------|----------------|-----------------|
| | Wild sources | Captive sources | Wild sources | Captive sources | Wild sources | Captive sources |
| <i>Rhinoclemmys</i> spp. (reported at Genus level) | 15 | 190 | 788 | 306 | | |
| <i>Rhinoclemmys annulata</i> | | 10 | | | | |
| <i>Rhinoclemmys areolata</i> | | 1 | | 13 | | |
| <i>Rhinoclemmys diademata</i> | | 702 | | 94 | | |
| <i>Rhinoclemmys funerea</i> | | 255 | 84 | 778 | | |
| <i>Rhinoclemmys melanosterna</i> | | 8 | | 3 | | |
| <i>Rhinoclemmys pulcherrima</i> | 7,765 | 63,662 | 27,587 | 34,046 | 600 | 2,170 |
| <i>Rhinoclemmys punctulata</i> | 6,685 | 450 | 2,911 | 442 | 210 | |
| <i>Rhinoclemmys rubida</i> | | 14 | | 236 | | |

Wild sources include trade reported as wild (W), ranches (R), and unknown (U), and captive sources include captive-bred (C) and captive-born (F). All trade was reported by number, and only cleared records are included. No data indicates that no trade was reported in those species/source combinations (LEMIS, 2022).

China and Hong Kong SAR were major reported destinations for (re-)exports from the USA, for both captive and wild-sourced live *Rhinoclemmys*, accounting for 90% of destinations for captive-sourced turtles, and 95% of wild-sourced (LEMIS, 2022). EU27 Member States were the next highest reported destination and accounted for 3% of captive-sourced and 1.5% of wild live *Rhinoclemmys* from the USA (LEMIS, 2022). The third-highest reported destination was Japan, accounting for 2% of captive and 1% of wild-sourced trade (LEMIS, 2022).

Imports of wild-sourced live *Rhinoclemmys* originated primarily from Guyana, El Salvador, and Guatemala (LEMIS, 2022). Captive-sourced individuals mainly originated from Nicaragua (68%, 41,000 individuals) and El Salvador (32%, 20,961 individuals), with low levels (<0.5%) from Venezuela, Costa Rica, and Mexico. In addition, some imports (0.1%) from non-native States were reported (LEMIS, 2022).

⁶ Wild-sourced includes the source codes "W" (Wild), "R" (Ranches), and "U" (Unknown) throughout this analysis.

⁷ Captive-sourced include the source codes "C" (captive-bred) and "F" (captive-born) throughout this analysis.

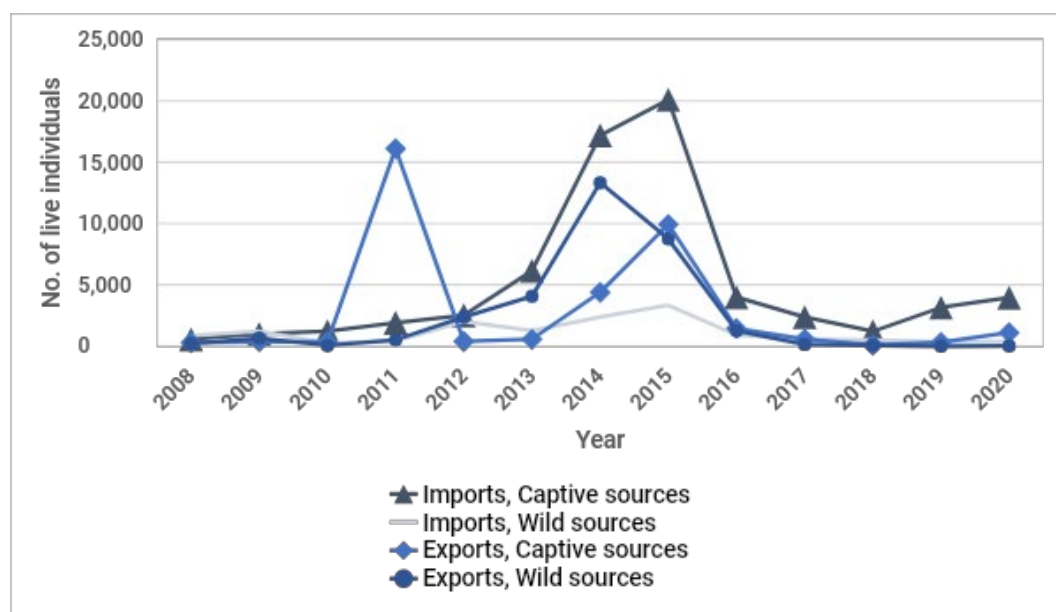


Figure 1. Imports and exports of live individuals of *Rhinoclemmys* reported by the USA between 2008–2020. Wild sources include trade reported as wild (W), ranched (R), and unknown (U), and captive sources include captive-bred (C) and captive-born (F). All trade was reported by number, and only cleared records are included (LEMIS, 2022).

Exports from Mexico from 2000–2021 are summarised in Table 5 and Figure 2. No trade was reported between 2000 to 2012, so this is not included in the table. Reported exports increased from 2015, with a sharp decrease in 2019, followed by a sharp increase in 2020. *Rhinoclemmys pulcherrima* and *R. rubida* were first registered in exports in 2019. It is unknown which countries these were exported to, and where the 2,000 turtles originated. No information was available on the source of this trade.

An annex to the proposal includes data submitted by the Mexican CITES Authorities, stating that verified exports of *Rhinoclemmys* (reported at the genus level) for commercial purposes from 2018–2022 totalled 298 individuals.

Table 5. Imports and exports of *Rhinoclemmys* spp. notified and registered by Mexico 2013–2021 (SEMARNAT, 2022).

| Species | Imports | Exports |
|--|---------|---------|
| <i>Rhinoclemmys</i> spp. (reported at Genus level) | 2,000 | |
| <i>Rhinoclemmys areolata</i> | | 914 |
| <i>Rhinoclemmys pulcherrima</i> | | 126 |
| <i>Rhinoclemmys rubida</i> | | 280 |

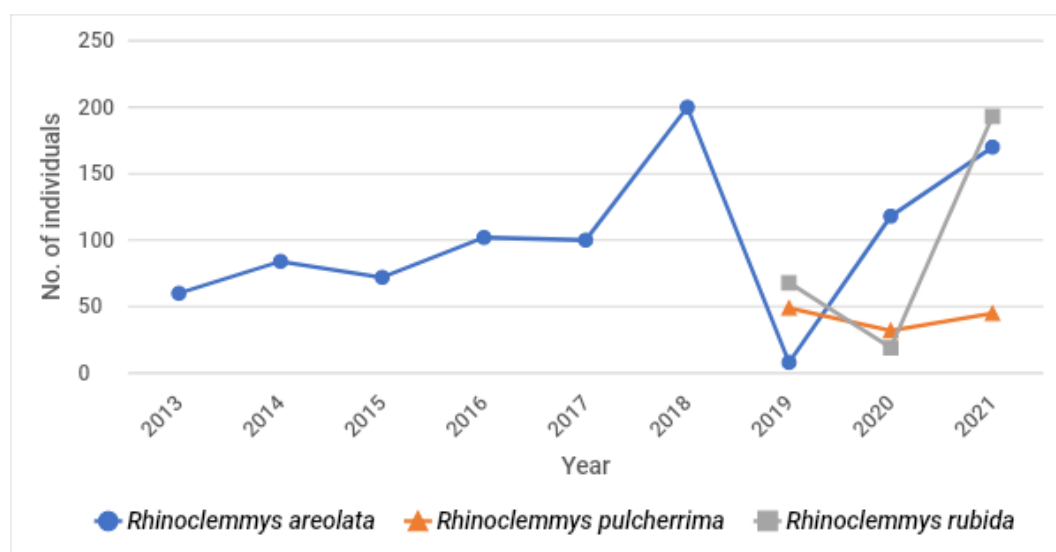


Figure 2. Exports of *Rhinoclemmys* reported by Mexico 2013–2021 (SEMARNAT, 2022).

Seizure records

The USA reported four seizures at import, involving *Rhinoclemmys pulcherrima*, *R. rubida*, and *Rhinoclemmys* spp. (reported at genus level), totalling one live *R. rubida* originating from Mexico in 2018, 37 scientific specimens of *R. pulcherrima* from El Salvador in 2015, and one carapace and one body reported at the genus level, originating from Guatemala in 2015 and Nicaragua in 2011 respectively (LEMIS, 2022).

There are six seizure incident records involving *Rhinoclemmys* in WiTIS (2022). Four incidents involved *Rhinoclemmys pulcherrima* seized in Mexico (two seizures involving four live individuals and one seizure of 266 live animals including an undisclosed number of live *Rhinoclemmys*) and Brazil (one seizure involving 57 live turtles of the genera *Rhinoclemmys* and *Kinosternon*), and two *R. areolata* seized in Guatemala (six live individuals) and Mexico (eight live individuals; WiTIS, 2022). It is unclear whether these seizures involved internationally traded specimens. There are no seizure records in EU-TWIX between 2011 and 2020 (EU-TWIX, 2022).

Between 1999 and 2021, Mexico reported seizures of *R. areolata* (649), *R. pulcherrima* (484), and *R. rubida* (33), with numbers seized increasing over the years, with some reports of turtles being destined for Europe. Similarly, in Colombia, seizures have increased 22-fold over five years, with ~3% of confiscations involving *Rhinoclemmys* species between 2005 and 2009.

It is unclear why any of the above-mentioned seizures were made, whether they were destined for international trade, and why they were deemed illegal.

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
- B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Additional information**Threats**

Rhinoclemmys annulata and *R. melanosterna* are primarily threatened by habitat loss and fragmentation. *Rhinoclemmys annulata* is additionally impacted by contamination of waterways due to the extraction of gold, the construction of highways, and human settlements. The habitats of *R. diademata* are considered endangered (dry forests of Maracaibo) and critically endangered (humid forests of Catatumbo), and the species is additionally impacted by habitat loss for agriculture and possibly oil extraction. In Costa Rica, boat traffic as a result of increased ecotourism may disturb the roost of *R. funerea*. Unnatural fires and drought may affect mortality rates of *R. punctularia* in south-eastern Brazil. *Rhinoclemmys rubida* and *R. pulcherrima* in Mexico also are affected by habitat loss in the Pacific (Macip Ríos, in litt., 2022). Seasonal deciduous forests are facing fragmentation and reduction throughout the range of these species (Macip Ríos, in litt., 2022).

Rhinoclemmys turtles are utilised by rural and indigenous peoples in many parts of their range and are used as a source of traditional medicine in some areas. Local trade, including sale as pets or as tourist souvenirs, has been recorded for several species. It is unclear whether this is a threat to any of the species.

Conservation, management, and legislation

Mexico: *Rhinoclemmys areolata* and *R. pulcherrima* are included in Category A (threatened) and *R. rubida* in Category "Pr" (under special protection) in the List of Species at Risk of Mexico (NOM-059- Semarnat-2010), however legal and sustainable trade in these species is still permitted. Pursuant to Article 420 of the Federal Penal Code, illegal trade in, or damage to, wild specimens can be prosecuted with a fine and prison time.

A community monitoring programme in Mexico at the Centro Mexicano de la Tortuga in Mazunte, Oaxaca, is in place to monitor *R. pulcherrima* and *R. rubida* (Harfush, in litt., 2022).

Captive breeding

Little information is available on the population management or trade of *Rhinoclemmys* species in their various range States. In the past, *Rhinoclemmys* species were considered difficult to keep in captivity. In Europe, animals entering the pet trade were said to be in poor health. It is unclear whether this poor health is indicative of wild-sourced specimens or disease. Mature females of all captive *Rhinoclemmys* species appear to be prone to egg attachment, often leading to death within about a year. However, there are records of the species in captive breeding centres as well as records suggesting they are easy to breed.

USA: *Rhinoclemmys pulcherrima* was described in 2003 as widely available in the USA through the reptile trade in recent years, where individuals were reported as easily and cheaply purchased from breeders, dealers, and chain pet stores in the USA. They were reported as easily kept in good health, as long as the proper conditions and diet are provided. A website based in California (USA) described *R. p. manni* and *R. funerea* as easy to breed.

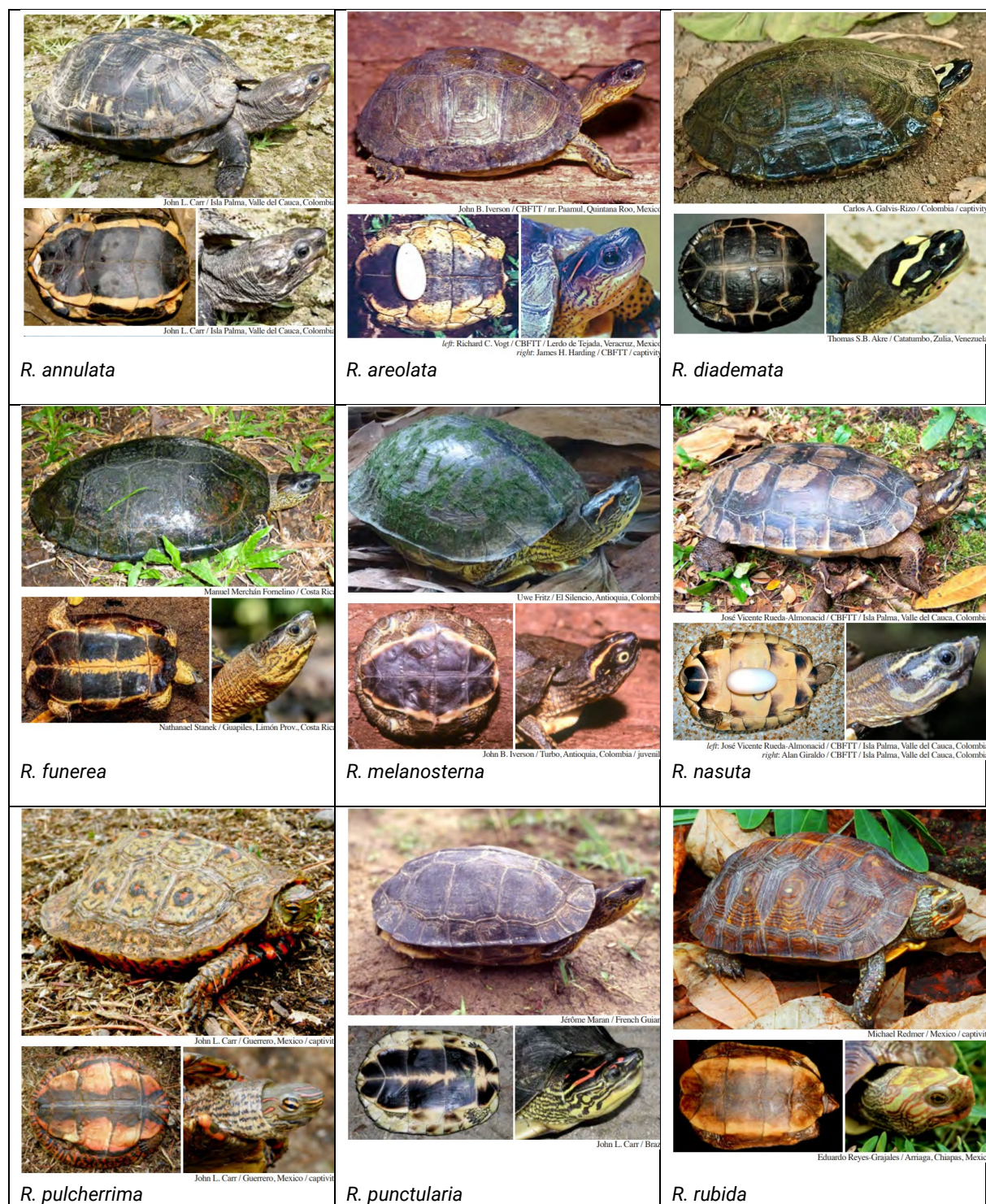
Brazil: *Rhinoclemmys punctularia* has been bred in Brazilian zoos, including Belém Zoo, which maintained a population of more than 200 individuals in the 1990s.

Mexico: There are 18 Management Units for the Conservation of Wildlife (UMA) in Mexico that maintain the three Mexican species (*R. areolata*, *R. pulcherrima* and *R. rubida*) in captivity, with some units raising turtles for trade. *Rhinoclemmys areolata* is bred in captivity at the Granja de Tortugas UMA and a tortoise breeding centre in Nacajuca, Tabasco. There is a breeding population of *R. rubida* at the (non-profit) Centro Mexicano de la Tortuga. Another centre, Turtle in Mazunte in Oaxaca, was robbed in 2005 with 90% of the captive colony stolen; it was presumed that this was to supply the pet trade. *A breeding facility in Oaxaca was believed to be highly successful in breeding the species (Van Dijk et al., 2007). Rhinoclemmys areolata was reported to breed easily in captivity in home gardens in Yucatan Peninsula (Enríquez-Mercado, 2022).*

In Coatzacoalcos, Veracruz and in the Mexican Turtle Center Veracruz, successful reproduction of R. areolata has been achieved (Harfush, in litt., 2022). In Mazunte in the 2021–2022 season more than 60 hatchlings were produced (Harfush, in litt., 2022).

Implementation challenges (including similar species)

Species of the genus are thought to be easily distinguished from each other; however, subspecies are more difficult to identify (Macip Ríos, in litt., 2022). Rhinoclemmys spp. occurring in Mexico are easily distinguished, requiring minimal training (Harfush, in litt., 2022). Species occurring in Colombia are thought to be easily distinguished, however with no training, there is a possibility that species could be misidentified with other species in the genus, or with species of the genus Trachemys (Páez, in litt., 2022), which are currently not listed in the CITES Appendices. It was noted that while species can be distinguished, customs officials would need training to do so (Macip Ríos, in litt., 2022).

Figure 3. *Rhinoclemmys* spp. turtles (TTWG, 2021).

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Inclusion of Narrow-bridged Musk Turtle *Claudius angustatus* in Appendix II

Proponent: Mexico

Summary: The Narrow-bridged Musk Turtle *Claudius angustatus* is a medium-sized (10–12 cm carapace length) semi-aquatic turtle occurring in Mexico, Belize, and Guatemala, where it is found in a wide variety of permanent or semi-permanent freshwater aquatic habitats including ponds in agricultural areas and flooded pastures. *C. angustatus* was classified as Lower Risk/Near Threatened by IUCN in 1996. It is classified as “at risk of extinction” in Mexico and as endangered in Guatemala due to inferred population declines and local extinctions.

The species *Claudius angustatus* is proposed for inclusion in Appendix II under the criteria in Annex 2a paragraph A of Resolution Conf. 9.24 (Rev. CoP17).

There are no estimates of the global population. The majority of the range is in Mexico where it has a projected distribution of around 74,000 km². Local extinctions have been observed. Measured density levels in south-eastern Mexico in 2012–2014 varied from 2–16 ind./ha. There are anecdotal reports of population declines in areas where the species has been harvested in the past. The primary impacts on the species were reported to be illegal collection and trade of wild specimens as meat for human food and the pet trade, with additional impacts from habitat loss due to agricultural practices, livestock and urban development.

In Mexico, the species is mostly sold for its meat with local estimates of substantial harvest for this purpose (e.g., 4000–5000 harvested annually in the region of Lerdo de Tejada, Veracruz state). The species is also locally traded for use in traditional medicine.

Claudius angustatus is apparently among the most traded Mexican turtle species, along with *Terrapene yucatana*, *T. mexicana*, *Staurotypus triporcatus*, and *S. salvinii* (the last two being the subject of CoP19 Prop. 30). Although under Mexican legislation the species can be harvested from the wild under specific circumstances, no authorisations have been issued by Mexico. Over 11,000 specimens reported as captive-bred were authorised for export by Mexican authorities from 2013–2019, although it is thought that a large proportion of these were likely to have been wild-collected. Most international export is destined for Asia (particularly mainland China and Hong Kong SAR) where it is found in trade for human food, the pet trade, and the traditional medicine market. There has been relatively limited trade with the USA (just over 1,000 imported from Mexico, more than 1,500 wild-sourced specimens in transit originating in Mexico and 3,400 exported (nearly 850 originating in Mexico) between 2008 and 2020, half of which were reported as ranched (the USA is not a range State).

The species commands relatively high prices in the international pet trade: online offers for sale range from EUR600–900 (equivalent to USD600–900 as of August 2022) in Europe and from USD200–695 for hatchlings or juveniles and USD600–2,500 for large adults in the USA, suggesting that demand for the species is mainly from specialist collectors. Most of the specimens offered in Asia appear to be individuals taken from the wild and therefore traded illegally. In 2020 just over 4,000 specimens of destined for mainland China were seized in Mexico in a larger consignment containing other turtle species as well, which may indicate a substantial parallel undeclared trade in turtles generally.

Analysis: *Claudius angustatus* is a relatively widespread species with most of its range in Mexico. There are no global population estimates but there are indications of low densities and declining numbers in at least parts of its Mexican range. Little is known of its status in Belize or Guatemala. In Mexico, the extent of suitable habitat is reported to be decreasing through degradation, although the precise impact on the species is unknown. There is demand in both domestic and international markets as food and in the pet trade. The high prices posted for juveniles and adults suggest a specialist market as pets. Indications, notably from seizures, are that a high proportion of the trade is

in (illegally) wild-collected specimens and there are anecdotal accounts of decreasing populations in areas subject to collection. Harvest is considered a driving threat, however the lack of data on current populations and on the extent of wild offtake for international trade (not authorised in Mexico) makes it difficult to assess the actual impact of harvest. Although the case for listing is not clear-cut, on balance the indications of impact of harvest and growing demand of chelonians from the region supports inclusion in Appendix II on the basis of Criterion B of Annex 2A 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

Claudius angustatus (Cope, 1865).

Synonyms: *Claudius megaloccephalus* (Bocourt, 1868); *Claudius macrocephalus* (Gray, 1873); *Claudius agassizii* (Smith and Taylor, 1950).

Range

Mexico, Guatemala, and Belize.

IUCN Global Category

Lower Risk/Near Threatened (assessed 1996, ver. 2.3).

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

Claudius angustatus inhabits the lowlands of the Gulf of Mexico and Caribbean slopes (altitudes range of 0–300 m above sea level) from central Veracruz, Tabasco, to northern Oaxaca, northern and eastern Chiapas, western Campeche, and north-western Guatemala and Belize (Vásquez-Cruz and Arleth Reynoso-Martínez, 2020). However, according to Cervantes-Lopez et al. (2021), the last verified records of *C. angustatus* specimens in Belize and Guatemala date back to the mid-1990s. One specimen of *C. angustatus* was recorded at an elevation of 458 m by Vásquez-Cruz and Arleth Reynoso-Martínez (2020). A study by Cervantes-Lopez et al. (2021) confirmed the presence of *C. angustatus* in the municipality of Marqués de Comillas, an unprotected part of the Selva Lacandona—this represents the southernmost record of the species and increased its geographic range by 12 km in a straight line. No empirical population estimates for the species are currently available but records of localised population declines from certain areas (e.g., within the regions of Coatzacoalcos and Minatitlán in Veracruz, Villahermosa in Tabasco, and at the Hondo River) have been reported. The density levels of certain local populations in the southeast of the country have been registered as relatively low, ranging from 2–16 ind./ha in 2012–2014.

The turtles deposit and bury their eggs on land, laying two to three clutches, each containing between one and six eggs (average clutch size: 2.5 eggs). *Claudius angustatus* has an oval-shaped carapace with three, relatively weak, longitudinal keels. The average straight carapace length (CL) is 116 mm in males, and 106 mm in females (Legler and Vogt, 2013); the average weight is 600 g and 350 g for males and females respectively. Reynoso et al. (2016) reported individuals throughout the area of distribution with a straight carapace length between 77–153 mm (average 116 mm) in males, and between 75 and 130 mm (average 105 mm) in females; males weighed between 79 and 490 g (average weight 259 g); and females between 80 and 430 g (average weight 195 g).

Claudius angustatus is apparently among the most traded Mexican turtle species, along with *Terrapene yucatana*, *T. mexicana*, *S. triporcatus*, and *S. salvinii*. The species is in high demand in international trade and is found in the pet, food and traditional medicine markets. The species is mostly exported to Asian countries, with high numbers reaching mainland China and Hong Kong SAR. *It is not clear what percentage of the individuals exported to mainland China and Hong Kong SAR are for traditional medicine or food, but the vast majority of trade in C. angustatus appears to be destined for these two markets rather than the pet market* (R. Macip Ríos, in litt., 2022). Only live specimens are internationally traded.

At the national level, the Dirección General de Vida Silvestre (DGVS-SEMARNAT) reports five UMAs (Wildlife Conservation Management Units) or PIMVS (Predios o Instalaciones que Manejan Vida Silvestre) mentioning the harvest and use (mainly as breeding stock or for the pet trade) of individuals of *C. angustatus*. Most of the authorisations given in recent years have been for the export of live individuals. From 2013–2019, DGVS-SEMARNAT authorised the collection of 11,218 *Claudius angustatus* individuals from captive breeding stock, with the highest number authorised in 2018. At present, no authorisations for the harvest of wild specimens have been issued by DGVS-SEMARNAT, although it is likely that wild specimens are illegally harvested and later traded

as captive-bred. All the authorisations for trade of captive-bred specimens from 2015–2019 had a commercial purpose, hence exclusively involving live individuals for export. A considerable increase in the number of specimens exported has been registered before and after 2015 (i.e., from hundreds to thousands of individuals).

Demand appears to have increased in recent years at the international level. In the period 2005–2019, 11,846 live captive-bred *C. angustatus* individuals were authorised for export from Mexico, and the Procuraduría Federal de Protección al Ambiente (PROFEPA), which is responsible for verifying permits and the correct documentation for trade and export of the species, has comparable results for the period 2009–2020, with 10,655 individuals. Overall, records from PROFEPA show the primary importers from 2013–2020 were mainland China (62% of the total), Hong Kong SAR (21%), and the USA (8%), with further instances of imported specimens occurring in Japan, Republic of Korea, Italy, and Spain. Among them, 9,412 specimens were from their native Mexico, while 1,243 were re-exports that originated in the USA.

The Supporting Statement mentions that the USA reported imports of 2,540 *C. angustatus* specimens originating from Mexico for commercial purposes from 2015–2019, and 1,117 of them were taken from the wild. Analysis of the LEMIS database reveals a total of 1,035 live individuals imported from 2008–2018, with 1,018 of them originating from Mexico (out of which 878 were captive-bred and 140 were captive-born). Additionally, US data indicate more than 1,500 wild-sourced specimens in transit to Hong Kong SAR and originating in Mexico, which had not authorised trade in wild specimens (40% of these were seized and the rest were cleared). The same data show a total of 3,411 exported live commercial specimens from 2008–2020, of which 2,564 originated from the USA while more than half the live specimens (1,751) labelled as ranched and originating in the USA (which seems highly improbable given that it is not a range State), followed by almost 29% (982) captive-born, and nearly 20% (679) captive-bred. Of these exports, 23% of the specimens originated from Mexico, with the remaining 77% from the USA. During the same period, almost 54% of live individuals (1,832 specimens) were exported to mainland China and nearly 25% (842 specimens) to Hong Kong SAR. LEMIS cleared data show a considerable increase in the annual number of exported live specimens from 2016–2018 (1,460% increase in the number of specimens in 2016 compared to 2018), followed by a rapid decrease in 2019–2020 that may partially be related to the impacts of the COVID-19 pandemic (Figure 1). Instances of legal trade also occur in Europe (mostly in Germany, Italy, Spain, and Poland), where the species is mainly sold in the pet market and prices range from EUR250–500 (equivalent to USD250–500 as of August 2022) per adult specimen.

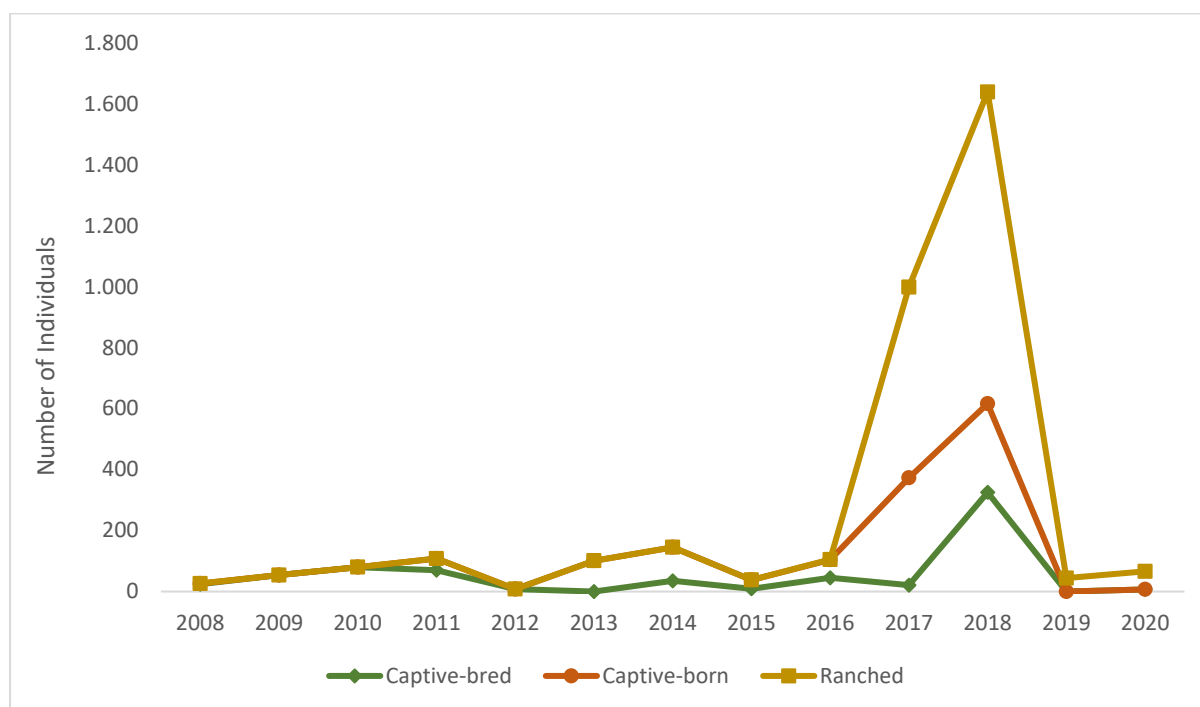


Figure 1. Number of live *C. angustatus* specimens exported by the USA for commercial purposes from 2008–2020. Specimens are grouped into the following: captive-bred, captive-born and ranched individuals. The highest number of annual exported specimens (1,638) was in 2018, of which 1,022 were reported as ranched. An overall increase of 1,460% in the number of live specimens exported was observed between 2016 and 2018. Source: LEMIS (2022).

According to the Supporting Statement, the vast majority of *C. angustatus* specimens in the Asian market appear to be individuals taken from the wild—and as DGVS-SEMARNAT reports no authorisations for the harvest of wild specimens in Mexico being issued, these were presumably traded illegally. According to PROFEPA, a total of

7,347 specimens were seized outside of Mexico from 2015–2021. According to WiTIS, a seizure of seven specimens took place at Mérida International Airport in 2018 and 15,053 turtles (including 4,216 *C. angustatus*) destined for mainland China were seized in the cargo area of Mexico City International Airport in 2020. The online trade of *C. angustatus* is substantial, with most of the sales occurring through social networks. Most of the online sales advertisements do not mention whether permits have been acquired or whether the turtles come from captivity. In the USA, advertised prices range from USD200–695 for a hatchling or juvenile individual to USD600–2,459 for a large adult. In Europe, *C. angustatus* individuals are illegally sold online from EUR600–900 (USD 600–900). A study by Sung et al. (2021) into the illegal turtle trade on Hong Kong-based online platforms and physical markets during 2017–2018 found 12 *C. angustatus* specimens offered for sale on social media, nine on internet fora, and 35 at the “Goldfish Market”, the largest pet market in Hong Kong SAR.

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 number of harvesting and export permits granted have increased substantially, mirroring the increase in demand for the species and larger volumes of illegal trade. The documented increase in harvesting levels to appease trade demand has the potential to cause the decline of wild *C. angustatus* populations in the future. As with numerous other turtle species, wild specimens are harvested and subsequently reported as captive-bred. Additionally, harvesting of large numbers of individuals can affect the occurrence and reproduction of local populations, thus leading to further population declines.

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

Claudius angustatus is not considered to be similar to any listed species.

Although the morphological characteristics of *C. angustatus* make it easy to identify and distinguish the species from others, adult specimens are sometimes confused with juveniles of the genus *Staurotypus*. However, *C. angustatus* can be differentiated in having (i) less pronounced carapace keels, (ii) non-jagged posterior marginal scutes and (iii) potentially roughened carapace scutes from growth rings. Moreover, young specimens may be identified by dark radial patterns.

Additional Information

Threats

The primary threat to wild populations of *C. angustatus* is illegal harvest for consumption as food and sale as live animals to private collectors/breeders. In Mexico, the species has been consumed for its meat by local communities since pre-Columbian times, and its sale generates an important revenue for local fishermen (Macip Ríos, in litt., 2022). Collection of specimens mostly occurs at the beginning and end of the rainy season (June and November respectively), which has an impact on population demographics since the spawning period comes to an end at the end of the dry season (generally May).

Additional major threats include habitat loss due to changes in land-use for agriculture, industrial farming and urban development, resulting in less suitable habitat for the settlement and reproduction of the species. In Mexico, an area of approximately 52,671 km², equivalent to 71% of the potential range of *C. angustatus* in the country, has been altered by agricultural activities and, to a lesser extent, by the growth of urban areas. Elevated risks particularly occur from agricultural operations—slash-and-burn practices kill turtles that are buried and dormant in estivation, tractors used in fields can mutilate the animals and the use of nitrogen fertilisers often poisons turtles. Populations of *C. angustatus* are also threatened by road construction, roadkill, water pollution as well as the use of pesticides and petrochemicals.

Conservation, management and legislation

In Mexico, part of the distribution range of *Claudius angustatus* is predicted to fall within seven Natural Protected Areas (NPAs) within the national territory, namely Sistema Lagunar de Alvarado, Reserva de la Biósfera de Los Tuxtlas, Humedales de la Laguna Popotera, Zona Arroyo Moreno, Pantanos de Centla, Sistema Lagunar Catatzajá, and Laguna de Términos. The inferred distribution area of the species within these seven NPAs may reach 2,045 km² (i.e., 2.7% of the total observed distribution of the species in Mexico). These reserves are inhabited by local people who are allowed to harvest from lagoons and lakes and to use water resources including aquatic animals, such as *C. angustatus* (Macip Ríos, in litt., 2022). Wild populations are currently not being monitored. This species, alongside other Mexican wildlife, is protected and regulated at the national level by the General Law of Ecological Balance and Environmental Protection (LGEEPA, SEDUE 1988), the General Wildlife Law (LGVS, SEMARNAT 2000) as well as its respective regulations (SEMARNAT 2006) and NOM-059-SEMARNAT-2010 List of species at risk—where *C. angustatus* is currently listed as endangered.

Due to its inclusion in the endangered species category within the Mexican government’s List of Species at Risk, *C. angustatus* can only be harvested from the wild under very particular circumstances (Articles 84 and 85 of the

LGVS). These two Articles allow harvesting of species at risk when it is a priority that individuals are collected for restoration, repopulation, or reintroduction operations or scientific research. If harvest of wild specimens is intended for any other use (e.g., commercial purposes), authorisation can only be given when at least one of the following circumstances is complied with: (i) the requested harvest rate is smaller than the natural repopulation rate of wild specimens, (ii) the requested harvest does not negatively affect the populations and will not change the life cycle of the specimen, and (iii) there is no manipulation that will permanently damage the specimens. Moreover, harvesting of specimens at risk must be coupled with actions that promote the development of the species' populations through programmes or projects such as captive breeding. Management plans that list and describe the actions for the species must be submitted and approved by relevant authorities. According to information from the DGVS-SEMARNAT, no authorisations have been issued for harvesting of wild specimens.

Captive breeding

The diet of *C. angustatus* is mainly freshwater shrimps of the genus *Procambarus*, which are not currently harvested in large numbers. This makes it difficult to establish captive populations of the species. However, several Wildlife Conservation Management Units (UMAs) are successfully carrying out captive breeding operations within the country, together with certain hobbyists in Europe.

Two specimens are currently housed in the herpetological collections of the University of Kansas Biodiversity Institute from Potrero Viejo in the municipality of Amatlán de los Reyes, Veracruz (Vásquez-Cruz and Arleth Reynoso-Martínez, 2020).

Implementation challenges (including similar species)

Claudius angustatus can be easily distinguished from other species due to its morphological characteristics.

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Inclusion of the genus *Kinosternon* in Appendix II and *K. cora* and *K. vogti* in Appendix I

Proponents: Brazil, Colombia, Costa Rica, El Salvador, Mexico, Panama, United States of America

Summary: *Kinosternon* is a genus of 22 species of small (10–20 cm carapace length) semi-aquatic turtles, known as mud turtles, endemic to the Americas. The highest diversity (17 species) is in Mexico; three species occur in South America. None are currently listed in the Appendices. Two recently described Mexican species (*K. vogti* and *K. cora*) are proposed for inclusion in Appendix I, and the remainder for inclusion in Appendix II.

- ***Kinosternon vogti*** has a very restricted distribution in Mexico with an area of occupancy of less than 100 km². The remaining habitat has been highly fragmented by housing development. The wild population size was estimated to total around 1,000 individuals and is believed to be decreasing. Since its description in 2018, three systematic surveys have been carried out resulting in only 56 live observations. The species was assessed as Critically Endangered on the IUCN Red List in 2022 owing to habitat loss and degradation. A decline in area of occupancy, as well as exploitation for the international live animal trade and competition from sympatric native and invasive turtle species was estimated to have reduced the population by more than 80%. The only reported use of this species was as live specimens for collectors, and there is an apparent growing demand in the international pet trade. It is legally protected in Mexico.
- ***Kinosternon cora*** is known from observations of only six individuals from an area of approximately 500 km² and is presumed to be rare. Waterbodies in its range are known to have been degraded and modified for agricultural purposes. It has not so far been assessed for the IUCN Red List. The species has been reported for sale in Asian markets.

Sixteen of the other 20 *Kinosternon* species have been assessed for the IUCN Red List: three as Vulnerable (*K. abaxillare*, *K. angustipons*, *K. dunni*) three as Near Threatened (*K. acutum*, *K. herrerae*, and *K. sonoriense*), seven as Least Concern (*K. baurii*, *K. chimalhuaca*, *K. creaseri*, *K. flavescens*, *K. hirtipes*, *K. integrum* and *K. subrubrum*) and three as Data Deficient (*K. alamosae*, *K. durangoense*, and *K. oaxacae*). The remaining species (*K. scorpioides*—the most widespread member of the genus, *K. leucostomum*, *K. steindachneri* and *K. steinegeri*) have not been assessed.

Information on total population status and trends for most species of the genus is incomplete or lacking. In Mexico it is reported that most of the waterbodies they inhabit are isolated and, even for species with large areas of distribution, suitable habitats may be occupied by no more than 500 individuals. This fragmentation of populations is believed to increase their vulnerability to overcollection and there are anecdotal observations of this occurring.

Kinosternon species are harvested for both human food consumption and the pet trade. There are no global trade data for any of the species. Data from the USA mostly reflects trade in its native species, of which there are seven. Live *Kinosternon* individuals reported as wild-caught, ranched or from unknown sources have been exported from the USA in relatively large numbers between 2008 and 2020, e.g., (around 47,000 *K. baurii*, 10,000 *K. flavescens*, 38,000 *K. scorpioides* (not a native US species) and 145,000 *K. subrubrum*). The largest numbers reported in trade were the Eastern Mud Turtle *K. subrubrum*, a widespread species endemic to the USA where it is not considered threatened. During the same period just under 1,400 *K. acutum* (not a native US species) were exported from the USA, reported as either wild-caught or ranched. The main destination for US exports was East Asia, including China, Hong Kong SAR, Macao SAR, and Japan. Data show over 7,440 live wild or ranched *K. scorpioides* and just over 43,000 declared as from captive sources imported by the USA from El Salvador.

The main threats to the species in this genus have been identified as habitat loss and degradation,

local human consumption for food (particularly in riverside communities), the introduction of invasive species, and collection for trade as pets for national and international markets. There has been no authorised collection of either *K. vogti* or *K. cora* and no reported legal exports. However, both species have been reported in Asian markets, where they appear to command high prices (reportedly up to USD10,000 in the case of *K. vogti*).

In Mexico, limited wild harvest of four *Kinosternon* spp. (*K. integrum*, *K. acutum*, *K. leucostomum* and *K. scorpioides*) has been authorised with just under 700 licensed in total from 2010 to 2022. Captive breeding of some species occurs in El Salvador, Brazil, Mexico, and the USA, and is well established. In Mexico, just over 30,000 specimens were reported as captive bred from 2010–2020, primarily (>90%) *K. leucostomum* with low quantities of *K. integrum*, *K. abaxillare*, *K. scorpioides* and *K. hirtipes*. During this same period, nearly 33,000 *Kinosternon* spp. were exported from Mexico, largely (94%) comprising *K. leucostomum*. However, there is also evidence of illegal trade: nearly 20,000 (mostly *K. leucostomum*) have been seized in recent years, the majority in 2020 and most showing evidence of capture from the wild. There is no authorisation in Mexico for the collection of wildlife for export purposes, so an illicit origin is presumed. For illegal trade, larger adults with striking colouration are targeted, which may adversely affect population dynamics.

Customs officials are unlikely to be able to tell the difference between species of *Kinosternon*, particularly juveniles and hatchlings, without specialised training. Identification practices are often based on colouration, which is unreliable.

Analysis: *Kinosternon cora* and *K. vogti* both appear to have a restricted area of distribution in which habitat quality is declining as a result of development and agricultural activities. Wild population sizes of both species are unknown, but likely to be small or very small; both are believed to have declined or declined markedly (*K. vogti*), with these declines inferred to be continuing. Despite legal protection in Mexico, both species have been reported in pet markets in Asia indicating international demand. *K. vogti* and *K. cora* both appear to meet the criteria for inclusion in Appendix I in Res. Conf. 9.24 (Rev. CoP17).

With regards to other *Kinosternon* spp., in general information on levels of collection for trade, the impact of collection and the status of wild populations more generally, is lacking. Several species are known to be harvested for food and for the international pet trade, with data from the USA (the only country for which extensive trade data exist) indicating that wild-sourced or ranched animals of some species of the genus have been exported in notable numbers. Seizure data indicate that some of those in trade, notably *K. leucostomum* exports from Mexico declared as captive-bred may in fact be wild-collected. However, overall, there is insufficient information to determine whether or not any of the *Kinosternon* species proposed for Appendix II listing meets the criteria set out in Annex 2a of Res. Conf. 9.24 (Rev. CoP17).

Specimens of *Kinosternon cora* and *K. vogti* are reported to be similar in appearance to *K. angustipons*, *K. dunni*, *K. herrerae* and *K. leucostomum*. Juveniles and hatchlings of all species are said to be difficult to distinguish from each other. If *K. cora* and *K. vogti* are accepted for inclusion in Appendix I, it would appear that all other members of the genus meet the criteria in Annex 2 b of Res. Conf. 9.24 (Rev CoP17) for inclusion in Appendix II on the basis of being lookalikes.

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 nomenclature of species to be included follows, as far as possible, the standard reference adopted for turtles in the Annex of Resolution Conf. 12.11 (Rev. CoP18), Fritz and Havas (2007). However, due to recent taxonomic changes for *Kinosternon* and the description of new species in 2018, the taxonomy herein is based on the updated Turtles of the World Checklist (TTWG; Rhodin *et al.*, 2021).

- a) Species proposed for inclusion in Appendix I: *K. cora* (Loc-Barragan *et al.*, 2020) and *K. vogti* (Lopez-Luna *et al.*, 2018, 2021)

- b) Species proposed for inclusion in Appendix II: *Kinosternon abaxillare* (Baur in Stejneger 1925), *K. acutum* (Gray 1831), *K. alamosae* (Berry and Legler 1980), *K. angustipons* (Legler 1965), *K. baurii* (Garman 1891), *K. chimalhuaca* (Berry, Seidel & Iverson 1997), *K. creaseri* (Hartweg 1934), *K. dunni* (Schmidt 1947), *K. durangoense* (Iverson 1979), *K. flavescens* (Agassiz 1857), *K. herrerae* (Stejneger 1925), *K. hirtipes* (Wagler 1830), *K. integrum* (Le Conte 1854), *K. leucostomum* (Duméril & Bibron in Duméril & Duméril 1851), *K. oaxacae* (Berry & Iverson 1980), *K. scorioides* (Linnaeus 1766), *K. sonoriense* (Le Conte 1854), *K. steindachneri* (Siebenrock 1906), *K. stejnegeri* (Hartweg 1938), and *K. subrubrum* (Bonnaterre 1789).

IUCN Global Category and Range

| Species | IUCN Global Category | Range |
|---------------------------------|--|--|
| Inclusion in Appendix I | | |
| <i>K. cora</i> | Not assessed | Mexico |
| <i>K. vogti</i> | Critically Endangered A4cde (assessed 2022, ver. 3.1) | Mexico |
| Inclusion in Appendix II | | |
| <i>K. abaxillare</i> | Vulnerable A2cd+4cd. (assessed 2020, ver. 3.1) | Mexico, Guatemala |
| <i>K. acutum</i> | Near Threatened (assessed 1996, ver. 2.3) | Mexico, Guatemala, Belize |
| <i>K. alamosae</i> | Data Deficient (assessed 2007, ver. 3.1) | Mexico |
| <i>K. angustipons</i> | Vulnerable B1+2c. (assessed 1996, ver. 2.3) | Nicaragua, Costa Rica, Panama |
| <i>K. baurii</i> | Least Concern (assessed 2010, ver. 3.1) | USA |
| <i>K. chimalhuaca</i> | Least Concern (assessed 2007, ver. 3.1) | Mexico |
| <i>K. creaseri</i> | Least Concern (assessed 2007, ver. 3.1) | Mexico, Belize |
| <i>K. dunni</i> | Vulnerable B1+2c. (assessed 1996, ver. 2.3) | Colombia |
| <i>K. durangoense</i> | Data Deficient (assessed 2007, ver. 3.1) | Mexico |
| <i>K. flavescens</i> | Least Concern (assessed 2010, ver. 3.1) | USA, Mexico |
| <i>K. herrerae</i> | Near Threatened (assessed 2007, ver. 3.1) | Mexico |
| <i>K. hirtipes</i> | Least Concern (assessed 2007, ver. 3.1) | USA, Mexico |
| <i>K. integrum</i> | Least Concern (assessed 2007, ver. 3.1) | Mexico |
| <i>K. leucostomum</i> | Not assessed | Mexico, Guatemala, Belize, Nicaragua, Costa Rica, Panama, Colombia, Ecuador, Honduras, Peru |
| <i>K. oaxacae</i> | Data Deficient (assessed 2007, ver. 3.1) | Mexico |
| <i>K. scorioides</i> | Not assessed | Mexico, Guatemala, Belize, Nicaragua, Costa Rica, Panama, Colombia, Ecuador, Honduras, Peru, Argentina, Bolivia, Brazil, El Salvador, Guyana, French Guiana, Paraguay, Bolivarian Republic of Venezuela, Suriname, Trinidad and Tobago |
| <i>K. sonoriense</i> | Near Threatened (assessed 2011, ver. 3.1) (<i>K. s. longifemorale</i> Critically Endangered A3c (assessed 2016, ver. 3.1)) | USA, Mexico |
| <i>K. steindachneri</i> | Not assessed | USA |
| <i>K. stejnegeri</i> | Not assessed | USA, Mexico |
| <i>K. subrubrum</i> | Least Concern (assessed 2011, ver. 3.1) | USA |

Assessment of *K. cora* and *K. vogti* **Biological criteria for inclusion in Appendix I**

A) Small wild population

The conservation status of *K. cora* cannot currently be assessed, because only six specimens are known from an area of approximately 500 km² (Loc-Barragán et al., 2020).

Kinosternon vogti was reported to have a global population of 1,000 individuals, and its population is severely fragmented and decreasing. Three systematic surveys since its description in 2018 recorded a total of only 56 live individuals.

B) Restricted area of distribution

The Pacific coastal lowlands between and beyond the known ranges of *K. cora* and *K. vogti* have not been thoroughly sampled for freshwater turtles.

Kinosternon cora:

In the Pacific coastal lowlands, human exploitation of freshwater turtles in the area is minimal, except for sporadic illegal collection for pets. *K. cora* could come under collection pressure from wildlife traffickers. Most bodies of water within the known range of *K. cora* have been modified to some degree for agricultural purposes. Unfortunately, the range of *K. cora* suffers from social and drug-related conflicts and it is not possible safely to monitor populations at this time. The six known specimens are from an area of only approximately 500 km².

Streams and small ponds have been dredged for use as water reservoirs. Large volumes of water are used for irrigation of crops in the area (e.g., mango, sorghum, and soy). Local farmers and ranchers use chemicals to control weeds which pollutes ponds and waterways, causing some intoxication or death of cattle (Loc-Barragan, personal observation). This extensive agricultural use results in rapid reductions in water as the dry season begins. Furthermore, during the dry season when mud turtles may be in estivation, it is common for ranchers to burn the edges of ponds to make them more accessible for cattle when the rains return. The impact of these activities on the freshwater turtles in the area is unknown, although it is possible that these artificial water basins can help turtles.

Kinosternon vogti

The estimated historic range of this species was 263 km² however, its area of occupancy was noted to be less than 100 km². It has been found from sea level to 15 m above sea level and recorded in five locations within the Ameca River valley. The species inhabits small streams and ponds in and near the city of Puerto Vallarta, Jalisco (Rhodin et al., 2021). A very rapid decrease in populations is estimated owing to the rapid disappearance of suitable habitat due to human development. The drying of waterbodies in the few places where it has been recorded is also reducing its range (unpublished data Taggart Butterfield). López-Luna et al. (2018) confirmed that the habitat is being negatively impacted by urban growth. The species is also found as road kills in Puerto Vallarta (Iverson, in litt., 2022).

The Ameca River valley has a dense human population that is growing rapidly, and there are many major hotel developments in progress to attract more tourism. Of the 22 appropriate habitat localities in the Ameca River valley that have been sampled for *K. vogti*, the species has only been confirmed in five, and 43% of the turtles found were in a single locality that is not protected. Two of the five known locations are being actively developed, three localities have high potential to be developed, and only one locality, a small creek inside the Centro Universitario de la Costa, is relatively protected in an area that is unlikely to be developed (Cupul-Magaña et al., 2022).

C) Decline in number of wild individuals

Assessed by IUCN as Critically Endangered indicating an estimated ≥80% population reduction based on a decline in the area of occupancy due to habitat loss, potential exploitation for the international pet trade and competition from sympatric native and invasive turtle species (Cupul-Magaña et al., 2022).

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

No export permits have been issued, but both *K. cora* and *K. vogti* have been reported in Asian markets, presumably as a result of illegal trade.

Both *K. cora* and *K. vogti* have appeared in international markets. Demand for *K. vogti* increased nationally and internationally after the species was described in 2018, and in 2021 *K. vogti* was found on internet fora for private collectors in Hong Kong SAR. An individual *K. vogti* was priced at USD10,000 in China according to information from social networks. Mexico's CITES Scientific Authority has also found evidence on social networks indicating that individuals were in possession of the species, presumably illegally given the lack of authorisation for collection. Several *K. vogti* specimens have been observed in Asian markets, and *K. cora* specimens were

observed there in 2022. The Mexican Government (DGVN-SEMARNAT) has not reported any authorised take of either species from the wild, with PROFEPA also reporting no exports of these species.

Assessment of the other 20 *Kinosternon* species

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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

Information on total population status and trends for most species of the genus *Kinosternon* is incomplete. In Mexico the waterbodies they inhabit are isolated (even more so in the dry season, lasting as long as 7–8 months) and even for species with large areas of distribution, each localised area of suitable habitat may be occupied by no more than 500 individuals and for some species as few as 100. This has been shown for the following species; *K. angustipons* (Iverson, 1980; Rhodin et al., 1974), *K. baurii* (Mushinsky and Wilson, 1992), *K. creaseri* (Macip-Rios et al., 2018), *K. dunni* (Iverson et al., 2012), *K. herrerae* (Iverson, 1982), *K. hirtipes* (Iverson 1982), *K. sonoriense* (Hulse, 1982), *K. stegneri* (Iverson, 1989), and *K. subrubrum* (Mahmoud, 1969) (Table 1). This fragmentation of populations is believed to increase vulnerability to decline through overcollection (L. Guillermo, in litt., 2022). It is unclear how many of these waterbodies are present throughout the species' ranges.

Table 1. Potential criteria on which to assess whether each species of *Kinosternon* meets Criteria 2a for inclusion in Appendix II.

| Species | IUCN global category | Presumed historic range (km ²) (TTWG, 2021) | Abundance and Population estimates | Use and Trade | Indication of trade in LEMIS records (see table 2) |
|-------------------------------|---|---|---|--|--|
| Include in Appendix I | | | | | |
| <i>K. cora</i> | Not assessed | 7,200 | Total population size unknown, but localised sub-population size <500 (Loc-Barragán et al., 2020) Rare/ extremely rare (Loc-Barragán et al., 2020) | No export permits issued, but have been reported in markets in Asia; Presumed to be part of illegal international pet trade | |
| <i>K. vogti</i> | Critically Endangered A4cde (assessed 2022, ver. 3.1) | 700 | Rare/extremely rare (Lopez-Luna et al., 2018) | | |
| Include in Appendix II | | | | | |
| <i>K. abaxillare</i> | Vulnerable A2cd+4cd. (assessed 2020, ver. 3.1) | 21,458 | Micro-endemic, little studied (Reyes-Grajales et al., 2021) Local wild populations have disappeared (López-León, pers. comm.) | Common in pet and domestic food trade (Reyes-Grajales and Guichard-Romero, 2021) | |
| <i>K. acutum*</i> | Near Threatened (assessed 1996, ver. 2.3) | 21,458 | Rare/extremely rare (Loc-Barragán et al., 2020) | Eaten locally (IUCN TFTSG, 2011) | x |
| <i>K. alamosae</i> | Data Deficient (assessed 2007, ver. 3.1) TFTSG Provisional RL: LC (2018) (Rhodin et al., 2021) | 51,888 | Rare/extremely rare (Loc-Barragán et al., 2020) | Presumed to be part of illegal international pet trade | |
| <i>K. angustipons*</i> | Vulnerable B1+2c. (assessed 1996, ver. 2.3) | 24,179 | Total population size unknown, but localised sub-population size <500 (Iverson, 1980; Rhodin et al., 1974) | | x |
| <i>K. baurii</i> | Least Concern (assessed 2010, ver. 3.1) | 355,427 | Total population size unknown, but localised sub-population size <500 (Mushinsky and Wilson, 1992) | Climate change models suggest a decline in suitable habitat over next few decades (IUCN TFTSG, 2021) | x |
| <i>K. chimalhuaca</i> | Least Concern (assessed 2007, ver. 3.1) | 6,701 | | Presumed part of illegal international pet trade | x |
| <i>K. creaseri*</i> | Least Concern (assessed 2007, ver. 3.1) | 120,545 | Total population size unknown, but localised sub-population size <500 (Macip-Rios et al., 2018) | Presumed part of illegal international pet trade | x |

| Species | IUCN global category | Presumed historic range (km ²) (TTWG, 2021) | Abundance and Population estimates | Use and Trade | Indication of trade in LEMIS records (see table 2) |
|-------------------------|---|---|--|---|--|
| <i>K. dunni</i> | Vulnerable B1+2c. (assessed 1996, ver. 2.3) | 21,653 | Total population size unknown, but localised sub-population size <500 (Iverson <i>et al.</i> , 2012) Because of its restricted range and apparent rarity, and because it is eaten locally, it could be more threatened than currently recognised (IUCN TFTSG, 2012) | Presumed to be part of the illegal international pet trade Eaten locally | |
| <i>K. durangoense</i> | Data Deficient (assessed 2007, ver. 3.1) TFTSG Provisional RL: LC (2018) (Rhodin <i>et al.</i> , 2021) | 27,807 | Rare/ extremely rare (Lopez-Luna <i>et al.</i> , 2018) | | |
| <i>K. flavescens</i> | Least Concern (assessed 2010, ver. 3.1) | 1,349,753 | | Presumed to be part of illegal international pet trade | x |
| <i>K. herrerae*</i> | Near Threatened (assessed 2007, ver. 3.1) | 70,887 | Total population size unknown, but localised sub-population size <500 (Iverson, 1982) | Consumed domestically for food and folk medicine, as well as souvenirs (van Dijk <i>et al.</i> , 2007) | x |
| <i>K. hirtipes</i> | Least Concern (assessed 2007, ver. 3.1) | 113,016 | Total population size unknown, but localised sub-population size <500 (Iverson, 1982) | Presumed to be part of illegal international pet trade | x |
| <i>K. integrum</i> | Least Concern (assessed 2007, ver. 3.1) | 309,188 | | Presumed part of illegal international pet trade | x |
| <i>K. leucostomum</i> | Not assessed TFTSG Provisional RL: LC (2011) (Rhodin <i>et al.</i> , 2021) | 67,999 | Local wild populations have disappeared (Guichard-Romero, pers.comm.) | Presumed part of illegal international pet trade Food consumption (Iverson, 2010; Pereira <i>et al.</i> , 2003) | x |
| <i>K. oaxacae</i> | Data Deficient (assessed 2007, ver. 3.1) TFTSG Provisional RL: LC (2018) (Rhodin <i>et al.</i> , 2021) | 21,275 | Local wild populations have disappeared (Reyes-Grajales. pers. comm.) | Presumed to be part of illegal international pet trade | x |
| <i>K. scorpioides</i> | Not assessed TFTSG Provisional RL: LC (2011) (Rhodin <i>et al.</i> , 2021) | 7,782,398 | Local wild populations have disappeared (Vogt pers.comm.) | Presumed part of illegal international pet trade Food consumption Extracted at high levels in Mexico and Guatemala as food, particularly during religious festivals, e.g. Lent (leading to local populations disappearing; Legler and Vogt, 2013) | x |
| <i>K. sonoriense</i> | Near Threatened (assessed 2011, ver. 3.1) | 168,307 | Total population size unknown, but localised sub-population size <500 (Hulse, 1982) | Presumed to be part of illegal international pet trade | x |
| <i>K. steindachneri</i> | Not assessed | 93,472 | Rare/ extremely rare (Lopez-Luna <i>et al.</i> , 2018) | Presumed to be part of illegal international pet trade | x |

| Species | IUCN global category | Presumed historic range (km ²) (TTWG, 2021) | Abundance and Population estimates | Use and Trade | Indication of trade in LEMIS records (see table 2) |
|----------------------|---|---|---|--|--|
| <i>K. stejnegeri</i> | Not assessed | 28,423 | Total population size unknown, but localised sub-population size <500 (Iverson, 1989) | Presumed to be part of illegal international pet trade | x |
| <i>K. subrubrum</i> | Least Concern (assessed 2011, ver. 3.1) | 1,401,264 | Total population size unknown, but localised sub-population size <500 (Mahmoud, 1969) Well documented population declines in subspecies <i>K. s. subrubrum</i> in Northeast and Midwest (IUCN TFTSG, 2017) | Presumed part of illegal international pet trade (small numbers) | x |

Kinosternon abaxillare is classified as *Vulnerable and decreasing* on the IUCN Red List (assessed 2020). There is reportedly a high capture rate for low capture effort and some local wild populations are reported to have disappeared. They are considered micro-endemic and understudied (Grajales et al., 2020). Although there is no legal trade data for this species, any collection of wild individuals could be detrimental.

Kinosternon hirtipes is classified by IUCN as Least Concern although the overall population size is believed to be decreasing, and the species is involved in legal and illegal trade.

Regarding legal trade, the DGVS-SEMARAT reported that from 2010–2022, 688 *Kinosternon* were authorised to be taken from the wild in Mexico (468 *K. integrum*, 85 *K. acutum*, 68 *K. leucostomum* and 67 *K. scorpioides*). During the same period, the commercial export of 32,883 specimens was authorised (30,843 *K. leucostomum*, 445 *K. integrum*, 170 *K. acutum* and 1,425 *Kinosternon* spp.), all supposedly from captive breeding stock. According to the USFWS, between 2000 and 2019, 1,393 specimens were imported from Mexico into the USA.

LEMIS export data indicate that some species of *Kinosternon* are heavily traded. *Kinosternon* species exported from the USA from 2008 to 2020 comprised a total of 373,038 individuals, almost all for commercial purposes (65% of individuals were wild-sourced (W) or ranched (R)) (Table 2).

The main destinations were mainland China (50%), Hong Kong SAR (30%), Macau SAR (7%), Republic of Korea (3%), Japan (3.5%), and Taiwan POC (2.5%). At a species level, from 2008–2020 the following were exported: 90,440 *K. baurii* (Least Concern) (47,168 from wild populations), 14,510 *K. flavescens* (Least Concern) (9,515 from wild populations), 5,223 *K. leucostomum* (not assessed) (2,351 from wild populations), 83,931 *K. scorpioides* (not assessed) (38,065 from wild populations), and 176,103 *K. subrubrum* (Least Concern) (144,386 from wild populations). It is noteworthy that the USA reported the export of large numbers of supposedly wild-caught *K. leucostomum* and *K. scorpioides* when it is not a range State for either species.

LEMIS import data were far higher than referenced in the proposal which notes 1,393 specimens from 2000–2019. They included large numbers of *K. scorpioides* and *K. leucostomum* and smaller numbers of other non-native species, e.g., *K. oxacae* (29 specimens) (Table 2). LEMIS data show imports of over 46,000 *K. scorpioides* (no subspecies reported) along with 5,600 *K. scorpioides cruentatum* from El Salvador between 2013 and 2020, contrasting with exports reported in the proposal of 46 specimens from 2013–2021.

Table 2. USA trade data for *Kinosternon* species 2008–2020. Only cleared records included (LEMIS, 2022).

| | Exports | | | | Imports | | | |
|---------------------------------|---------|--------|---|--------|---------|-----|----|-------|
| | C+F | R+W | U | Total | C+F | R+W | U | Total |
| Inclusion in Appendix I | | | | | | | | |
| <i>K. cora</i> | | | | | | | | |
| <i>K. vogti</i> | | | | | | | | |
| Inclusion in Appendix II | | | | | | | | |
| <i>Kinosternon</i> spp. | 948 | 359 | | 1,307 | 212 | 585 | 52 | 849 |
| <i>K. abaxillare</i> | | | | | | | | |
| <i>K. acutum</i> * | 51 | 1,393 | | 1,444 | 4 | | | 4 |
| <i>K. alamosae</i> | | | | | | | | |
| <i>K. angustipons</i> * | 11 | | | 11 | | | | |
| <i>K. baurii</i> | 43,272 | 47,168 | | 90,440 | 4 | | | 4 |
| <i>K. chimalhuaca</i> | | | | | 4 | | | 4 |
| <i>K. creaseri</i> * | 3 | | | 3 | 40 | | | 40 |
| <i>K. dunni</i> | | | | | | | | |
| <i>K. durangoense</i> | | | | | | | | |

| | Exports | | | | Imports | | | |
|--------------------------|---------|---------|---|---------|---------|--------|----|--------|
| | C+F | R+W | U | Total | C+F | R+W | U | Total |
| <i>K. flavescens</i> | 4,995 | 9,515 | | 14,510 | | | | |
| <i>K. herrerae</i> * | 17 | | | 17 | 3 | | | 3 |
| <i>K. hirtipes</i> | | | | | 5 | | | 5 |
| <i>K. integrum</i> * | | 3 | | 3 | 34 | | | 34 |
| <i>K. leucostomum</i> * | 2,872 | 2,351 | | 5,223 | 3,528 | 317 | | 3,845 |
| <i>K. oaxacae</i> * | 14 | | | 14 | 6 | | 23 | 29 |
| <i>K. scorpioides</i> ** | 45,866 | 38,065 | | 83,931 | 65,143 | 9,483 | 1 | 74,627 |
| <i>K. sonoriense</i> | 29 | 2 | | 31 | | 1 | | 1 |
| <i>K. steindachneri</i> | | 1 | | 1 | | | | |
| <i>K. stejnegeri</i> | | | | | | | | |
| <i>K. subrubrum</i> | 31,717 | 144,386 | | 176,103 | 10 | 512 | | 522 |
| Total | 129,795 | 243,243 | 3 | 373,038 | 68,993 | 10,898 | 76 | 79,967 |

* Not native to the USA

** Data by subspecies were merged and represented together as *K. scorpioides*.

In Argentina from 2000–2008, 306 *Kinosternon* were imported from the USA (*K. subrubrum*, *K. baurii*, *K. flavescens* and *K. leucostomum*), with no imports authorised after 2008.

Peru reported the export of 239 live individuals of *K. scorpioides* between 2019–2020 from captive breeding farms in the Loreto region.

El Salvador reported 46 exports of *K. scorpioides* in the period 2013–2021, with 76% of the specimens destined for the USA and the rest to Hong Kong SAR, Macao SAR, and Taiwan POC.

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

Regarding lookalikes, customs officials are unlikely to be able to differentiate between species of *Kinosternon* without training and the use of identification guides, dichotomous keys or digital identification apps. Individuals of the recently described *K. vogti* were misidentified by experts as *K. chimalhuaca* for over two decades. Identification practices are often based on colouration, which is extremely unreliable. Additionally, identification is particularly complicated for juvenile rather than adult turtles. *K. integrum* and *K. creaseri* (and possibly also *K. herrerae*, *K. acutum*, *K. oaxacae* and *K. chimalhuaca*) are recorded in trade and appear to meet criteria 2b as lookalikes with other *Kinosternon* species.

Additional information

Threats

The main threats to *Kinosternon* spp. have been identified as habitat loss (fragmentation, loss of vegetation cover, desiccation and/or contamination of waterbodies) and overexploitation for local consumption or the illegal international pet trade. Additional threats include invasive species, expanding road networks (turtles are crushed by cars), and droughts (a drought from 2019–2021 wiped out 48% of a population of *K. alamosae*).

A very rapid decrease in the *K. vogti* population is believed to have taken place following the rapid drying out of waterbodies in one of its few known locations. López-Luna et al. (2018) reported that the habitat is being negatively impacted by urban growth. Introduced populations of *K. integrum* and other non-native species are also considered a threat to *K. vogti* populations.

Most bodies of water within the known range of *K. cora* have been modified for agriculture, including dredging and creation of reservoirs, extraction for irrigation, with a rapid reduction of water during the dry season, burning at pond edges, and use of herbicides. The net impact of these activities is unknown, as it is possible that these artificial water basins can help turtles (Loc-Barragán et al., 2020).

In Brazil, a study reported 98% of people interviewed frequently consumed *K. scorpioides*, and another study in Belém, Pará, found that restaurants demanded on average 260 animals/month. Based on a total of 400 restaurants this equated to over a million animals/year. The supply of turtle species to this market was reported to be derived from wild populations as there is no authorised commercial breeding in Brazil.

Conservation, management and legislation

No *Kinosternon* species is included in the CITES Appendices.

Mexico does not permit the collection or export of *K. cora* or *K. vogti*. The Official Mexican Standard NOM-059-SEMARNAT-2010 (DOF, 2019) includes *K. vogti* in the category endangered. *Kinosternon scorpioides* is categorised as an insufficiently known taxon. In Argentina, and in Colombia it is categorised as vulnerable along with *K. dunni*. Guatemala categorises *K. acutum*, *K. leucostomum* and *K. scorpioides* as vulnerable. In Mexico, *K. sonoriense* and *K. vogti* are classified as endangered, and *K. acutum*, *K. alamosae*, *K. herrerae*, *K. hirtipes*, *K. integrum*, *K. leucostomum*, *K. oaxacae* and *K. scorpioides* are subject to special protection.

Information on harvest and trade controls is limited, primarily references from general wildlife regulations. Strict legal controls appear to be in place in Mexico, the range State for those species of greatest conservation concern. Harvest and trade are not regulated at the federal level in the USA, however, wildlife declaration forms are required for the import and export of these and other fish and wildlife species. *Kinosternon sonoriense longifemorale* is listed as Critically Endangered under the US Endangered Species Act (harvest and trade therefore banned). Controls on harvest and trade vary widely by US state (e.g., see <https://www.allturtles.com/turtle-laws/>).

Captive breeding

Mexico: there are 31 “Unidades de Manejo para la conservación de la Vida Silvestre” (UMAs) registered with the DGVS-SEMARNAT that function as hatcheries for conservation purposes for *Kinosternon*; in 21 of them *K. leucostomum* is raised, in six *K. integrum*, in three *K. acutum*, in three *K. scorpioides* and in only one the following species are raised: *K. abaxillare*, *K. creaseri*, *K. flavescens*, *K. herrerae*, and *K. hirtipes*. There are also 20 “Predios e Instalaciones que Manejan Vida Silvestre” (PIMVS)—breeding farms for commercial purposes—eight for *K. leucostomum*, seven for *K. integrum*, four for *K. scorpioides*, two for *K. acutum*, and only one for *K. alamosae*, *K. chimalhuaca*, *K. flavescens*, *K. hirtipes*. There is no UMA record for *K. vogti* and *K. cora*.

Argentina: According to the information provided, there are no formal hatcheries that reproduce *Kinosternon* for legal trade.

Peru: Management of *K. scorpioides* for export purposes stems from captive management.

USA: The USA reports the export of captive-born and captive-bred specimens.

El Salvador: Three authorised wildlife farms for the captive breeding of *K. scorpioides* are known.

Implementation challenges (including similar species)

The species *K. cora* and *K. vogti* are very similar, apart from male *K. vogti* that have a bright yellow nasal scale. It would be hard for enforcement officers to distinguish between them, and nearly impossible for females and juveniles. Additionally, *K. cora* and *K. vogti* are similar in appearance to *K. angustipons*, *K. dunni*, *K. herrerae* and *K. leucostomum* (the latter a common species present in both the legal and illegal trade).

Identification of these species has been based on general colour patterns (e.g., *K. integrum*, *K. scorpioides* and *K. leucostomum*), which is now known to be unreliable as species’ distinctive taxonomic characteristics are based on measurements, the presence/absence of various characters, but not colour. Furthermore, identification of immature individuals is almost impossible, as most diagnostic characteristics were derived from adult specimens.

Furthermore, adult and juvenile members of the genus *Kinosternon* are reported to be similar in appearance to other species in the Kinosternidae family such as *Claudius angustatus*, *Staurotypus* spp. and *Sternotherus* spp.

Potential benefit(s) for trade regulation of a transfer from Appendix II to I

Across their range, *Kinosternon* spp. are collected and trafficked for the pet trade (it is the fourth most trafficked turtle genus after *Podocnemis*, *Trachemys*, and *Chelonoidis*), and at least in some areas, adults are also actively hunted for consumption or for elaborate traditional medications. Regulating trade is needed to limit the unsustainable harvesting of wild populations which could be detrimental to some species, especially those that are poorly studied and whose habitats are already degraded.

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Inclusion of the Mexican Musk Turtle *Staurotypus triporcatus* and the Pacific Coast Giant Musk Turtle *Staurotypus salvinii* in Appendix II

Proponents: El Salvador, Mexico

Summary: *Staurotypus* is a genus of aquatic turtles, commonly known as giant musk turtles, Mexican musk turtles, or three-keeled musk turtles. The genus contains two recognised species: the Mexican Musk Turtle *Staurotypus triporcatus* found in Belize, Honduras, El Salvador, Guatemala, and Mexico, and the Pacific Coast Giant Musk turtle *Staurotypus salvinii* occurring in El Salvador, Guatemala, and Mexico. *Staurotypus salvinii* has been reported as an introduced species in Florida, USA; its current status there is unknown. *Staurotypus* species are typically much larger than other species of Kinosternidae. Both were categorised as Near Threatened on the IUCN Red List (1996), but these assessments may not reflect the species' current conservation status.

No recent population or status information is available for *S. triporcatus* or *S. salvinii*. Density estimates for *S. triporcatus* in 2009–2010 from the Rio Hondo region along the border of Belize and Mexico were relatively high (160 individuals/ha), but lower (40 individuals/ha) in areas near settlements. A 2013 study found that the species had virtually disappeared from the southern Mexican state of Chiapas. Another study in Veracruz state in Mexico found the population structure of *S. triporcatus* had changed from a high proportion of adults (80%) in 1997 to almost all juveniles (90%) in 2004.

The main threats to *S. triporcatus* have been identified as collection for human consumption and the pet market, as well as habitat loss. *Staurotypus triporcatus* is reported to be a favoured species for food in Mexico. Much of its potential habitat has been modified and fragmented by land-use changes. Habitat modification has increased harvest pressure on *S. triporcatus* by making small, localised populations more accessible to hunters. *S. salvinii* is likely to be affected by similar threats to *S. triporcatus*.

In Mexico, there are 14 captive breeding facilities registered for the management and use of *S. triporcatus*. Between 2013 and 2020 Mexico reported the export of just under 16,000 captive-bred live specimens of *S. triporcatus* mainly to mainland China (86%) and Hong Kong SAR (11%). Observations from seizures indicate that a proportion of specimens exported as captive-bred may in fact be wild-collected. In this period, but particularly from 2016 onwards, over 2,300 individuals exported from the USA (not a range State) were recorded as ranched. Illegal trade in *S. triporcatus* was reported in Mexico in 2020 with 503 individuals seized from a shipment at the Mexico City International Airport amongst a shipment of 15,000 turtles destined for China.

Little is known about trade in *S. salvinii*. USA data include records for nearly 3,000 individuals exported from the USA from 2008–2020, mostly registered as captive-bred, but also around 550 records of ranched and 40 of wild-sourced individuals originating in the USA. This same data show records for just over 100 wild-sourced individuals of this species imported by the USA from Guatemala from 2008–2010 (no imports from Guatemala have been recorded since then). This species is also globally available online, priced at USD140–995 (per hatchling) and advertised as captive-bred.

Differentiating between *S. triporcatus* and *S. salvinii* was reported to be difficult in adults and practically impossible in juveniles, and the defining characteristics were noted to be unreliable.

The species *Staurotypus triporcatus* is proposed for inclusion in Appendix II under the criteria in Annex 2a paragraph B of Resolution Conf. 9.24 (Rev. CoP17) and paragraph 2(a) of Article II of the Convention, and *S. salvinii* is proposed under the criteria in Annex 2b paragraph A.

Analysis: The giant musk turtles *Staurotypus* are relatively widespread Central American species that are harvested for food and for the international pet trade. Both have been assessed as Near

Threatened although these assessments have not been updated since 1996. *Staurotypus triporcatus* has been exported by Mexico in notable numbers (around 2,500 per year in the last ten years), principally to China. Exports are reported as captive-bred but there are indications that at least a proportion of these are wild-caught. Information on the wild status of both species is sparse but there are indications of population decreases in areas where the species is known to have been collected. It is not known how extensive wild collection is across the range of either species or what the impact of such collection is on their global populations. There is insufficient information to determine whether or not either species meets the criteria for inclusion in Appendix II in Res. Conf. 9.24 (Rev. CoP17). Juveniles of both species resemble each other: if either were considered to meet the criteria for inclusion in Appendix II, the other would meet lookalike criteria for inclusion in the Appendix.

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 Checklist of Chelonians of the World (Fritz and Havaš, 2007), the CITES Standard Reference, includes the proposed species *Staurotypus triporcatus* and *Staurotypus salvinii*.

Range

Staurotypus triporcatus: Belize, Honduras, El Salvador, Guatemala, and Mexico.

Staurotypus salvinii: El Salvador, Guatemala, and Mexico. *Reported as present in Florida, USA, (Smith et al., 2011) where suspected to be breeding. Its current status there is unknown.*

Introduced USA (Turtle Taxonomy Working Group, 2021)

IUCN Global Category

Staurotypus triporcatus – Near Threatened (assessed 1996 ver. 2.3)

Staurotypus salvinii – Near Threatened (assessed 1996 ver. 2.3)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

A study in 2004 found 90% of the individuals sampled were juveniles, with the lack of adults a potential indication of excessive extraction. In 2009 a survey of fishmongers and markets in Veracruz, Mexico found that *S. triporcatus* was most frequently consumed as adults due to their large size and consequently providing more meat.

In 2009–2010, Calderon-Mandujano measured densities of 40 individuals/ha of *S. triporcatus* around La Union on the Rio Hondo along the border of Belize and Mexico and densities of 160 individuals/ha in areas away from settlements.

National trade

In Mexico and Belize *S. triporcatus* is widely consumed. In Mexico, the use and legal trade of the species is carried out in “Unidades de Manejo para la Conservación de la Vida Silvestre” (UMA) which are Management Units for the Conservation of Wildlife, or “Predios e Instalaciones que Manejan Vida Silvestre” (PIMVS) which are Premises and Facilities that manage wildlife. According to the General Directorate of Wildlife (DGVS-SEMARNAT), trade has only been authorised from captive breeding stocks. According to data from the DGVS-SEMARNAT (2020), there are 14 UMA/PIMVS in Mexico that have registered the management and use of *S. triporcatus*. From 2015–2020, a total of 21,689 live *S. triporcatus* individuals were authorised for commercial use, all from captive breeding facilities, to be traded either at the national or international level.

International trade

Staurotypus triporcatus is highly commercialised within the international reptile market. Compared to other turtles such as *Claudius angustatus*, *S. triporcatus* is sold in larger quantities: an individual can fetch prices from USD79 for a hatchling or juvenile, to USD1,000 for a large adult.

In Mexico, since the year 2000 approximately 24,500 individuals have been authorised for export, most of them in the last 10 years. The Procuraduría Federal de Protección al Ambiente (PROFEPA) (the Federal Attorney for Environmental Protection), is responsible for monitoring compliance with the legal framework and verifying the permits and correct documentation for the trade and export of the species. *They indicate that in the seven-year period from 2013–2020 they have exported 65% of the total live specimens of S. triporcatus recorded since 2000*

(i.e., 15,884); the main destination for exports is mainland China (86%) followed by Hong Kong SAR (11%), and Japan (2%) and USA (1%).

According to the Supporting Statement from 2015–2020, 719 *S. triporcatus* specimens were imported by the USA from Mexico, of which 255 were wild-sourced for commercial purposes. Previously, in 1999–2014, the USA reported importing 619 individuals and exporting 2,666 (USFWS; LEMIS Data, 1999–2015).

Our analysis found that the USA trade in *S. triporcatus* increased between 2008 and 2020, with higher levels of *S. triporcatus* in trade compared to *S. salvinii* (Figure 1). From 2015–2020, the US Fish & Wildlife Service (USFWS) recorded the entry of 316 individuals into the USA from Mexico, however, for the same time period PROFEPA only registered 50 exports.

According to LEMIS 595 *S. triporcatus* individuals were imported into the USA from 2013–2020, apparently representing 1% of the total global exports. Among them, 395 were captive-bred and 200 were from wild populations. In 2015, 155 wild sourced *S. triporcatus* individuals were imported into the USA from Mexico and between 2008 and 2010, 23 wild-sourced individuals of the same species were imported into the USA from Guatemala. During the same time period, 111 wild-sourced *S. salvinii* individuals were imported into the USA from Guatemala. Between 2013 and 2015, 390 captive-born *S. triporcatus* were imported into the USA from Mexico (LEMIS).

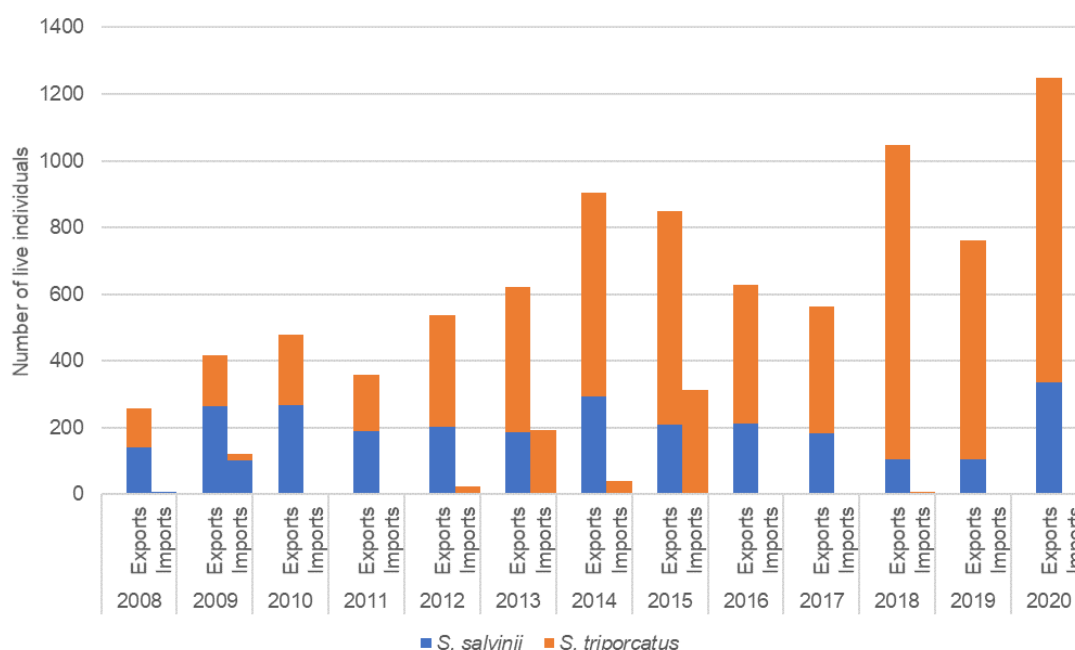


Figure 1. Imports and exports of both *S. triporcatus* and *S. salvinii* into and from the USA, 2008–2020 (LEMIS).

Additionally, 300 *S. triporcatus* individuals were recorded as in transit to Hong Kong SAR and originating in Mexico (100 in 2017 and 200 in 2020), which had not authorised trade in wild specimens. From 2016 onwards in particular, close to 2,300 *S. triporcatus* individuals (and an additional ~500 *S. salvinii*) were recorded in exports from the USA and coded as ranched and originating in the USA (*S. salvinii* has been reported as introduced to the USA, but there is no indication of this regarding *S. triporcatus*). *Staurotypus salvinii* is available online, priced from USD140–USD995 (per hatchling) and advertised as captive-bred (The Reptile Distributors, 2022; The Turtle Source, 2022).

A study by Sung et al. (2021) found that between 2017–2018, 97 individual *S. triporcatus* were recorded for sale; 28 were advertised on social media (Hong Kong SAR-based Facebook group specialising in turtle trade), 12 on an internet forum (publicly accessible Hong Kong SAR-based forum containing advertisements and husbandry advice) and 57 being sold in a physical market (Goldfish Market—the largest pet market in Hong Kong SAR). In the same study, 22 *S. salvinii* were found, 22 advertised on social media, five on an internet forum, but none found in physical markets.

Illegal Trade

Most of the seized turtles in the past decade come from two events that occurred in 2020: in the cargo area of Mexico City International Airport, a shipment of 15,000 turtles bound for China was intercepted due to not having the correct permits for export (503 were identified as *S. triporcatus*); and 368 *S. triporcatus* and 135 *S. salvinii*

were seized from a warehouse along with *Claudius angustatus* and five species of the genus *Kinosternon*. Although it was initially claimed the animals were captive-bred, several turtles still had hooks in their mouths, strongly suggesting they were wild-sourced. Such seizure events for *S. triporcatus* may represent only a part of the clandestine trade. The high demand for the species, from Asia in particular, makes it very prone to being illegally trafficked.

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

Between genera:

The genus *Staurotypus* can be distinguished from any of the turtles in the family Kinosternidae (genera *Kinosternon*, *Claudius*, *Sternotherus*), even for juvenile specimens: the plastron is reduced, and there are pronounced keels. To differentiate *Staurotypus* from *Claudius angustatus*, it is necessary to look at the carapace: in *C. angustatus* it is almost completely oval (three barely perceptible keels), and in *Staurotypus* the three keels are well-marked. *Staurotypus triporcatus* adults are larger than any of the tortoises in the family Kinosternidae.

Between the two species of the genus:

There are difficulties in identifying *S. triporcatus* (Mexican Musk Turtle) from *S. salvinii* (Pacific Coast Giant Musk Turtle), the only two species in the genus *Staurotypus*. The two species have an allopatric distribution: the first restricted to the Atlantic coastal plain of Mexico, Belize, Guatemala, and Honduras, and the second to the Pacific coastal plain of Mexico, Guatemala, and El Salvador. They differ in size: *S. salvinii* is much smaller, and often lighter coloured; in adults, the maximum size reached is 250 mm carapace length compared to the 400 mm carapace length of *S. triporcatus*. Some features that can help identification but are not completely reliable include the length of the inter-abdominal seam (14–19% of the length of the plastron in *S. salvinii*), the mottled or unicoloured head of *S. salvinii*, and (if present) the non-highlighted mottled/reticulate pattern of *S. salvinii*, unlike *S. triporcatus*. The dorsolateral carapacial keels of *S. salvinii* do not extend from the anterior to posterior margins, as they do in *S. triporcatus*. In the juvenile phase, it is almost impossible to differentiate the two species. Furthermore, adult *S. salvinii* and juvenile *S. triporcatus* may be similar sizes. Due to this resemblance, there is a possibility that *S. salvinii* is misdeclared in trade as *S. triporcatus*.

Additional Information

Threats

According to Reynoso *et al.* (2016, 2021), the main risks to populations of *S. triporcatus* are the *unauthorised or unregulated* collection of individuals from the wild for human food for sale in domestic and international markets. Fishermen prefer three-backed turtles (*Staurotypus* spp.) over other species, since they are larger and therefore provide more meat. The selective removal of reproductive adults poses a great risk to the population; large numbers are captured, especially at the beginning and end of the rainy season when their capture is easier due to the shallow depth of water bodies. In Oaxaca, a new road gave hunters access to an important turtle gathering area: in less than six months the numbers of adult turtles at this site were reduced by 90% (Legler and Vogt, 2013).

Conservation, management and legislation

The main legal instruments regulating the use and conservation of wild species in Mexico, as well as their habitats and ecosystems, are the Ley General de Equilibrio Ecológico y Protección al Ambiente (LGEEPA) ("General Law of Ecological Balance and Environmental Protection"), as well as the Ley General de Vida Silvestre (LGVS) ("General Law of Wildlife"). The LGVS establishes the types of management and exploitation that can be carried out in Mexico. It is only possible to harvest wild species through Management Units for the Conservation of Wildlife (UMAs) or Properties or Facilities that Manage Wildlife (PIMVS) with a management plan that details particular actions for the species. In the Norma-Oficial-Mexicana NOM-059-SEMARNAT-2010 (SEMARNAT, 2010), *S. triporcatus* is listed as threatened. Any species on the list of species at risk (NOM-059-SEMARNAT-2010) in the threatened category, can only be extracted from the wild when priority is given to collection and capture for restoration activities, repopulation, reintroduction and scientific research, and that 1) the specimens are the product of controlled reproduction, which in turn contributes to the development of populations in programmes, and 2) contributes to the development of populations through controlled reproduction, in the case of specimens of wild species in free life (Art. 84 and 85 of the LGVS).

Staurotypus triporcatus was reported to occur in seven Natural Protected Areas (ANPs) in Mexico (Los Tuxtlas Biosphere Reserve, Pantanos de Centla Biosphere Reserve, Laguna Popotera wetland area, Laguna de Terminos Flora and Fauna Protection Area (APFF), APPF Chan-Kin, Arroyo Moreno and Laguna Catatzajá System and the Calakmul Biosphere Reserve). The distribution of *S. triporcatus* that overlaps with the ANPs is 7,905 km², equating to around 9% of the species' total distribution in Mexico.

Staurotypus salvinii is listed as “subject to special protection” on NOM-059, and therefore it has the same management, legislation and protection as *S. triporcatus*.

Captive breeding

According to data from the DGVS-SEMARNAT (2020), there are 14 UMA/PIMVS in Mexico that have registered the management and use of *S. triporcatus*. From 2015–2020, a total of 21,689 live individuals of *S. triporcatus* were authorised for commercial use, all from captive breeding.

Implementation challenges (including similar species)

See section on Inclusion in Appendix II to improve control of other listed species.

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Inclusion of all Musk turtles in the genus *Sternotherus* in Appendix II

Proponent: United States of America

Summary: The genus *Sternotherus* commonly known as musk turtles comprises four species (*Sternotherus depressus*, *S. minor*, *S. odoratus* and *S. carinatus*.), which occur in the freshwaters of the North American continent from southeastern Canada to eastern Florida. *Sternotherus* species are small aquatic turtles, with the largest species reaching a maximum shell length of 17 cm (*S. carinatus*). Musk turtles are in international trade, primarily for the pet trade in Asia. Unlike many other turtle species, these musk turtles are generally not used for human food as their musk glands secrete an odour when threatened, making them less appetising. *Sternotherus* spp. appear to be most impacted by habitat modification, degradation, and loss.

All *Sternotherus* species are proposed as meeting Criterion B of Annex 2a of Res. Conf. 9.24 (Rev CoP17).

- The Razor-backed Musk Turtle ***S. carinatus*** is endemic to the USA and has the second largest range of any *Sternotherus* (ca 365,000 km²), largely in Louisiana, as well as parts of Arkansas, Oklahoma, Texas, Alabama, and Mississippi. It was assessed as Least Concern on the IUCN Red List in 2010 although in a 2003 study was ranked highly as a species vulnerable to live capture. Population numbers are currently unknown but are thought to be generally stable. Records show that over 830,000 individuals have been exported by the USA since 2008 (~64,000 per year) with 85% reported as ranched or sourced from the wild. An unknown proportion of these may be captive-born or captive-bred.
- The Flattened Musk Turtle ***S. depressus*** has the most restricted range (~7,000 km²) of any *Sternotherus*, being restricted to the Black Warrior River watershed in north-central Alabama. It was listed as Critically Endangered on the IUCN Red List in 2010, with an unknown population size. Since the 1960s, *S. depressus* has declined severely throughout much of its range, largely attributed to habitat degradation. Since 1987, *S. depressus* has been listed as threatened on the US Endangered Species Act (ESA), and any import, export or take of the species is prohibited. USA LEMIS data show some export of ranched specimens in 2018 although this may be a reporting error.
- The Loggerhead Musk Turtle ***S. minor*** occurs mostly in east-central Georgia and has a range of over 150,000 km². It was listed as Least Concern on the IUCN Red List in 2010 and is typically found at densities of over 100 individuals/ha, with the highest densities reported at 2,857 individuals/ha in 1979 in northwestern Florida. Population numbers are currently unknown but are likely to be in the tens of millions. Records show that over 75,000 individuals have been traded since 2008 (>6,000 per year). Over 86% of these turtles exported by the USA were reported as ranched or sourced from the wild.
- The Common Musk Turtle ***S. odoratus***, occurring from southern Canada to Florida with an estimated range exceeding 2 million km², is the most abundant species in the genus, and is a popular choice of pet. The species was assessed as Least Concern on the IUCN Red List in 2010 with an overall stable population other than in southern Canada where it was reported as decreasing. It is abundant in suitable habitat across its range, with densities up to 700 individuals/ha. Records show that over 1,300,000 individuals have been exported by the USA for the international pet trade since 2008 (~100,000 per year) with 79% reported as ranched or sourced from the wild. An unknown proportion of these may be captive-born or captive-bred.

Musk turtles in the genus *Sternotherus* are very similar to the American mud turtles in the genus *Kinosternon* but tend to be larger and have a more domed carapace, with a distinctive keel along their centre. Musk turtles resemble other members of the family Kinosternidae (*Claudius angustatus* and *Staurotypus* spp.), however, close inspection of the plastron and other head characteristics and marginal scutes allow immature individuals to be differentiated between genera. The plastron can also be used to differentiate between the genera *Sternotherus* and *Kinosternon*.

Analysis: The four species of *Sternotherus* musk turtle are largely confined to the USA. One species, *S. depressus*, has a restricted range and fragmented distribution, and has been assessed by IUCN as Critically Endangered. The other three are widespread or very widespread species which may be at least locally abundant, capable of reaching high population densities (over 2,800 individuals/ha) in ideal conditions. It seems likely that their global populations are very large. The commonest of the species (*S. odoratus*) is a popular pet and has been exported in relatively large numbers (~100,000 per year). A significant proportion of these are recorded as wild-collected or ranched although indications from other chelonians in trade in the US indicate that some may be captive-born or captive-bred. There is no information available on the impact of collection for export on wild populations and harvest has not been specifically identified as a cause of concern for any of the species. *Sternotherus depressus* has been legally protected from collection for trade since 1987. Overall, none of the species of *Sternotherus* is likely to meet the criteria for inclusion in Appendix II in 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

Sternotherus carinatus (Gray, 1856), *Sternotherus depressus* (Tinkle and Webb, 1955), *Sternotherus minor* (Agassiz, 1857), *Sternotherus odoratus* (Latreille in Sonnini and Latreille, 1801) as defined in the CITES Standard Taxonomic Reference for turtles, Fritz and Havaš (2007).

A taxonomic analysis of *Sternotherus minor* in 2018 led to the elevation of *S. peltifer* and the recognition of an additional species *S. intermedius* (Scott, Glenn, and Rissler, 2018). However, this proposal follows the taxonomy of Fritz and Havaš (2007).

Range

Sternotherus odoratus: Canada, USA, Mexico (unconfirmed according to the SS)

Sternotherus carinatus: USA

Sternotherus minor: USA

Sternotherus depressus: USA

IUCN Global Category

Sternotherus depressus - Critically Endangered A2bce+4bce (assessed 2010, ver. 3.1)

Sternotherus carinatus - Least Concern (assessed 2010, ver. 3.1)

Sternotherus minor - Least Concern (assessed 2010, ver. 3.1)

Sternotherus odoratus - Least Concern (assessed 2010, ver. 3.1)

Biological criteria for inclusion in Appendix I

B) Restricted area of distribution

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

Endemic to North America, the genus *Sternotherus* occurs in eastern USA and southern Québec and southeastern Ontario, Canada and possibly northern Mexico.

Sternotherus odoratus

The most wide-ranging species of the genus, found from southern Maine, Québec and Ontario (Canada), south to Florida, west into central Texas, and eastern Oklahoma and Kansas. The record of a single specimen of *S. odoratus* collected in 1903 in Chihuahua, Mexico, is unconfirmed. The remaining species in the genus are more localised within the southern region of the USA.

The species is listed as Least Concern and appears to have a stable population size. *Sternotherus odoratus* was reported to be common to extremely abundant in suitable habitat within its range. Density reports for this species range from 8–700 ind./ha, and 8.4–41.7 kg/ha biomass.

- Individuals mature at three years in the southern portion of their range (e.g., Florida), but at 5–8 years in the north (e.g., Oklahoma).
- Females typically produce 2–4 eggs per clutch (clutch size ranging from 1–13).

According to LEMIS, over 1.3 million live *S. odoratus* individuals were exported by the USA from 2008–2020 for commercial purposes. Of this trade 58% (>750,000) were reported as wild-sourced individuals, primarily imported by Hong Kong SAR (18%), the UK (17%), and mainland China (15%). An unknown proportion of individual chelonians in trade reported as wild-sourced may be captive-bred or captive-raised (Association of Fish and Wildlife Agencies, undated).

Sternotherus carinatus

This species is found largely in Louisiana, as well as portions of Arkansas, Oklahoma, Texas, Alabama, and Mississippi. The species was listed as Least Concern in 2010, with an unknown population size. Anecdotal evidence indicates that *S. carinatus* is abundant and stable in a variety of locations across its range.

It was ranked as the fifth most vulnerable non-marine turtle species in the USA in 2003 due to its vulnerability to the commercial pet trade. The ranking was based not on its “value” to dealers, but on the species’ demography and limited range.

- Individuals mature at 4–8 years.
- Females produce a clutch of 3–7 eggs per nesting season.

A total of 836,240 live *S. carinatus* individuals were exported by the USA from 2008–2020 for commercial purposes (LEMIS, 2022). Of this trade 53% (>400,000) were in wild-sourced individuals, primarily imported by Hong Kong SAR (49%) and mainland China (21%). Regarding illegal trade, 1,103 *S. carinatus* individuals were found as part of 1,529 tortoises in luggage intercepted in Manila (Philippines) and 7,242 turtles seized at Lantau Island (Hong Kong SAR) (TRAFFIC, 2022).

Sternotherus minor

Occurs mostly in east-central Georgia, neighbouring northern Florida and extreme southeastern Alabama. *Sternotherus minor* are “generally abundant in suitable habitat.” Loggerhead Musk Turtles reach among the highest densities known for any turtle species. The species has been found consistently at densities of over 100 ind./ha, with the highest density calculated at 2,857 ind./ha under ideal conditions in a northwestern Florida springhead.

The species is listed as Least Concern, with an unknown population size.

- Individuals mature at 6–8 years.
- Females produce typically three (ranging from 1–5) clutches per year, of 1–5 eggs per clutch, however, their known reproductive potential is only 6–12 eggs per year.

Very rough extrapolation estimates, based on presumed historic ranges and densities suggest that the population size could be in the tens of millions.

A total of 75,149 live *S. minor* individuals were exported by the USA from 2008–2020 for commercial purposes (LEMIS, 2022). Of this trade, 55% (>40,000) were wild-sourced individuals, primarily imported by Hong Kong SAR (71%). Regarding illegal trade, in 2017 over 200 live baby turtles were seized, and half died (TRAFFIC, 2022).

Table 1. Information of presumed historic home range, density, occupancy and population trend (TTWG, 2021).

| | <i>S. carinatus</i> | <i>S. minor</i> | <i>S. odoratus</i> | <i>S. depressus</i> |
|---------------------------|--|---|--|---------------------------------------|
| Presumed Historical range | 365,144 km ² | 154,167 km ² (over 15 million ha) | 2,104,014 km ² (over 210 million ha) | 7,285 km ² |
| Density information | - | 100–2857 ind./ha | 8–700 ind./ha | - |
| Occupancy | - | - | - | Only 7% of historic range undisturbed |
| Population | Stable (Decline in Pascagoula River population) | Seems stable but not enough information | Stable (Declines in Canada) | - |

Sternotherus depressus

The Flattened Musk Turtle *Sternotherus depressus* has the most limited distribution of all species in the genus. It is restricted to the Black Warrior River watershed in north-central Alabama, above the Bankhead Dam.

Home ranges for *S. depressus* populations can range up to one km, but typically consist of a few hundred metres of stream length or less (Jenkins, 2019). While they have relatively small home ranges, they sometimes make large movements that greatly expand their home ranges, thus they require large sections of intact stream habitat to thrive.

Sternotherus depressus was assessed as Critically Endangered in 2010 with an unknown population size. According to Dodd (1990), the species' historical suitable habitat had been degraded to the point that the population had been extirpated from 57% of its former areas, with 37% severely degraded and containing remnant populations, and only 7% of its original habitat remaining reasonably unaffected by pollution, sedimentation and impoundments. Today, it is now confined to areas where pollution, sedimentation, and impoundments have not entirely altered its habitat.

Sternotherus depressus is listed as threatened under the ESA with import, export or take of this species prohibited. Populations of *S. depressus* occur in the Bankhead National Forest, including the Sipsey Wilderness Area. However, no designated protected reserves within the national forest include this species.

- There is no information for age of maturation.
- Females produce 1–2 clutches of 1–4 eggs per year.

In 2018, 640 live, ranched individuals were exported by the USA to mainland China for commercial purposes (LEMIS, 2022).

Sternotherus spp.

According to LEMIS, 18,017 individuals were exported at genus level by the USA from 2008–2020. Of this trade 60% (>10,000) were in wild-sourced individuals, primarily imported by Hong Kong SAR (32%).

Over 2.2 million live individuals of *Sternotherus* species were exported for commercial purposes by the USA, 2008–2020, comprised predominantly of *S. odoratus* (58%) and *S. carinatus* (37%) (Figure 1).

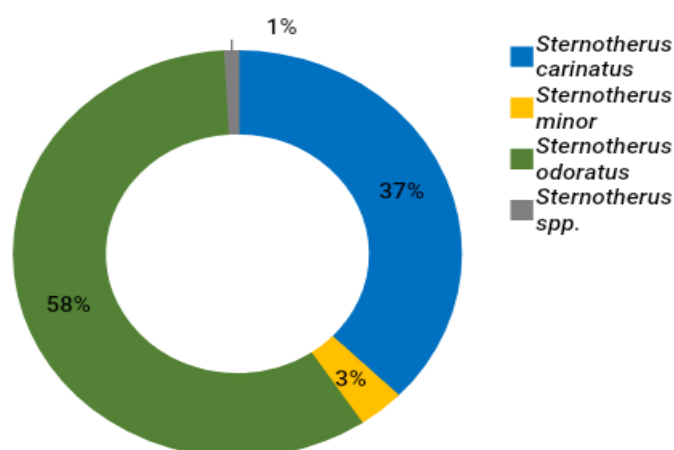


Figure 1. Proportions of live *Sternotherus* individuals exported by the USA 2008–2020 (640 *S. depressus* not included in figure; LEMIS, 2022).

Table 2. Numbers of individuals legally exported from the USA, 2008–2020 (LEMIS, 2022).

| Species | Source | No. of individuals | Percentage of total |
|---------------------|--------------|--------------------|---------------------|
| <i>S. carinatus</i> | Captive-bred | 87,317 | 10 |
| | Captive-born | 40,411 | 5 |
| | Ranched | 261,933 | 31 |
| | Wild | 446,579 | 54 |
| | Total | 836,240 | |
| <i>S. minor</i> | Captive-bred | 6,710 | 9 |
| | Captive-born | 3,651 | 5 |
| | Ranched | 23,237 | 31 |
| | Wild | 41,641 | 55 |
| | Total | 75,239 | |
| <i>S. odoratus</i> | Captive-bred | 158,543 | 12 |
| | Captive-born | 122,422 | 9 |
| | Ranched | 271,360 | 21 |

| Species | Source | No. of individuals | Percentage of total |
|---------------------|--------------|--------------------|---------------------|
| | Wild | 756,638 | 58 |
| | Total | 1,308,963 | |
| <i>S. depressus</i> | Captive-bred | 0 | |
| | Captive-born | 0 | |
| | Ranched | 640 | 100 |
| | Wild | 0 | |
| | Total | 640 | |

Sternotherus species are popular in Asian markets. For example, a study by Sung et al. (2021) found three of the four *Sternotherus* species on sale in Hong Kong SAR between 2017 and 2018, advertised through social media (a Hong Kong-based Facebook group specialising in turtle trade), on an internet forum (publicly accessible Hong Kong-based forum containing advertisements and husbandry advice) and in a physical market (Goldfish Market – the largest pet market in Hong Kong SAR) (see Table 3).

Table 3. Information on the species found offered for sale on a social media group, an internet forum and physical market between July 2017 and June 2018 (excluding October 2017).

| Species | Social Media | Internet Forum | Physical Market |
|---------------------|--------------|----------------|-----------------|
| <i>S. carinatus</i> | 26 | 22 | 688 |
| <i>S. minor</i> | 21 | 9 | 65 |
| <i>S. odoratus</i> | 19 | 6 | 583 |
| <i>S. depressus</i> | 0 | 0 | 0 |

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

Additional Information

Threats

Sternotherus species are primarily threatened by habitat modification, degradation and loss. Sedimentation, snagging operations to remove deadwood from waterways for navigation purposes, mining for sand and gravel, impoundment, hydrologic changes, and toxic pollutants have a direct negative effect on musk turtles and their molluscan and insect prey base. Increased boat traffic on rivers and springs disturbs the habitat of these turtles. The wakes from boats cause increased turbidity of the water and erosion of the shoreline, thus adversely impacting aquatic vegetation and prey sources, and reducing the long-term suitability of their habitat.

Musk turtles are also susceptible to collection for the pet trade. Reed and Gibbons (2004) ranked *S. carinatus* as the fifth most vulnerable non-marine turtle species in the USA with regards to its vulnerability to the commercial pet trade. The ranking was based not on its “value” to dealers, but on the species’ demography and limited range. Given their small size and presence of a musk gland that can taint food, *Sternotherus* are not typically used for food or medicinal use in Asia.

Sternotherus minor and *S. odoratus* were reported to be injured or killed as a result of boat propeller strike which was noted to be a marked cause of injury or mortality.

For the highly restricted *S. depressus*, which has likely been extirpated from more than half of its former range due to habitat modifications to the stream and river channels in the Warrior River Basin (Alabama), individuals in the remaining viable habitats continue to be vulnerable to disease and human-related disturbance, collection for the pet trade and habitat modification. Disease has already played a role in the significant decrease of at least one population of *S. depressus*, with impacts on other populations known but not quantified. Fragmented habitats of small populations increase their susceptibility to human-caused impacts.

Conservation, management and legislation

Different USA states have differing laws. In Georgia, prior to 2011, a permit allowed a person to harvest up to 1,000 turtles from the wild, with this now reduced to 300 individuals (The Augusta Chronicle, 2022). However, from 2018/2019 a commercial turtle permit is needed to collect 10 or more freshwater turtles unless state or federally protected. No permit is needed for fewer than 10 but it is illegal to collect any native turtle eggs from the wild (Georgia Wildlife, 2022). Although there is legislation protecting *S. depressus*, there are no federal regulations for the remaining *Sternotherus* species at a national level.

Sternotherus carinatus

Although habitat preserves have not been designated specifically for *S. carinatus*, populations of this species are found in eight National Forests, one State Wilderness Area, 17 National Wildlife Refuges, one National Preserve, a Ramsar Convention Wetland of International Importance, and two private nature preserves.

Sternotherus depressus

Listed as threatened under the ESA with import, export or take of this species prohibited. Populations of *S. depressus* occur in the Bankhead National Forest, including the Sipsey Wilderness Area, however, no designated protected reserves within the national forest include this species. In 2019, the U.S. Fish and Wildlife Service initiated a five-year status review of the species; however, to date, the status review has not yet been completed.

Sternotherus minor

Occurs in a substantial number of protected springheads and spring runs in Florida. Management measures include public awareness and education to reduce wanton destruction of this and other turtle species, appropriate management of protected areas and other suitable habitats, and monitoring of key populations.

Artificial propagation/captive breeding

Due to their small size and easy care, musk turtles are a popular choice for a pet, with S. odoratus currently reported as the top choice among aquatic turtle species. The popularity of these small turtles has made them widely available at pet and reptile stores. In captivity, it is common for these turtles to live for 30–50 years.

Musk turtle species are known to be produced domestically by turtle farms in the southeastern USA, however, the quantity of individuals produced remains unknown. It is unknown if turtle farms in Asia are also producing *Sternotherus* species. For many slow-reproducing species of turtle, such as *Sternotherus*, the high cost of maintenance in captivity is understood to make large-scale commercial breeding unprofitable.

Sternotherus species are sold globally online, for example hatchlings of S. minor priced at GBP45 (~USD53) each advertised on a UK website (Preloved, 2022). Another UK website advertised S. odoratus hatchlings priced at GBP90.00 each (~USD106; Angelfish Aquatics, 2022) while another was selling captive-bred hatchlings for GBP27.50 (~USD33; the Reptilium, 2022). Online searches show many Sternotherus individuals for sale globally are hatchlings and claim to be captive-bred.

Implementation challenges (including similar species)

Musk turtles in the genus *Sternotherus* are very similar to the American mud turtles in the genus *Kinosternon*, but tend to have a more domed carapace, with a distinctive keel down the centre. Each species in the genus *Sternotherus* can be easily distinguishable from each other.

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Inclusion of Softshell Turtles *Apalone* spp. in Appendix II

Proponent: United States of America

Summary: The genus *Apalone* comprises three species (*A. ferox*, *A. mutica* and *A. spinifera*) of freshwater turtles from the family Trionychidae, all native to the USA, with *A. spinifera* also native to Canada, and Mexico. The species were included in Appendix III by the USA in 2016 and one subspecies of *A. spinifera* (*A. s. atra*) has been included in Appendix I since 1975.

Few population studies have been conducted on members of the Trionychidae family and the genus *Apalone* is one of the least studied softshell turtle genera in North America with no complete population estimates of any species.

- ***Apalone ferox*** inhabits shallow, still and brackish waters in South Carolina, Georgia, Alabama, and Florida. No population estimates are available, although the species was considered common across its range in 2010. Limited studies suggest some localised declines.
- ***Apalone mutica*** is restricted to medium to large rivers with sandbars widely distributed across 22 states in the USA. Despite a lack of empirical data on population size and trends of *A. mutica*, there are anecdotal reports of declining populations over the years.
- ***Apalone spinifera*** has the largest range in the genus, occurring in most of the USA east of the Rocky Mountains as well as southern parts of Canada and northern Mexico. Populations were considered to be generally stable in 2010, but historical declines have been postulated by a number of studies.

Apalone species are productive, with mature females capable of producing multiple egg clutches annually (up to 7 in the case of *A. ferox*). Average clutch size is highly variable but averages over 20 in mature *A. ferox*.

All three species were classified as Least Concern by IUCN in 2010. The Canadian population of *A. spinifera* was designated as endangered in 2016 by the Committee on the Status of Endangered Wildlife in Canada and is currently listed as threatened on Schedule 1 of the Species at Risk Act (SARA; Environment and Climate Change Canada 2018).

Turtles have a life history strategy that entails slow growth and late maturity (e.g., *A. mutica* males mature at four years and females at nine) and longevity (over 30 years).

Softshell turtle species can be readily farmed and freshwater turtle species from North America are generally viewed as easier to breed than most species native to Asia—this has resulted in increased demand for *Apalone* specimens from turtle breeding farms based in the USA. Demand for wild-caught specimens as breeding stock for such farms continues as these are considered generally of superior genetic stock.

Hatchlings, juveniles and adults of softshell turtles are traded for different purposes. Hatchlings either enter the pet trade or are exported to establish turtle breeding farms overseas. Turtles larger than 3.5 to 4.5 kg are generally sold as meat, whereas smaller turtles are killed, frozen, and sold whole. Turtles under 3 kg are usually sold live to be either used in the pet trade or to be raised and later killed for meat. From 2017 to 2020, exports of live turtles accounted for almost all trade (>99%).

According to US data, between 2008 and 2020 just over three million *Apalone* specimens were exported (87% *A. ferox*, 13% *A. spinifera* and >1% *A. mutica*). Over half of these (57%) were reported as ranched, followed by 28% captive-bred, 8% captive-born and 7% wild. The genus was listed in CITES Appendix III in 2016. Trade patterns appear to have altered since then. CITES trade data show just under 100,000 specimens of *Apalone* spp. in trade for the period 2017–2020, virtually all recorded as captive-born *A. ferox*, with negligible numbers of *A. spinifera*. Less than 1% of the total was reported as wild-collected and no trade was reported in *A. mutica*. Most live exports were destined for

mainland China, followed by Macao SAR and Hong Kong SAR. Analysis of US data found an overall decrease of 93.5% in the annual number of directly exported live *A. ferox* specimens for commercial purposes from the USA from 2010 to 2020. The shift in declaration of specimens from captive-bred to captive-born may reflect a stricter interpretation of the former source category, requiring closed-cycle breeding with no or minimal involvement of wild-caught founder stock.

There is evidence of some illegal trade, although this appears to be generally at a low level.

Regulations regarding harvest and trade of the three species vary across their distribution range. *Apalone* spp. are protected in Canada. In the USA, states have their own regulations. At present, Mexico has no legal instruments in place for either species, but use of all wildlife has been managed through the Wildlife Management and Sustainable Use Units (UMAs) since 1996.

The genus *Apalone* is proposed for inclusion in Appendix II under the criteria in Annex 2a paragraphs A and B of Resolution Conf. 9.24 (Rev. CoP17), with the exception of the subspecies (*Apalone spinifera atra*), already included in Appendix I.

Analysis: The three *Apalone* species are all widespread in North America and all were most recently (in 2010) assessed as Least Concern on the IUCN Red List. Two of the species (*A. ferox* and *A. spinifera*) have been exported in large numbers in the past, chiefly to East Asia where they are widely consumed as food. The great majority of recent trade is in *A. ferox* and most exports are reported as captive-born within the USA, with very small numbers declared as wild-collected. Production of captive-born stock may to some extent depend on input from wild-collected animals, but information on numbers of animals collected for this purpose is lacking and there is little indication of depletion of wild populations through over-collection. No recent trade has been reported in *A. mutica* and very little in *A. spinifera*. Little information is available on current populations or any impact of wild collection. Given their widespread distribution, local abundance and the predominance of captive sourced specimens in exports it seems unlikely that any species of the genus meets the criteria for inclusion in Appendix II in 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 proposal concerns three different species:

- *Apalone ferox* (Rafinesque, 1832 1817)
- *Apalone mutica* (LeSueur, 1827)
- *Apalone spinifera* (LeSueur, 1827)

CITES standard reference: Fritz, C. and Havaš, P. (2007). Checklist of chelonians of the world. *Vertebrate Zoology*: 57: 149-368.

Apalone spinifera comprises six distinct subspecies: *A. spinifera spinifera* (LeSueur, 1827), *A. spinifera atra* (Webb and Legler, 1960), *A. spinifera aspera* (Agassiz, 1857), *A. spinifera emoryi* (Agassiz, 1857), *A. spinifera guadalupensis* (Webb, 1962), and *A. spinifera pallida* (Webb, 1962).

Apalone spinifera atra (which is currently in Appendix I) is not considered in the proposal and remains unchanged.

Range

Apalone ferox: USA

Apalone mutica: USA

Apalone spinifera: Canada, Mexico, USA

IUCN Global Category

- *Apalone ferox*: Least Concern (assessed 2010, ver. 3.1)
- *Apalone mutica*: Least Concern (assessed 2010, ver. 3.1)
- *Apalone spinifera*: Least Concern (assessed 2010, ver. 3.1)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)**A) Trade regulation needed to prevent future inclusion in Appendix I**

Whilst all three species are native to the USA, *Apalone spinifera* is also found in Canada and Mexico. More specifically, the distribution range of *Apalone ferox* extends from southern South Carolina through eastern and southern Georgia, south-eastern Alabama and virtually all of Florida (van Dijk, 2011a). *Apalone mutica* occurs within the greater Mississippi basin from Louisiana up to North Dakota, Minnesota and western Pennsylvania, in addition to the river systems of Colorado, Brazos, Sabine, Pearl, and Escambia (van Dijk, 2011b). *Apalone spinifera* inhabits the southern regions of Ontario and Quebec in Canada, northern Mexico from Chihuahua to Tamaulipas, and most of the USA east of the Rocky Mountains (van Dijk, 2011c).

Apalone ferox females lay 2–7 clutches a year, with clutch size correlating with body size. Development takes an average of 76.4 days. One study of 32 eggs calculated a hatching success rate of 81.25%. Iverson and Mohler (1997) found an average clutch size of over 20 (max 38), with females in the breeding season potentially producing a new clutch every three weeks. The species had one of the highest egg production rates known among reptiles.

In *A. mutica* female clutch sizes can vary from 1–33 eggs, with an average of 6–8 eggs and up to three clutches laid yearly. One study examining *A. mutica* in Louisiana found an 82% hatch rate and 75% nest survivorship rate. *Apalone spinifera* females nest in sandy areas and clutch sizes range from 3–39 eggs with 2–3 clutches laid per year.

Only partial quantitative estimates on population size are available. *Apalone ferox* is described as common, but the species' population size has not yet been quantified. There are anecdotal reports of declining populations of *A. mutica*. Aggregations of up to 88 individuals have been recorded (Trauth et al., 2004). More data, although outdated, are available on *A. spinifera*. The Supporting Statement mentions studies from the 1990s that report differences in the number of individuals across their range in the USA. In Canada, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) stated that the population of mature individuals amounted to 1,000 specimens and the last population in Quebec counted fewer than 50 adult females in 2016. Changes in population numbers over time are unknown and the limited data available do not allow for an updated assessment of the population status of each species.

In the USA, trade of the species is regulated by state. For instance, in Florida, regulations include a ban in (i) taking turtle eggs from the wild, (ii) harvesting softshell turtle specimens from the wild from 1 May to 31 July, and (iii) selling, buying or possessing any specimen that has been taken from the wild. Canadian legislation prohibits the capture, harassment or killing of the species.

Turtle species, particularly softshell turtles, are farmed for sale as food in China on a very large scale. One 2008 study (Haitao et al., 2008) estimated annual production of the Chinese Softshell Turtle *Pelodiscus chinensis* at over 300 million individuals. The study noted that non-native species were also bred in increasing numbers.

Specimens of the *Apalone* genus are internationally traded within both the food and the pet trades and mainly as live turtles. Most live exports (as reported by the USA) are destined for mainland China, followed by Macao SAR and Hong Kong SAR. Austria, Belgium, Canada, Germany, Italy, Japan, Morocco, Mexico, the Netherlands, Portugal, and the UK altogether constituted only 2.1% of total live exports from 2016–2021.

The three species *A. ferox*, *A. mutica* and *A. spinifera* were included in Appendix III in 2016. Since then, the CITES Trade Database has registered a total of 96,830 live turtles being exported from 2017 to 2020 (99.6% were *A. ferox*, followed by 0.4% of *A. spinifera*). Among these, 99.3% were captive-born and 0.7% taken from the wild, and 88% of the exports were from the USA.

According to LEMIS, a total of 171,007 live *A. ferox* (87% of them captive-born) and 1,623 live *A. spinifera* (82% of them taken from the wild) were traded from 2016–2021. Reported exports from 2016–2021 showed >99% of trade involved live turtles, with low quantities of other trade commodities reported.

Analysis of LEMIS data found a total of 3,039,375 live specimens reported as exported from 2008 to 2020, among which 87% were *A. ferox*, 13% were *A. spinifera* and >1% were *A. mutica*. Examination of data from 2010 to 2020 shows evidence of a decrease in direct live *A. ferox* and *A. spinifera* exports from the USA for commercial purposes after 2015 (Figure 1).

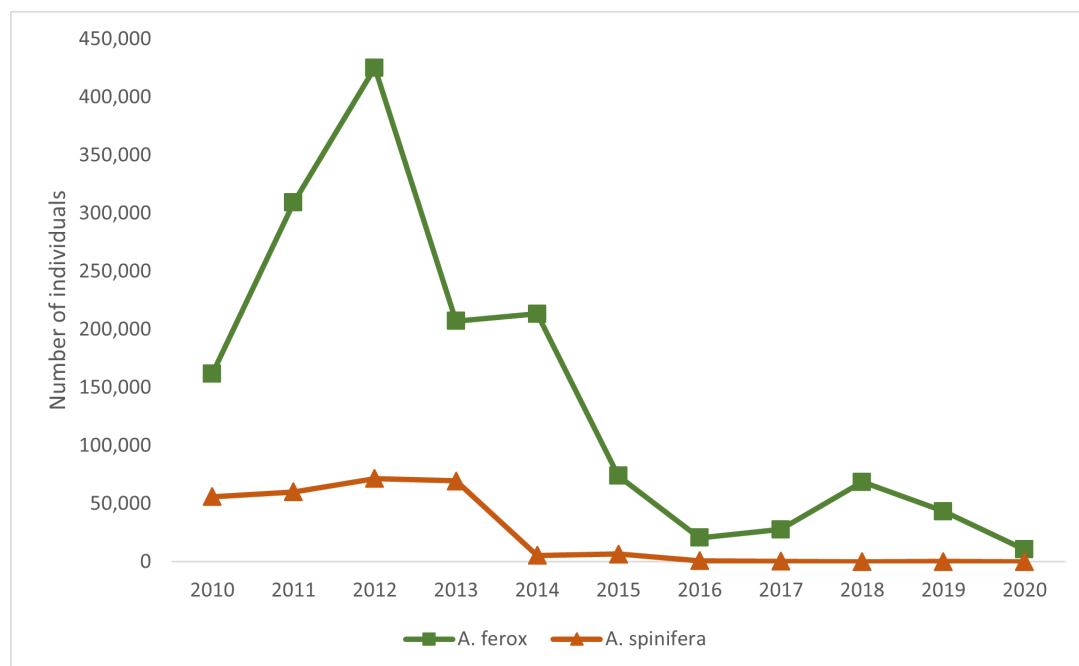


Figure 1. Number of live specimens of *A. ferox* and *A. spinifera* directly exported from the USA for commercial purposes 2010–2020. The vertical axis indicates number of specimens. Trade data includes captive-bred, captive-born, ranched and wild-sourced specimens. *Apalone mutica* is not shown on the graph as only one instance of export of 230 live specimens was reported from 2012, (LEMIS, 2022).

LEMIS data from the same time period (2010–2020) for *A. ferox* alone (i.e., the most traded species among the three—approximately 85% of all exported specimens), shows apparent decreases in captive-bred and ranched specimens being directly exported from the USA for commercial purposes over the years, with zero individuals registered from 2016 onwards (Figure 2). Conversely, reported direct exports in captive-born individuals increased considerably from 2016 to 2018, to then decrease again. Overall, this analysis of LEMIS data found a 93.5% decrease in the annual number of exported live *A. ferox* specimens from 2010–2020 (see Table 1 for yearly differences in number of exported specimens 2010–2020).

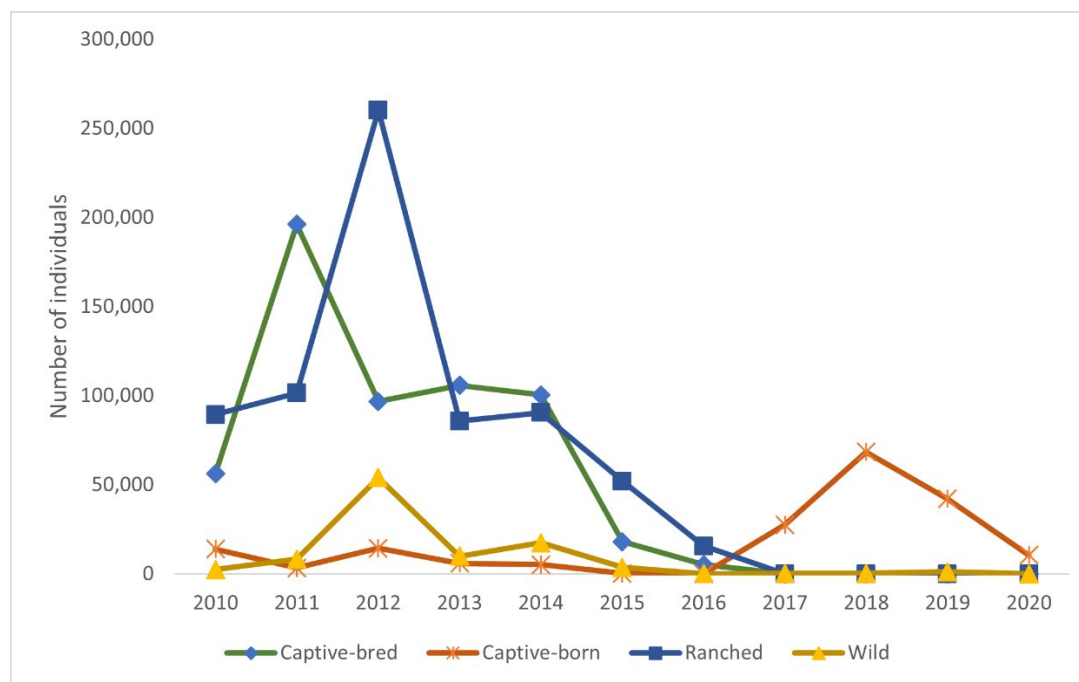


Figure 2. Number of live specimens of live *A. ferox* directly exported from the USA for commercial purposes 2010–2020. The highest annual number of exported specimens (260,131) occurred in 2012 and involved ranched individuals. No records of exports from the USA in either captive-bred or ranched live specimens for commercial purposes have been registered since 2016, (LEMIS, 2022).

Table 1. Total number of exported *A. ferox* specimens per year. Considerably lower numbers of specimens have been exported since 2015 onwards. Trade data include cleared records of captive-bred, captive-born, ranched and wild-sourced individuals directly exported from the USA for commercial purposes, (LEMIS, 2022).

| Year | Total number of yearly exported specimens |
|------|---|
| 2010 | 161,856 |
| 2011 | 309,156 |
| 2012 | 425,126 |
| 2013 | 207,108 |
| 2014 | 213,444 |
| 2015 | 73,913 |
| 2016 | 20,662 |
| 2017 | 27,835 |
| 2018 | 68,535 |
| 2019 | 43,276 |
| 2020 | 10,500 |

LEMIS data may follow CITES export source code definitions in distinguishing between animals in trade that are designated captive-bred (animals bred in captivity in accordance with CITES Res. Conf. 10.16 (Rev.) and animals that are designated captive-born (animals born in captivity (F1 or subsequent generations) that do not fulfil the definition of “bred in captivity” in Res. Conf. 10.16 (Rev.) (Association of Fish and Wildlife Agencies, no date). More specifically the former designation requires: a second generation produced in captivity, with no or minimal involvement of initially gravid females in any founder breeding stock collected from the wild, and minimal addition of wild-collected animals to existing stock.

A study by Sung et al. (2021) on the illegal turtle trade on Hong Kong-based online platforms and physical markets found 16 specimens of *A. ferox* offered for sale at the “Goldfish Market”, i.e., the largest pet market in Hong Kong SAR, and no specimens advertised for sale on either social media or other online platforms during a 2017–2018 survey.

Reports of *A. ferox* being poached in Florida to be sold to Asian markets have regularly been reported over the years. For instance, from 2018–2019, a total of 4,000 turtles (including *A. ferox*) were illegally taken for this purpose. There have been concerns regarding the illegal harvest of *A. spinifera* for commercial use occurring in Montana (Ruggles, in litt., 2022). Softshell turtles being laundered through businesses have been observed in Florida (Wildlife Official from the Florida Fish and Wildlife Conservation Commission, in litt., 2022). Illegal harvest of both juvenile and adult *A. spinifera* specimens has been confirmed in Canada.

WiTIS recorded a total of 230 specimens of *A. ferox* seized in 2007 (220) and 2011 (10). Three live specimens of *A. spinifera* were seized in 2013 (two individuals) and 2020 (one individual). An additional seizure of an estimated 600 specimens took place in 2019, although this particular seizure involved other species and it is not possible to infer how many of the seized individuals belonged to either *A. ferox* or *A. spinifera*.

Although the percentage of captive-born versus wild-sourced hatchlings in exports cannot be determined, it is very unlikely that adult turtles being exported are captive-born due to their size and the amount of space they would require, as well as their aggressive nature when in close quarters (Buhlmann, in litt., 2022). At present, state legislation in Florida prohibits collection and sale of specimens taken from the wild and therefore all exported native *Apalone* species from this region should be captive-bred (Wildlife Official from the Florida Fish and Wildlife Conservation Commission, in litt., 2022). However, in the neighbouring state of Georgia, no protections are in place (i.e., commercial harvest is allowed) and laundering across state lines is possible (Sawyer, in litt., 2022).

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

Due to turtle life history traits, harvest of specimens from the wild puts the three species at great risk, and even marginal overexploitation can have important negative impacts on their persistence. A study by Zimmer-Shaffer et al. (2014) determined that the rate of annual harvest did not result in population declines of *A. mutica* and *A. spinifera* in Missouri only when demographic rates of the two species (compiled from the literature) were at their maximum level, which is unlikely to occur in wild populations.

Softshell turtle species can be readily farmed, although *A. mutica* is likely the most difficult to rear in farm ponds. Regardless, freshwater turtle species from North America are easier to breed than most species native to Asia. This has led to demand for *Apalone* specimens from turtle breeding farms in the United States where wild-caught specimens are thought to be of superior genetic stock. Not only does this place harvest pressure on wild *Apalone* species, it also discourages the breeding of freshwater turtle species that are native to Asia, even further

eliminating any sort of “safety net” population for those species that are decimated in the wild. *Apalone ferox*, *A. spinifera* and *A. mutica* may be at great risk from overexploitation due to the “boom-and-bust” pattern that is usually seen within the global turtle trade. Normally, after more restrictive regulations on trade of certain species comes into force, less protected and similar species become targeted. Several Asian and African softshell turtle species were listed in CITES Appendix II at the 16th and 17th meetings of the Conference of the Parties to CITES, thus suggesting future increased levels of demand for *Apalone* specimens.

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 three species of softshell turtles outlined in the proposal look distinctly different from both African/Middle Eastern and Asian softshell turtles (Ruggles, in litt., 2022). Both groups of turtles (genera *Trionyx* and *Pelodiscus* respectively) are traded for their meat and commonly taken from the wild to meet demand despite the occurrence of breeding farms within Asia.

Apalone specimens may be confused with one other species from the Trionychidae family, i.e., the Pig-nosed Turtle *Carettochelys insculpta* (included in Appendix II since 2005). The species may be differentiated on the basis of (i) the two claws on the forelimbs (softshell turtles have three), (ii) a shorter snout, and (iii) its distribution range covering Australia, Indonesia, and Papua New Guinea.

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Additional Information

Threats

Aside from overexploitation for trade, additional impacts have been identified for each species. These include commercial take of adults (as accidental bycatch in seine and trotline fisheries), recreational fishing, roadkill of female adults when these are moving to or from nesting sites and predation by foxes, skunks, raccoons, bears, and fish crows, as well as moles, canids, fire ants, and fly larvae (van Dijk, 2011a). Populations of *A. mutica* are believed to be further impacted by water pollution (due to their high dependence on riverine habitats) and flooding events (van Dijk, 2011b).

According to the 2011 IUCN Red List assessment for *A. spinifera*, specific subspecies have been impacted by different factors. Hybridisation between *A. spinifera atra* and *A. spinifera emoryi* has been recognised as a primary threat due to populations of the latter entering the already occupied Cuatro Ciénegas basin from the Rio Grande through the Rio Nadadores and Rio Salado (van Dijk, 2011c). The Mexican populations of *A. spinifera emoryi* are also impacted by water diversion and decreasing groundwater levels from irrigation and groundwater pumping (van Dijk, 2011c).

Conservation, management and legislation

At the national level, states within the USA have their own regulations on trade of the species. For instance, legislation in Montana prohibits commercial harvest of the species, but regulations regarding consumption or pet ownership do not exist (Ostovar et al., 2021), whereas the state of Maryland prohibits the keeping of spiny softshell turtles as pets (Grange, in litt., 2022). The United States Food and Drug Administration prohibits turtles with a carapace length of less than four inches (10 cm) to be either sold, held for sale or offered for any other type of commercial or public distribution (with the exception of live turtles that are intended for export only). Turtle farming operations are regulated by state.

In Canada, it is illegal to capture, harass or kill any *Apalone* species. Moreover, they are considered as threatened under the Act Respecting Threatened or Vulnerable Species (R.S.Q. Chapter E-12.01, s.10) in Quebec and threatened under the Endangered Species Act (S.O. 2007, Chapter 6) in Ontario, where it is also afforded protection offered by the Fish and Wildlife Conservation Act (S.O. 1997, Chapter 41). According to Environment and Climate Change Canada (2018), one or more action plans will be completed for *A. spinifera* and posted on the SARA registry by December 2023.

Mexico has no legal instruments in place at the moment for either species. Since 1996, the Wildlife Management and Sustainable Use Units (UMAs) have enabled local landowners to benefit through sustainable use of their wildlife and this has led to the protection of critical habitat for species at risk, including turtles and tortoises.

All three species can be found throughout several protected areas, such as populations of *A. ferox* inhabiting the Wekiwa Springs State Park in Florida (established in 1970) and populations of *A. spinifera* occurring at the Missisquoi National Wildlife Refuge in Canada. However, estimates on the extent of occurrence of the three species in protected areas is currently unknown. Despite domestic control measures to protect the three species in place in the USA, these are likely inadequate in regulating harvest pressure and resources. For instance, expertise and time are needed to differentiate between hatchlings of *A. spinifera* versus *A. mutica*.

In the USA, management measures are currently determined at the state level. According to Ostovar et al. (2021), the management of these three species, as well as other chelonians, should not be based on the state- or country-wide level, but rather on the watershed level due to different river systems having different population abundance capacities.

Monitoring of the populations in the USA is tied to state required documentation of commercial use, which has historically been defective. For instance, only 35% of harvest permit owners in Arkansas reported their numbers in 2019. Therefore, science-led decisions on harvest limits are often impossible to make. In Canada, monitoring mostly focuses on the number of nests rather than the number of specimens themselves. At present, population monitoring efforts are not known to occur in Mexico, although isolated sampling in certain areas has been carried out.

Artificial propagation/captive breeding

Freshwater turtle farming represented a lucrative business throughout the south-eastern regions of the USA in the early 1990s. Since then, the total number of turtles produced from US-based farms has steadily decreased—albeit the business is still thriving in many areas. The decrease in numbers of softshell turtles from farming facilities is likely due to the establishment of self-sustaining turtle farms in Asia.

At present, Florida registers over 50 certified breeding facilities. In Mexico, there are ten active captive breeding centres for *Apalone spinifera* (three captive breeding Wildlife Management and Sustainable Use Units (UMAs) and seven Wildlife Farms and Facilities (PIMVS)). Canada does not at present have any licensed large-scale turtle farming facilities.

Implementation challenges (including similar species)

The three species can be distinguished from each other by the following morphological features.

- *Apalone ferox*: adults are dull in colour with a grey, brown, or olive coloured carapace, while juveniles are relatively darker with yellow stripes on the head that are lost with age.
- *Apalone mutica*: characterised by a smooth anterior end of their carapace; hatchlings, juveniles, and most adult males have a tan carapace with circular spots, whereas adult females are tanner in colour with less noticeable spots.
- *Apalone spinifera*: intermediate in size, the anterior margin of the carapace has pointed and small spines.

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Transfer of Leith's Softshell Turtle *Nilssonia leithii* from Appendix II to Appendix I

Proponent: India

Summary: Leith's Softshell Turtle *Nilssonia leithii* is a large freshwater turtle endemic to Peninsular India, where it occurs in east–west flowing rivers and large reservoirs. The species was assessed as Critically Endangered on the IUCN Red List in 2018 on the basis of a very large (>90%) estimated population decline over the past 30 years and a projected continuing decline. *Nilssonia leithii* was included in Appendix II in 2013. It is one of five species of softshell turtle in the genus *Nilssonia*, all included in the appendices.

No current population estimates are available. Formerly relatively widespread in India from the Ganges Basin and Andhra Pradesh south to Karnataka and Tamil Nadu, *N. leithii* populations are now only known to occur with certainty in the Kali River (Karnataka) and the Manjira and Shivaram Wildlife Sanctuaries in Telengana.

The species has been harvested for its meat for both Indian and international food markets. However, the largely domestic trade documented in the 1980s and 1990s appears to have declined as the species has become scarcer, and there are no recent records of international trade.

National legislation prohibits the export for commercial purposes of wild specimens, and authorisation is required for any non-commercial use. The species is the subject of surveys and other conservation measures undertaken by Madras Crocodile Bank Trust and the Turtle Survival Alliance.

Analysis: *Nilssonia leithii* populations are thought to have undergone a marked decline over the past 30 years which is likely to be continuing. Known remaining populations are fragmented and believed small. The species appears to meet the biological criteria for inclusion in Appendix I in Res. Conf. 9.24 (Rev CoP17). There is little indication of current harvest, apparently because population levels are now too low to make hunting worthwhile, and no recent records of international trade exist. Export of wild specimens for commercial purposes is banned. It appears that the species is not currently affected by international trade, however any level of international demand is likely to be detrimental to the species. An Appendix I listing would reflect the national regulations in place for this 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

Nilssonia leithii (Gray, 1872).

Synonyms: *Aspideretes leithii* (Gray, 1872); *Aspilus gataghol* (Gray, 1872); *Trionyx javanicus* (Gray, 1830); *Trionyx leithii* (Gray, 1871); *Trionyx sulcifrons* (Annandale, 1915); *Nilssonia leithii* (Praschag et al., 2007).

Range

India

IUCN Global Category

Critically Endangered A2cd+4cd (assessed 2018, ver 3.1)

Biological criteria for inclusion in Appendix I

A) Small wild population

No quantitative data regarding the population size of *Nilssonia leithii* are available since the species is extremely rare, even within protected areas. *Field surveys have shown that populations in the southern part of its range are small, fragmented and scattered, and the overall population trend is decreasing (Praschag et al., 2021). By 2011, no*

viable population was known, and overall there have only been a few reliable records of the species over the last two decades with the exception of the states of Karnataka and Telangana. *The Turtle Survival Alliance (TSA) encountered a total of four and two specimens over a month-long survey along the Kali River in Karnataka in 2012 and 2013 respectively (Singh, in litt., 2022). Other individuals have been recorded a couple of times from the Moyar River approximately 10 years ago, and more recently in (Pune) Maharashtra, and Karnataka (Singh, in litt., 2022).*

Surveys, potentially engaging local communities, are urgently needed to determine population size and density across its range, as well as to monitor subpopulations of the species (Praschag et al., 2021).

B) Restricted area of distribution

*Endemic to Peninsular India, the species' historical range extends from south of the Ganges Basin and Andhra Pradesh to Karnataka and Tamil Nadu (Praschag et al., 2021). Over the years, *N. leithii* has disappeared from many historical sites (including Odisha and Chhattisgarh, where the last specimens were recorded in 1984 and 1991 respectively), and populations are now only known with certainty from the Kali River (Karnataka), the Manjira Wildlife Sanctuary, and the Siwaram Wildlife Sanctuary in Telangana (Praschag et al., 2021).*

Currently the available habitat of *N. leithii* is rapidly shrinking due to sand mining, pollution and changes in river morphology from hydrological projects. The range of the species has declined from most of the large river systems and reservoirs of the Deccan Peninsula, and the species is now geographically limited to the states of Karnataka, arashtra, and Tamil Nadu.

C) Decline in number of wild individuals

*Data collected from the participants of the Lucknow Red List Workshop from 2005 inferred that the overall population of *N. leithii* declined more than 90% over the preceding 30 years due to intensive exploitation for illegal trade and habitat degradation (Praschag et al., 2021). The population is believed to still be subject to a large-scale decline. Moreover, the species appears to be locally extinct in parts of its historical range and is now restricted to Karnataka, Maharashtra, and Tamil Nadu.*

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

The IUCN Red List assessment noted the harvest of this species for both the Indian and international meat trade was of concern.

*In the 1980s and 1990s, the species was reportedly widely exploited for consumption (Praschag et al., 2021). According to Kumar (2004), *N. leithii* was sold in local markets and toddy (liquor made from coconut) shops in the region of Kerala for INR100–INR300 (USD1.2–USD3.8) depending on the specimen's size. Research in the 1990s indicated that extensive collection efforts appeared to focus on the southern areas of the species' range, with trade directed to northern and north-eastern India, and beyond (Praschag et al., 2021). Later surveys highlighted use of calipee (dried, processed carapacial cartilage) apparently for traditional medicine (Das et al., 2014). In 2011, interviews with local hunters revealed that the species was no longer pursued at the time due to low populations and challenges with finding individuals (Praschag et al., 2021).*

*However, there is no confirmed record of international trade of *N. leithii* as of yet (Singh, in litt., 2022). Neither the CITES Trade Database, the LEMIS database, nor TRAFFIC's WiTIS contain records of either legal or illegal trade of the species.*

The recently increasing illegal trade of calipee of large freshwater turtles for traditional medicine may have an impact on the populations of *N. leithii*. *However, since the species is rare and restricted in range, it is estimated to contribute to less than 5% of the total illicit calipee trade in India (Singh, in litt., 2022). Indeed, the commercial hunting of the species for its calipee would doubtless not be profitable given *N. leithii* occurs only at low population densities anywhere within its range (Praschag, in litt., 2022). Species specific identification of calipee would require genetic analysis, although no such studies have been undertaken (Praschag, in litt., 2022).*

Additional information

Threats

Sand mining, pollution and changes in river morphology from hydrological projects are seriously damaging the habitat of *N. leithii* and are thus considered primary threats to its survival. The Supporting Statement notes that large-scale exploitation for the illegal meat trade at both the national and international levels occurs.

Conservation, management and legislation

Nilssononia leithii has been listed in CITES Appendix II since 2013 and is protected under Schedule IV of India's Wild Life (Protection) Act (1972), which prohibits hunting and collection of the species (Section 9 of the Act). Commercial use of the species requires authorisation (Section 44 of the Act) and is prohibited for wild populations (Section 48). The species is expected to be uplisted to Schedule I of the proposed Wild Life

(Protection) Act. Furthermore, *N. leithii* is culturally protected due to some populations being worshipped in temple ponds across the country.

Over time, *N. leithii* has been recorded from many Protected Areas across its range, including Nagarjunsagar National Park (3,568 km²), Sathymangalam Tiger Reserve (Tamil Nadu, 1,409 km²), Manjira Wildlife Sanctuary (Andhra Pradesh, 20 km²), Kudremukh National Park (600 km²), Bheemeshwari Wildlife Reserve/Cauvery Wildlife Sanctuary, Tungabhadra River Sanctuary, Dandeli Anshi Tiger Reserve and Sharavati Wildlife Sanctuary (Karnataka, 413 km²), Mudumalai Wildlife Sanctuary (Tamil Nadu, 321 km²), and Shivaram Wildlife Sanctuary (AP, 30 km²), and might occur in the Cauvery Protected Area (527 km²).

The Madras Crocodile Bank Trust and Centre for Herpetology and the TSA are currently conducting species surveys, and TSA intends to develop captive assurance colonies at regional zoos within the species' range.

Captive breeding

At present, no individuals of this species are held in private collections around the world (Praschag, in litt., 2022). There is one individual in captivity at the Madras Crocodile Bank Trust (Singh, in litt., 2022). Other specimens are found in captivity in regional zoos across India (e.g., one adult female at Nehru Zoological Park, Hyderabad). At present there are no captive breeding populations in India.

Implementation challenges (including similar species)

*Nilssonina leithii does not occur throughout the core range of other softshell species, such as its congeners *N. gangetica* and *N. hurum* (Praschag, in litt., 2022). Both juveniles and adults can be readily distinguished from *N. gangetica* and *N. hurum* through genetic analysis (Praschag, in litt., 2022).*

Potential benefit(s) for trade regulation of a transfer from Appendix II to I

Under India's regulations, no export of wild sourced specimens of species in Appendix I or II are permitted. Therefore, an Appendix I listing would reflect national legislation.

References

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- Praschag, P. (2022). *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.
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- Singh, S. (2022). *In litt.* to the IUCN/TRAFFIC Analyses Team, Cambridge, UK.

Inclusion of glass frogs Centrolenidae in Appendix II

Proponents: Argentina, Brazil, Costa Rica, Côte d'Ivoire, Dominican Republic, Ecuador, El Salvador, Gabon, Guinea, Niger, Panama, Peru, Togo, United States of America

Summary: Glass frogs, the collective term for species within the family Centrolenidae, are charismatic nocturnal frogs with large eyes and transparent skin inhabiting lowland and montane moist tropical forests. Their taxonomy is in a state of flux: currently, there are 12 genera in the Centrolenidae family with around 158 species widely distributed in 19 countries across Central and South America. The whole family Centrolenidae is proposed for inclusion in Appendix II. Twelve lead species (*Cochranella euknemos*, *Cochranella granulosa*, *Espadarana prosoblepon*, *Hyalinobatrachium aureoguttatum*, *Hyalinobatrachium fleischmanni*, *Hyalinobatrachium valerioi*, *Hyalinobatrachium iaspidiense*, *Hyalinobatrachium mondolfii*, *Sachatamia albomaculata*, *Sachatamia ilex*, *Teratohyla pulverata*, and *Teratohyla spinosa*) from five genera have been identified in trade and are proposed under criterion B in Annex 2a of Res. Conf. 9.24 (Res. Conf. CoP17), with the remaining species within the family proposed for inclusion as lookalikes under criterion A of Annex 2b.

There are no quantitative population data available for any species in the family. Of the 12 lead species, 10 are classified as Least Concern on the IUCN Red List (assessed 2019). *Hyalinobatrachium mondolfii* and *H. iaspidiense* were assessed in 2004 as Least Concern and Data Deficient respectively, but both will be classified as Least Concern in the IUCN Red List update in December 2022. All species listed as Least Concern are stated to have a wide distribution, and are therefore presumed to have large populations, with *H. iaspidiense* stated to be “not uncommon”. Five of the 12 species are assessed as having stable populations, five decreasing populations, and two with unknown population trends. None of the Red List assessments identify trade as a threat and only one (for *H. valerioi*) mentions the species is traded. The other species of glass frog include 10 species assessed as Critically Endangered, 34 as Endangered, 20 as Vulnerable, 11 as Near Threatened, 26 as Data Deficient, and 44 as Least Concern, while 11 species have not yet been assessed.

Glass frogs are in demand for international trade mainly as pets. Prices vary according to species, location and market type, and are reported to range from USD25–USD950. Global trade data are not available for any glass frog species. Four species (*Cochranella granulosa*, *Hyalinobatrachium fleischmanni*, *H. valerioi*, and *Teratohyla pulverata*) have records of live individuals being imported into the USA for commercial purposes. In total, around 9,200 live individuals of these four species were imported between 2010 to 2020. Almost all imports were reported since 2017. This may indicate increasing supply or demand but could also reflect improved reporting. Almost all (98%) of the trade was reported to be in captive-bred specimens, with records of around 200 wild-sourced individuals, all of which were *Hyalinobatrachium fleischmanni*. This species accounted for 84% of all reported imports, almost all of which originated from Nicaragua. This species has been assessed as having a stable population and is described by one expert as locally common. It was assessed as Least Concern by the IUCN in 2019 on account of its wide distribution and presumed large population, and its due to be reclassified as Least Concern in December 2022. No evidence for any other wild-sourced specimens of glass frog species was recorded in US trade data.

There is additional evidence of trade for all 12 lead species from a combination of seizures, surveys of online advertisements, and records from physical markets in Tokyo and Europe. Only one other species in the family, *N. grandisone*, is recorded as offered for sale online in one survey but the volume of individuals for sale and frequency of advertisements are not stated. This species was classified on the IUCN Red List as Least Concern in 2004 and is due to be reclassified as Least Concern in December 2022. Quantitative data are available for seizures reported from the proponents' research, with a total of 95 individuals from species including *Hyalinobatrachium valerioi*, *Sachatamia ilex*, and *Teratohyla spinosa*. The only other quantitative data are from a TRAFFIC report,

where surveyors observed 15 live individuals of species including *H. fleischmanni*, *H. valeroi*, *Teratohyla pulverata*, and *Cochranella granulosa* offered for sale at a reptile fair in Tokyo in 2020.

For most online advertisements the count of individuals offered for sale is not stated and in most cases it is difficult to clarify if the animals are captive-bred or wild-sourced. A survey by UNEP-WCMC identified a total of 28 advertisements, 12 of them for *Hyalinobatrachium fleischmanni*. The proponents state that they have found 75 advertisements in a preliminary analysis of species for sale online. A brief online survey for this analysis identified four pet stores hosted in the USA with glass frogs for sale, with all only offering *H. fleischmanni* and half of the stores explicitly stating the species was captive-bred. *Hyalinobatrachium fleischmanni* is the species most frequently offered for sale online. It is known that two other glass frog species can be bred in captivity (*H. valeroi* and *Sachatamia albomaculata*) and there is evidence of commercial captive breeding of glass frog species for export in Canada (*H. valeroi*), Nicaragua (species unknown), and Ecuador (*H. aureoguttatum*).

There is no evidence of large volumes involved for some of the 12 species identified as being in trade; for example, *Hyalinobatrachium iaspidiense* and *H. mondolfi* have been found advertised online in one EU survey, with unknown volumes offered for sale. Only six of the 12 species have any form of quantitative trade data to indicate potential volumes. For two of these, the only available data are from seizures with relatively low numbers reported seized: 63 individuals for *Sachatamia illex* and 14 for *Teratohyla spinosa*.

In most range States, harvest of glass frogs from the wild for commercial trade is currently prohibited or requires a permit. Glass frog experts state that species within the Centrolenidae family cannot be easily distinguished based on morphological features, even by specialists.

Analysis: There are few quantitative data available on populations of glass frog species and no population estimates for the 12 species identified by the proponents as being in trade. Recent and forthcoming IUCN Red List assessments classify all these 12 species as Least Concern, with large population distributions. None of the Red List assessments identifies trade as a threat: identified threats are habitat loss, fragmentation, and disease.

There are limited data for trade volumes and no information on the impact of wild harvest on species' populations. US import data at the species level are only available for four out of the 12 species proposed under criterion B in Annex 2a of Res. Conf. 9.24 (Rev. CoP17), with the majority reported from captive sources. *Hyalinobatrachium fleischmanni* is the species most frequently reported in trade from all available sources and has the greatest volume of imports reported by the USA (averaging almost 2,000 a year for the period 2017–2020 with almost no trade from 2010 until that time). Almost all of those in trade were reported to be captive-bred. The 2019 Red List assessment for this species stated that it had a stable population with a wide distribution. A reassessment of the species for the Red List update in December 2022 will reaffirm the previous (2019) assessment of Least Concern. There is little information on trade in this species to other known markets for glass frogs in Europe and Asia. There is minimal evidence of large trade volumes for the other 11 species and minimal evidence of any trade in the other species in the family.

Based on available information it does not appear that any of the 12 lead species identified in the proposal or any other species in the family Centrolenidae are likely to meet the criteria for inclusion in Appendix II set out in Res. Conf. 9.24 (Rev. CoP17). Had any of the species been considered to meet the criteria in Annex 2a other species would meet the criteria in Annex 2aB as it appears that it is difficult to differentiate between species of glass frogs.

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 utilises the CITES Standard Reference for amphibians of Frost (2015), with reference to the online database *Amphibians of The World*. Currently 158 species have been described, of which two were described during 2022. These recognised species of glass frogs are grouped into 12 genera in the Family Centrolenidae, with the genera and number of species as follows: *Celsiella* (2), *Centrolene* (24), *Centrolene incertae sedis* (6), *Chimerella* (2), *Cochranella* (8), *Cochranella incertae sedis* (7), *Espadarana* (5), *Hyalinobatrachium* (33), *Ikakogi* (2), *Nymphargus* (41), *Rulyrana* (6), *Sachatamia* (5), *Teratohyla* (5), and *Vitreorana* (10).

The taxonomy of glass frogs (family Centrolenidae) continues to change with the discovery and description of new species and revisions of phylogenetic hypotheses creating new combinations of genus and species names. The intent of this proposal is explicitly to include in Appendix II in the future any as yet undiscovered species of the family Centrolenidae through the regular process of updating the nomenclature of species in the CITES Appendices as directed by Resolution Conf. 12.11 (Rev. CoP18) *Standard nomenclature*.

Binomial names, synonyms and common names for the 12 species of glass frogs the proponents have listed for inclusion in Appendix II in accordance with criterion B of Annex 2a of Res. Conf. 9.24 (Rev. CoP17) are listed in Table 1.

Table 1. Binomial names, synonyms and common names for the 12 lead species, proposed for Appendix II under criterion B of Annex 2a of Res. Conf. 9.24 (Rev. CoP17).

| Binomial name | Binomial name synonyms* | Common name* |
|--|--|----------------------------|
| <i>Cochranella euknemos</i> | <i>Centrolenella euknemos</i> | San Jose Cochran Frog |
| <i>Cochranella granulosa</i> | <i>Centrolenella granulosa</i> | Grainy Cochran Frog |
| <i>Espadarana prosoblepon</i> | <i>Hyla prosoblepon</i> , <i>Centrolene prosoblepon</i> | Nicaragua Giant Glass Frog |
| <i>Hyalinobatrachium aureoguttatum</i> | <i>Centrolenella aureoguttata</i> | Atrato Glass Frog |
| <i>Hyalinobatrachium fleischmanni</i> | <i>Cochranella decorata</i> , <i>Cochranella millepunctata</i> , <i>Hylella fleischmanni</i> , <i>Hylella chrysops</i> | Fleischmann's Glass Frog |
| <i>Hyalinobatrachium valerioi</i> | <i>Centrolene valerioi</i> , <i>Centrolenella valerioi</i> , <i>Cochranella reticulata</i> , <i>Cochranella valerioi</i> | La Palma Glass Frog |
| <i>Hyalinobatrachium iaspidiense</i> | None listed | Yuruani Glass Frog |
| <i>Hyalinobatrachium mondolfii</i> | None listed | None listed |
| <i>Sachatamia albomaculata</i> | <i>Centrolenella albomaculata</i> , <i>Centrolenella albomarginata</i> , <i>Cochranella albomaculata</i> | White-spotted Cochran Frog |
| <i>Sachatamia ilex</i> | <i>Centrolenella ilex</i> , <i>centrolene ilex</i> | Sachatamia ilex |
| <i>Teratohyla pulverata</i> | <i>Centrolenella petersi</i> , <i>Cochranella petersi</i> , <i>Cochranella pulverata</i> , <i>Hyalinobatrachium pulveratum</i> , <i>Hyla pulverata</i> | Chiriqui Glass Frog |
| <i>Teratohyla spinosa</i> | <i>Cochranella spinosa</i> | Spiny Cochran Frog |

*Synonyms and common names are sourced from IUCN Red List assessments

IUCN Global Category

As of June 2022, the IUCN Red List assessed 10 species of glass frog as Critically Endangered, 34 as Endangered, 20 as Vulnerable, 11 as Near Threatened, 27 as Data Deficient, and 55 as Least Concern, with 11 not evaluated (Table 2) (IUCN Red List, in litt., 2022). The taxonomy utilised by the IUCN Red List may differ to the taxonomy utilised by the proponents, with the number of species recognised by each differing. The 12 species proposed for inclusion in Appendix II according to Criterion B in Annex 2a of Res. Conf. 9.24 (Rev. CoP17) are highlighted in bold in Table 1. Of these 12, 10 were classified as Least Concern in 2020, and one of the remainder was classified as Least Concern in 2004 the other as Data Deficient. Both the latter two species are due to be classified as Least Concern later in 2022, with *H. fleischmanni* additionally due to be reclassified as Least Concern in 2022.

Table 2. Global IUCN Red List Status of glass frog species June 2022, inclusive of changes made in the July update to Global Status. The 12 lead species, proposed for inclusion in Appendix II based on criterion B in Annex 2a of Resolution Conf. 9.24 (Rev. CoP17), are highlighted in bold.

| Critically Endangered | Endangered | Vulnerable | Near Threatened | Least Concern | Not evaluated | Data Deficient |
|-------------------------|-----------------------------|---------------------------|-----------------|---------------|---------------|----------------|
| <i>Celsiella</i> | | | | | | |
| | <i>C. vozmediano</i> (2020) | <i>C. revocata</i> (2020) | | | | |

| Critically Endangered | Endangered | Vulnerable | Near Threatened | Least Concern | Not evaluated | Data Deficient |
|---|--|---|--|--|---|---|
| Centrolene | | | | | | |
| <i>C. geckoidea</i> (2019) <i>C. gemmatum</i> (2004)* <i>C. pipilata</i> (2019) | <i>C. altitudinalis</i> (2020) <i>C. ballux</i> (2019) <i>C. charapita</i> (2022) <i>C. condor</i> (2022) <i>C. hesperium</i> (2018) <i>C. huilense</i> (2019) <i>C. lynchi</i> (2022)* <i>C. medemi</i> (2019) <i>C. petrophilum</i> (2017) <i>C. sanchezi</i> (2022) <i>C. scirtetes</i> (2019)* <i>C. solitaria</i> (2019) | <i>C. buckleyi</i> (2010)** <i>C. heloderma</i> (2019) <i>C. quindianum</i> (2017) <i>C. sabini</i> (2017) | <i>C. antioquiense</i> (2017) | <i>C. daidalea</i> (2020) <i>C. hybrida</i> (2019) <i>C. notosticta</i> (2020) <i>C. peristictum</i> (2019) <i>C. robledo</i> (2014) <i>C. savagei</i> (2017) <i>C. venezuelense</i> (2020) | | <i>C. acanthidiocephalum</i> (2019) <i>C. azulae</i> (2019) <i>C. bacatum</i> (2010)* <i>C. guanacarum</i> (2019)* <i>C. lema</i> (2004)* <i>C. lemniscatum</i> (2019) <i>C. muelleri</i> (2019) <i>C. ocellifera</i> (2004)* <i>C. paezorum</i> (2019) |
| Chimerella | | | | | | |
| | | | | <i>C. mariaelenae</i> (2018) | | <i>C. corleone</i> (2018) |
| Cochranella | | | | | | |
| <i>C. euhystrix</i> (2018) | <i>C. megistra</i> (2017)* | <i>C. duidaeana</i> (2020) <i>C. litoralis</i> (2019) <i>C. riveroi</i> (2020) <i>C. xanthocheridia</i> (2017) | <i>C. mache</i> (2020) <i>C. ramirezi</i> (2019) | <i>C. erminea</i> (2018) <i>C. euknemos</i> (2020) <i>C. geijskesi</i> (2018) <i>Cochranella granulosa</i> (2020) <i>C. guayasamini</i> (2018) <i>C. nola</i> (2020) <i>C. resplendens</i> (2018) | | <i>C. phryxa</i> (2020) |
| Espadarana | | | | | | |
| | | | | <i>E. andina</i> (2020) <i>E. audax</i> (2018) <i>E. callistomma</i> (2019) <i>E. durrellorum</i> (2019) <i>E. prosoblepon</i> (2020) | | |
| Hyalinobatrachium | | | | | | |
| | <i>H. anachoretus</i> (2018) <i>H. durante</i> (2020) <i>H. esmeralda</i> (2017) <i>H. guairarepanense</i> (2020) <i>Hyalinobatrachium orocostale</i> (2020) | <i>H. orientale</i> (2020) | <i>H. fragile</i> (2020) <i>H. pallidum</i> (2020) <i>H. pellucidum</i> (2014)** <i>H. tatayoi</i> (2020)** | <i>H. aureoguttatum</i> (2020) <i>H. bergeri</i> (2004)** <i>H. chirripoi</i> (2008)** <i>H. colymbiphyllum</i> (2020) <i>H. crurifasciatum</i> (2004)* <i>H. diana</i> (2020) <i>H. fleischmanni</i> (2020)** <i>Hyalinobatrachium ibama</i> (2020) <i>H. kawense</i> (2018) <i>H. mondolfii</i> (2004)** | <i>H. adespinosai</i> ** <i>H. cappellei</i> ** <i>H. carlesvilai</i> ** <i>H. mashpi</i> <i>H. muiraquitana</i> ** <i>H. nouns</i> <i>H. viridissimum</i> * <i>H. yaku</i> ** | <i>H. eccentricum</i> (2004)* <i>H. iaspidiense</i> (2004)** <i>H. igniocus</i> (2004)* <i>H. mesai</i> (2020) |

| Critically Endangered | Endangered | Vulnerable | Near Threatened | Least Concern | Not evaluated | Data Deficient |
|---|--|--|--|---|---|---|
| | | | | <i>H. munozorum</i> (2010)** <i>H. nouraguense</i> (2010)* <i>H. ruedai</i> (2010)* <i>H. talamancae</i> (2020) <i>H. taylori</i> (2020) <i>H. tricolor</i> (2018) <i>H. valerioi</i> (2020) <i>H. vireovittatum</i> (2020) | | |
| <i>Ikakogi</i> | | | | | | |
| | | <i>I. tayrona</i> (2017) | | | <i>I. ispacue</i> | |
| <i>Nymphargus</i> | | | | | | |
| <i>N. anomalus</i> (2004)** <i>N. armatus</i> (2017) <i>N. laurae</i> (2008)** <i>N. manduriacu</i> (2020) <i>N. mixomaculatus</i> (2018) <i>N. truebae</i> (2017) | <i>N. balionotus</i> (2020) <i>N. bejaranoi</i> (2020) <i>N. caucanus</i> (2019) <i>N. colomai</i> (2022)* <i>N. cristinae</i> (2019) <i>N. luminosus</i> (2017) <i>N. luteopunctatus</i> (2019) <i>N. megacheirus</i> (2004)** <i>N. megista</i> (2017)* <i>N. phenax</i> (2018) <i>N. pluvialis</i> (2020) <i>N. siren</i> (2022) | <i>N. cochranae</i> (2010)** <i>N. garciae</i> (2019) <i>N. prasinus</i> (2017) <i>N. rosada</i> (2017) <i>N. ruizi</i> (2017) | <i>N. chami</i> (2019) <i>N. spilotos</i> (2019) | <i>N. grandisonae</i> (2004)** <i>N. griffithsi</i> (2018) <i>N. ignotus</i> (2020) <i>N. mariae</i> (2020) <i>N. oreonympha</i> (2019) <i>N. posadae</i> (2018) | | <i>N. buenaventura</i> (2008)** <i>N. cariticommatu</i> s (2004)** <i>N. chancas</i> (2013)** <i>N. humboldti</i> (2022) <i>N. nephelophila</i> (2019) <i>N. ocellatus</i> (2018) <i>N. vicerteruedai</i> (2019) <i>N. wileyi</i> (2008)** |
| <i>Rulyrana</i> | | | | | | |
| | <i>R. saxiscandens</i> (2017) | <i>R. adiazeta</i> (2017) | <i>R. spiculata</i> (2020) <i>Rulyrana susatamai</i> (2017) | <i>R. flavopunctata</i> (2004)** | | <i>R. mcdiarmidi</i> (2009)** |
| <i>Sachatamia</i> | | | | | | |
| | <i>S. electrops</i> (2019) | <i>S. punctulata</i> (2017) | | <i>S. albomaculata</i> (2020) <i>S. ilex</i> (2020) <i>S. orejuela</i> (2019) | | |
| <i>Teratohyla</i> | | | | | | |
| | | | | <i>T. ameliae</i> (2019) <i>T. midas</i> (2004) <i>T. pulverata</i> (2020) <i>T. spinosa</i> (2020) | | <i>T. adenocheira</i> (2006)** |
| <i>Vitreorana</i> | | | | | | |
| | <i>V. castroviejoi</i> (2020) | <i>V. antisthenesi</i> (2020) <i>V. helenae</i> (2020) | | <i>V. eurygnatha</i> (2010)** <i>V. gorzulae</i> (2020) <i>V. uranoscopa</i> (2010)** | <i>V. baliomma</i> ** <i>V. franciscana</i> ** | <i>V. parvula</i> (2004)** <i>V. ritae</i> (2004)** |

* Taxonomy will be changed in July's 2022 IUCN Red List update

** New IUCN Red List assessment submitted for December 2022

Range and population

Members of the family Centrolenidae are widely distributed in 19 range States in Central and South America. The highest numbers of species are found in Colombia (74), Ecuador (51), Peru (32), and Venezuela (27) but species within the family are also extant in Argentina, Belize, Brazil, Costa Rica, El Salvador, French Guiana, Guatemala, Guyana, Honduras, Mexico, Nicaragua, Panama, Plurinational State of Bolivia, Suriname, and Trinidad and Tobago. Data on the population size of glass frogs are very limited with one expert reporting that no population size estimates exist for any species of Centrolenidae (Cisneros-Heredia, via IUCN SSC Amphibian Specialist Group., in litt., 2022). Overall, of 153 species with assessments on the IUCN Red List in August 2022, most (46%) have decreasing populations, 18% stable and the remaining unknown.

Of the 12 species listed for inclusion in Appendix II for meeting criterion B in Annex 2a of Res. Conf. 9.24 (Rev. CoP17)—the only 12 Centrolenidae species for which evidence of trade is available—quantitative population size estimates are not available in recent IUCN Red List assessments. For all 11 species listed as Least Concern, this categorisation is in part due to their large distributions and presumed large populations (IUCN, 2022). *H. iaspidiense* (Data Deficient, 2004) is said to be not uncommon (IUCN, 2022). Eight species are considered common in at least one country in their range, with *S. albomaculata* and *E. prosoblepon* said to be common across their range. *H. valerioi* is the only species not described as common in at least one range State but is still assessed as Least Concern due to its wide distribution across four range States.

Table 3 summarises the range and population trends of the 12 species proposed for meeting criterion B in Annex 2a of Resolution Conf. 9.24 (Rev. CoP17).

Table 3. Range and Population trend for the 12 lead species, proposed for inclusion in Appendix II for meeting criterion B in Annex 2a of Resolution Conf. 9.24 (Rev. CoP17). All species are assessed against Versions 3.1 of the IUCN Red List categories and criteria in 2019 unless stated otherwise (IUCN Red List, 2022).

| Species | Population trend** | Range |
|--|--------------------|---|
| <i>Cochranella euknemos</i> | Decreasing | Colombia, Costa Rica, Panama |
| <i>Cochranella granulosa</i> | Decreasing | Costa Rica; Honduras; Nicaragua; Panama, maybe Colombia |
| <i>Espadarana prosoblepon</i> | Stable | Colombia; Costa Rica; Ecuador; Honduras; Nicaragua; Panama |
| <i>Hyalinobatrachium aureoguttatum</i> | Stable | Colombia, Ecuador, Panama |
| <i>Hyalinobatrachium fleischmanni</i> | Stable | Belize; Colombia; Costa Rica; El Salvador; Guatemala; Guyana; Honduras; Mexico; Nicaragua; Panama; Suriname |
| <i>Hyalinobatrachium valerioi</i> | Decreasing | Colombia; Costa Rica; Ecuador; Panama |
| <i>Hyalinobatrachium iaspidiense</i> (DD, 2004)* | Unknown | Venezuela |
| <i>Hyalinobatrachium mondolfii</i> (LC, 2004)* | Stable | Venezuela |
| <i>Sachatamia albomaculata</i> | Decreasing | Colombia; Costa Rica; Ecuador; Honduras; Nicaragua; Panama |
| <i>Sachatamia ilex</i> | Decreasing | Colombia; Costa Rica; Ecuador; Nicaragua; Panama |
| <i>Teratohyla pulverata</i> | Unknown | Colombia; Costa Rica; Ecuador; Honduras; Nicaragua; Panama |
| <i>Teratohyla spinosa</i> | Stable | Colombia; Costa Rica; Ecuador; Honduras; Nicaragua; Panama |

* Due to be classified as Least Concern in December 2022 IUCN Red List assessment

** Inferred from habitat loss (Angulo, in litt., 2022)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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

International trade

Glass frogs are charismatic species with large eyes and transparent skin that are internationally traded commercially as live animals to supply the exotic pet trade. Exports are also recorded for scientific research, live and dead individuals and samples including blood, tissue, eggs, and extracts. Glass frogs are increasingly being promoted in the media as resembling "Kermit the Frog" (from The Muppet Show), and the demand for these

animals by the international pet trade is said to have increased. Most specimens are known to be traded in Europe, the USA and Canada, although a lack of trade data currently hinders our understanding of the main species involved in the trade, as well as the main trade routes.

In 2018, nine species were reported in trade. In preparing this proposal, trade records for 12 species of glass frogs were obtained from online advertisements or retrieved from trade data in the LEMIS database. Twelve species were successfully identified and confirmed in international trade.

Trade evidence from USA import data from the LEMIS database

Trade records from LEMIS presented by the proponents show a total of 15,329 imports of live glass frogs from 2010–2021; imports of live glass frogs were fairly constant between 2010 and 2016, but by 2021 they had increased dramatically. From 2016–2021 the number of live glass frogs imported into the USA increased by more than 44,000% from 13 in 2016 to over 5,700 in 2021. *The cause of this increase is not clear but could be due to increased demand or improved reporting.*

Further analysis of LEMIS data between 2010 and 2020 for records of cleared imports of live individuals for commercial purposes show that four out of the 12 species are reported in trade (Table 4). In total, around 9,200 live individuals from these four species were imported into the USA from 2010–2020; 2% were reported to be wild-sourced, the remainder reported as captive-bred. The majority of captive-bred individuals (99%) originated from Nicaragua, with the rest from Panama, and Suriname. LEMIS data were additionally searched for evidence of imports in all other species of glass frog, but no records of commercial imports in other species were found.

Table 4. A summary of trade evidence available for glass frog species. Sources: LEMIS 2010–2020, Proposal, TRAFFIC survey (Kitade and Wakao, 2022). Sources of evidence from online advertisements and physical markets: Proponents' own research = P; TRAFFIC Report (Kitade and Wakao, 2022) = T; UNEP-WCMC survey referenced by Proponents = U; TRAFFIC brief preliminary online search = O.

| Species (IUCN Red List status) | Cleared commercial imports of live individuals in LEMIS commodity, source code) | Origin country for cleared imports in LEMIS (%) | Evidence from seizures | Evidence from online advertisements (source*) | Evidence from physical markets seizures (source*) |
|---|--|--|-------------------------------|--|--|
| <i>Cochranella euknemos</i> (LC) | | | | Y (P) | |
| <i>Cochranella granulosa</i> (LC) | 1,297 (captive-bred) | Nicaragua (100) | | Y (P, U) | Y (T) |
| <i>Espadarana prosoblepon</i> (LC) | | | | Y (U) | |
| <i>Hyalinobatrachium aureoguttatum</i> (LC) | | | | Y (T,U) | |
| <i>Hyalinobatrachium fleischmanni</i> (LC) | 203 (wild) 7,488 (captive-bred) | Nicaragua (99) Panama, Suriname (1) | | Y (O, P, U) | Y (T) |
| <i>Hyalinobatrachium valerioi</i> (LC) | 50 (captive-bred) | Panama (100) | 18 | Y (P,U) | Y (P,T) |
| <i>Hyalinobatrachium iaspidiense</i> (DD) | | | | Y (U) | |

| Species (IUCN Red List status) | Cleared commercial imports of live individuals in LEMIS commodity, source code) | Origin country for cleared imports in LEMIS (%) | Evidence from seizures | Evidence from online advertisements (source*) | Evidence from physical markets seizures (source*) |
|---|---|--|------------------------|---|---|
| <i>Hyalinobatrachium mondolfii</i> (LC) | | | | Y (U) | |
| <i>Sachatamia albomaculata</i> (LC) | | | | Y (P,T) | |
| <i>Sachatamia ilex</i> (LC) | | | 63 | Y (P) | |
| <i>Teratohyla pulverata</i> (LC) | 150 (captive-bred) | Nicaragua (100) | | Y (U,P) | Y (P,T) |
| <i>Teratohyla spinosa</i> (LC) | | | 14 | Y (P) | |
| <i>Cochranella</i> spp. | 155 (captive-bred) 1 (wild) | Ecuador (100) | | | |
| <i>Hyalinobatrachium</i> spp. | 253 (captive-bred) 10 (rancher) 13 (wild) | Nicaragua (96) Ecuador (5) Suriname (1) | | | |
| <i>Centrolene</i> spp. | 46 (captive-bred) | Canada (49) Ecuador (43) Suriname (5) Germany (1) Costa Rica (1) | | | |

All commercial imports reported by the USA occurred from 2017 onwards, except for three live wild-sourced *H. fleischmanni* in 2015 and 140 live captive-bred *Hyalinobatrachium* spp. reported at the genus level from 2011–2016. For *C. granulosa*, 75% of imports occurred in 2020 and 25% in 2019. All *T. pulverata* imports also occurred in 2019. For *H. valerioi*, all imports occurred in 2017. For *H. fleischmanni*, no wild-sourced imports occurred after 2017 when 200 wild and 639 captive-bred specimens were imported from Nicaragua. In 2018, 2,137 specimens were imported, 2,110 in 2021, and 2,602 in 2020, all but 50 of which were reported as from Nicaragua. The LEMIS data are indicative of a recent increase in the volume of live individuals of *H. fleischmanni* and *C. granulosa* reported in commercial trade, with trade very low or minimal prior to 2017. Close to 100% of the individuals imported were reported to be captive-bred, although this information is difficult to verify, as is the true origin of the species. *H. fleischmanni* individuals account for 84% of all reported imports and all wild-sourced imports.

The proponents stated that at least 30% of live glass frogs imported for commercial purposes into the USA, according to LEMIS data, were identified only to the genus level or simply as a "non-CITES amphibian". Trade volumes for each of the 12 species proposed for inclusion in Appendix II for satisfying criterion B in Annex 2a of Resolution Conf. 9.24 (Rev. CoP17) could therefore be larger than reported here. A search for live commercial

imports of unidentified species within the 12 genera of glass frogs (*Celsiella*, *Centrolene*, *Chimeralla*, *Cochranella*, *Espadarana*, *Hyalinobatrachium*, *Ikakogi*, *Nymphargus*, *Rulyrana*, *Sachatamia*, *Teratohyla* and *Vitreorana*) into the USA from 2010–2020 found a relatively small volume of 478 records from three genera: *Cochranella* (156), *Hyalinobatrachium* (276) and *Centrolene* (46). Most individuals (85%) were reported as captive-bred, with 14 reported as wild-sourced and 10 as ranched.

Trade evidence from seizure data

As glass frogs are nationally protected in many range States, it is believed that many specimens appearing in international trade may have been obtained through illegal sources. There is some evidence of smuggling of glass frogs. In 2014, a German national was caught in Costa Rica trying to smuggle species including 18 *H. valerioi* and 20 *S. illex* individuals into Germany. A few days before the seizure, the smuggler's business partner had advertised several species of glass frogs on the website www.terraristik.com for sale at the Terraristika trade fair in Hamm, Germany; these included *S. illex*, *H. valerioi*, *S. albomaculata*, *C. granulosa*, *C. euknemos*, *T. spinosa*, and *T. pulverata*. In 2019, at Juan Santamaría International Airport in Costa Rica, a Russian citizen was arrested with more than 100 specimens in their hand luggage, including 43 *S. illex* and 14 *T. spinosa* specimens reported to be captured in the national territory.

Although not evidenced in seizure data, one species expert stated that legal trade of frogs in Ecuador has greatly facilitated illegal activities across the country, with reports from local people at several localities in northwestern Ecuador that collectors extract dozens of glass frogs per night. Glass frogs from other countries are also said to have been traded in Ecuador, with frogs apparently from Colombia traded at least twice in recent years (Cisneros-Heredia, via IUCN SSC Amphibian Specialist Group, in litt., 2022).

There are no records of seizures in any glass frog species in the WiTIS database or EU-TWIX.

Trade evidence from online advertisements and physical markets

There is evidence of online advertisements in Europe for some of the 12 species proposed for inclusion in Appendix II under the trade criteria. UNEP-WCMC conducted an online search from 21st–25th June 2021 to document the availability of glass frogs for sale within the European Union. A total of 82 online retailers, marketplaces, discussion forums and Facebook groups were surveyed of which 11 (13%) were found to contain advertisements for glass frogs. Overall, 28 advertisements featuring glass frogs were identified, listing six species (*C. granulosa*, *E. prosoblepon*, *H. aureoguttatum*, *H. fleischmanni*, *H. valerioi*, and *T. pulverata*) for sale by EU-based traders. *H. fleischmanni* was most frequently documented in advertisements (12), followed by *C. granulosa* and *E. prosoblepon*. Close to half (54%) of advertisements described captive-bred frogs, two listed individuals of *H. fleischmanni* as being of wild origin and the remaining 11 did not specify the source. In addition, three advertisements from sellers based in the UK were identified as selling species including *H. fleischmanni* and *H. valerioi* and *Nymphargus grandisonae* (classified as Least Concern in 2004 and due to be reclassified as Least Concern in December 2022). The species *H. iaspidiense* and *H. mondolfii* were also identified as being offered for sale in this survey but no further details from the advertisements for these species are given by the proponents.

A preliminary analysis by the proponents of the availability of glass frogs for sale online in recent years found more than 75 active listings, many of which offered more than one specimen for sale. There were over 100 sites with specimens available for sale. These listings were found primarily on sites based in the USA, Europe and Japan. Within Europe, the majority of offers for sale came from Spain, Germany, and the Netherlands. The country of origin was not indicated in the description of most of the specimens offered for sale, but some websites indicated that they were captive-bred. According to the proponents, in 2017, a Dutch trader advertised a large number of *T. spinosa* on the website www.terraristik.com, specifying that they were captive-bred specimens from Costa Rica. However, Costa Rican authorities confirmed that there were no registered breeding operations for this species and that any export of wild-taken specimens was illegal. In 2019, the same Dutch trader also offered a blue-green variety of *C. granulosa* from Costa Rica, as well as *H. fleischmanni*. In October 2017, the same platform offered glass frogs of the species *H. valerioi*. While several species of glass frogs were found for sale, listings of *H. fleischmanni* were more common than any other species in the family Centrolenidae. It was not possible from the information given to verify the volume of individuals offered for sale online in these advertisements in order to quantify the magnitude of trade.

The price of the specimens found in the proponents' survey ranged from USD25–USD150. Reports from Interpol Germany indicate that glass frogs can fetch between USD900 and USD955, making them among the most expensive species. *H. valerioi* sells for USD150 in the USA. In Spain, *H. valerioi* is advertised on the Internet at USD90 per specimen and *H. fleischmanni* at USD110 per pair.

A brief preliminary search for this analysis of glass frogs for sale online identified four online pet stores selling live individuals, all in the USA and all offering *H. fleischmanni* for sale. Two platforms stated the species were captive-bred. Quantities were not stated and prices ranged from USD39–USD99.

Glass frogs are also sold at European reptile and amphibian fairs, in particular Terraristika, which is held in Hamm (Germany) four times a year. The participating dealers come from Austria, Belgium, the Czech Republic, Germany, the Netherlands, Poland, Spain, and the UK. Prices of glass frogs vary between around USD45 and USD350, with *S. albomaculata* being the most expensive species. *H. valerioi* and *T. pulverata* were on sale in November and December 2017, and subsequently in May and June 2018. The online platform www.terraristik.com is used to offer specimens for future events. *H. fleischmanni* specimens were offered for USD45 each at the Terraria Fair in Houten, the Netherlands. Spanish traders also use www.terraristik.com to advertise *H. valerioi* sales at the Expoterraria in Madrid.

A TRAFFIC report of a rapid survey of physical and online markets in Japan, covering the period January 2020 to April 2021, found evidence that glass frogs are gaining in popularity, with *H. fleischmanni* ranking within the top 20 species at a reptile fair in Tokyo in 2020 (Kitade and Wakao, 2022). In total, 10 individuals were observed offered for sale at this fair, with at least one specimen described as wild-caught. Two individuals each of *H. valerioi* and *T. pulverata* and one individual *C. granulosa* were also offered for sale at the fair but it was not recorded if they were described by sellers as wild-caught (Kitade, in litt., 2022). Individuals of *H. aureoguttatum* and *S. albomaculata* were also observed offered for sale online, although none were specifically described as wild-caught by sellers. No count data were available for online advertisements (Kitade, in litt., 2022).

A summary of all evidence of international trade data for glass frog species, including accounts of species offered or promoted for sale online, for sale in markets, seized, and imported into LEMIS, is presented in Table 4. Although there is some evidence for international trade in all 12 species identified in trade by the proponents, some of this is restricted to one instance in which they have been advertised online, for example in the cases of *H. iaspidiense* and *H. mondolfii*. Quantitative data on the count of individuals observed in online advertisements are lacking, with numbers reported in seizures relatively low. The only data source that reports source codes for all species is LEMIS and in this only one out of the 12 species identified in trade is reported to have imports of wild-sourced individuals into the USA, in small volumes. No evidence could be found for trade in other species of glass frogs from the sources used by the proponents or the sources consulted in research for this analysis.

Experts consulted by the IUCN SSC Amphibian Specialist Group differed in opinion in relation to the impact of trade on the species; one commented that in 25 years of fieldwork he had observed threats including farming and hunting but never observed evidence of collection of glass frogs for the pet trade (Catenazzi, via IUCN SSC Amphibian Specialist Group, in litt., 2022). Another pointed out that trade data to clarify the magnitude of trade in glass frogs species were lacking (Colombian CITES Scientific Authority, via IUCN SSC Amphibian Specialist Group, in litt., 2022). One expert, however, stated that illegal trade in Ecuador had impacted populations of several species nationally, including *H. fleischmanni* and *E. prosoblepon* (Cisneros-Heredia, via IUCN SSC Amphibian Specialist Group, in litt., 2022).

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

Identification of members of the family Centrolenidae is very complex: their similarity makes it almost impossible to identify species with the naked eye. For some species differentiation is only possible with DNA techniques.

Regular taxonomic changes to the Centrolenidae species provide further support for a family listing to avoid implementation challenges; for example, of 38 new IUCN Red List assessments due to be published in December 2022, 17 will incorporate an updated taxonomy from their previous assessment.

Multiple experts confirmed that identifying species within the glass frog family would be difficult for law enforcement, with some identifications challenging even for specialists (Catenazzi and Cisneros-Heredia, via IUCN SSC Amphibian Specialist Group, in litt., 2022; Klocke, in litt., 2022). Any effective implementation of a CITES Appendix II listing for glass frogs would therefore require all species to be listed to avoid difficulties in identification by enforcement officials.

Additional information

Threats

Wild populations of most glass frog species are affected by severe habitat loss and fragmentation. The main cause of forest loss in these regions is the expansion of commercial agriculture. Habitat loss has also increased as a consequence of logging and timber extraction, mining, human settlements and hydropower projects. Water pollution from herbicides, pesticides, oil spills and illegal crop spraying are also identified as major threats, in addition to the introduction of alien species and emerging infectious diseases such as the chytrid fungus *Batrachochytrium dendrobatidis*.

Of the 12 species identified in trade, trade is not listed as a threat for any and trade is mentioned as a factor for only one, *H. valerioi* (IUCN SSC Amphibian Specialist Group, 2020). According to the Red List assessment: "This species is sustainably exported from Costa Rica for the pet trade". However, there are no official records of a management site in Costa Rica with permits for commercial breeding of this species and therefore there are no exports for legal commercial purposes. The expert referenced in this statement from the IUCN Red List for *H. valerioi* indicated that captive breeding of this species in Costa Rica occurred over a decade ago, with the breeding stock from Costa Rica subsequently exported to a captive breeding centre in Canada to generate conservation funds for the species (Klocke, in litt., 2022).

Conservation, management and legislation

Some glass frogs species are known to inhabit protected areas:

- *Hyalinobatrachium*: 17 of the 36 species inhabit protected areas.
- *Centrolene*: the range of 25 of the 41 species is within or partially within the boundaries of a protected area.
- *Cochranella*: the habitat of 10 of the 24 species is protected.
- *Sachatamia*: three of the four species are within protected areas.

In the range State consultation, no population management plans for glass frog species were reported and there is no known systematised monitoring system.

National regulations governing the breeding, transport, trade and export of wildlife specimens are in place in most Central and South American countries where glass frogs are found, and each country requires a permit for species that are not endangered. Some range States, such as Panama and Ecuador, have allowed the legal export of small quantities of specimens. In some countries, such as Colombia, Costa Rica, and Panama, trade is allowed provided that permits are obtained only for the very specific purposes permitted by law (e.g., scientific purposes). Costa Rica allows commercial trade only in first generation captive-bred species for those not threatened with extinction. No sites in Costa Rica are currently authorised for the commercialisation of glass frogs.

Captive breeding

Studies have shown that captive breeding can be successfully achieved in *H. valerioi* and *S. albomaculata* (Hill et al, 2012; Redbond, 2019). A company based in Canada states that they have individuals of *H. valerioi* available for sale, and that they can export internationally (Understory Enterprises, 2022). The online platform for this company does not explicitly comment on the origin or source of their animals, but one expert commented that *H. valerioi* individuals owned by this company are captive-bred and that the breeding stock originated in Costa Rica (Klocke, in litt., 2022).

A search for captive breeding operations in Nicaragua, the biggest exporter of captive-bred live glass frogs to the USA according to LEMIS, found evidence of a breeding centre "Exotic Fauna", which states on its online platform that they are licensed by the government and exports species including glass frogs to the USA, Canada, and Asia (AFP, 2022), although it was not clear which species. It is also not clear from Exotic Fauna's online platform if amphibians exported from there are captive-bred, captive-born, wild-sourced, or ranched (Exotic Fauna, 2022). It was not possible to verify if the centre is licensed by the government in Nicaragua.

A frog farm in Ecuador operated by a business called Wikiri is reported to be breeding *H. aureoguttatum* in captivity for export and claims to do so to combat poaching. The online platform confirms this species is currently available to order (Wikiri, 2022).

Implementation challenges (including similar species)

Other genera, and in particular the genus *Boana*, have frog species that share some, but not all, of the key features of glass frogs. The genus *Boana* is found throughout South America and contains more than 70 species. Some species, and in particular *B. atlantica* and *B. punctata*, are strikingly similar in colour and pattern to a variety of species in the family Centrolenidae, but differ in the absence of transparent skin on their undersides and in the patterns and colours of their eyes.

One expert pointed out that it can be difficult to ascertain whether species of glass frogs traded are wild-sourced or captive-bred (Catenazzi, via IUCN SSC Amphibian Specialist Group, in litt., 2022), which could make implementation challenging without strict national regulations to verify captive breeding facilities and certification systems. There are indications that laundering of wild individuals that are claimed as captive-bred occurs in some range States: one expert said that interviewed buyers in Ecuador claimed they have evidence that many frogs sold as "bred in captivity from legally obtained parents" through social media are actually taken from the wild and this illegal trade is impacting populations of several species in the country, including *H. fleischmanni* and *E. prosoblepon* (Cisneros-Heredia, via IUCN SSC Amphibian Specialist Group, in litt., 2022). Another expert commented that although exports from Ecuador of *S. albomaculata* are likely to be in captive-bred species from Wikiri, exports of glass frogs from

Nicaragua reported in LEMIS (H. fleischmanni, H. valeroi and T. pulverata) are likely to be wild-caught (Klocke, in litt., 2022).

In Costa Rica, the Attorney General's Office ordered an analysis of all authorised wildlife management sites because anomalies have been detected in the management of species and their reproduction. Laundering was confirmed in an arthropod zoo for export purposes and investigations are open for other sites nationwide. *For any CITES listing to be effective it is important that captive breeding facilities can be inspected and that source codes can be verified so they are accurately reported on permits.*

Potential risk(s) of a listing

The Colombian Management Authority pointed out that including the entire family may discourage efforts to identify taxa that actually have problems with illegal trafficking and to identify and evaluate glass frog populations. They stated that this factor, combined with trade not having been documented by national or global experts as a threat to the species, meant that a family listing was unlikely to have benefits for species conservation (Colombian CITES Scientific Authority, via IUCN SSC Amphibian Specialist Group, in litt., 2022).

A researcher with expertise in glass frogs also commented that a family listing could have disproportionately negative effects on research with subsequently few benefits for conservation. If the family were listed, this would require permits for export of samples such as tissue and skin swabs that contribute to genetic studies relating to phylogeny, species descriptions and monitoring of disease including chytridiomycosis, which could subsequently delay and complicate efforts by researchers (Catenazzi, via IUCN SSC Amphibian Specialist Group, in litt., 2022).

Other comments

The Nomenclature Specialist of the Animals Committee was consulted during the preparation of this proposal to ensure accurate nomenclature for the glass frog family. All his comments and recommendations were incorporated into this proposal.

All range States and France for French Guiana, the other countries of Latin America, the Caribbean and the USA, European Union countries, the UK and other countries globally were consulted.

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Inclusion of Lemur Leaf Frog *Agalychnis lemur* in Appendix II with a zero annual export quota for wild-taken specimens traded for commercial purposes

Proponents: Colombia, Costa Rica, European Union, Panama

Summary: The Lemur Leaf Frog *Agalychnis lemur* is a moderate-sized, canopy-dwelling nocturnal green tree frog that inhabits sloping areas in humid lowland and montane primary forests in Colombia, Costa Rica, and Panama. Females have an average clutch size of 22 eggs in the wild, although clutches of 15 to 70 eggs have been observed in captivity.

Overall extent of occurrence is estimated to be around 80,000 km². The species was assessed by the IUCN Red List in 2019 and classified as Critically Endangered due to an estimated population decline of 80–95% in three generations since 1998, ascribed to habitat loss and potentially the disease chytridiomycosis. The current population is extremely fragmented and restricted mainly to western and central Panama and three reserves in Costa Rica, with the status of subpopulations in Colombia unknown and thought to be rare.

The genus *Agalychnis* was included in Appendix II at CoP15 in 2010. At that time the CITES Standard Reference, Frost (2004), recognised five species. There are currently 14 recognised species in the genus, some of which, including *A. lemur*, were considered to belong to other genera at CoP15 and therefore not included in the original listing. *A. lemur* was not recognised as belonging to this genus until 2010 after CoP15, having previously been included in the genera *Hylomantis* and (before 2005) in *Phyllomedusa*.

No global international trade data exist for this species. US data for this taxon comprised imports for commercial purposes of over 150 live, captive-bred individuals between 2008 and 2020, mostly from non-range States. Around 800 wild-sourced *Agalychnis* individuals, not identified to the species level, were additionally directly exported from range States of *A. lemur* between 2000 and 2014, with most of these exported from Panama (87%) for commercial purposes in 2001.

There are no clear reports of illegal trade in or seizures of the species but there is evidence of demand through the presence of some online advertisements. However, most explicitly state that individuals offered are captive-bred. There is demand for similar species demonstrated through over 46,000 exports of *A. callidryas* between 2010 and 2020 reported in the CITES Trade Database. Trade is not highlighted as a threat to the species in the IUCN Red List assessment and although the assessors indicate that there is demand for the species as pets, it is not clear if individuals are currently being collected from the wild for international trade.

The species is protected from wild harvest for commercial purposes in Costa Rica and Colombia. In Panama the use and transport of wildlife, including *A. lemur*, is prohibited without prior authorisation from the National Directorate of Protected Areas and Wildlife.

Agalychnis lemur is distinguishable from other *Agalychnis* species due to a lack of webbing between its toes, and there are identification guides available for all species in this genus. The species is proposed for listing in Appendix II with a zero annual export quota for wild-taken specimens traded for commercial purposes.

Given recent taxonomic changes, it may be prudent to assess the other species newly reclassified in the genus against CITES listing criteria, particularly as some species are listed as threatened on the IUCN Red List (*A. medinae* has been classified as Endangered).

Analysis: *Agalychnis lemur* has undergone a marked recent population decline in the wild, primarily due to habitat loss. It was assessed as Critically Endangered on the IUCN Red List in 2019. The species already meets the biological criteria for inclusion in Appendix I.

Although *Agalychnis* frogs are popular in the pet trade, there is little evidence of international trade in wild-sourced individuals of this species. However, given the vulnerability of this species to any degree of wild harvest, it meets the criteria for inclusion in Appendix II Criterion A of Annex 2a of Res. Conf. 9.24 (Rev.CoP17). The inclusion of a zero annual export quota for wild-taken specimens traded for commercial purposes would afford the species equivalent protection to an Appendix I listing. Any amendment to this zero quota would need the approval of a future 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.

Taxonomy

Agalychnis lemur was originally placed in the genus *Phyllomedusa*; the species was moved to *Hylomantis*, then subsequently moved from *Hylomantis* to *Agalychnis* in 2010. Scientific synonyms include *Hylomantis lemur* and *Phyllomedusa lemur*.

A proposal to list the genus *Agalychnis* in Appendix II was successful at CoP15 in 2010 (CoP15 Prop. 13); however, it explicitly limited the listing to the five species recognised in the genus by the then-valid nomenclatural CITES Standard Reference Frost (2004): *A. annae*, *A. callidryas*, *A. moreletii*, *A. saltator* and *A. spurrelli*. At that time, *A. lemur* was recognised as belonging to the genus *Hylomantis*, so was therefore not included in the Appendix II listing.

Range

Colombia, Costa Rica, Panama

IUCN Global Category

Critically Endangered A2ace (assessed 2019, ver. 3.1)

Population size and trends

Quantitative population information for the species is not available (IUCN SSC Amphibian Specialist Group, 2020).

The species' population was described as "severely fragmented" occurring in a limited number of locations. Overall, the estimated extent of occurrence for *A. lemur* is around 80,000 km². Since 1998, the species has disappeared from areas where it was once considered common, including in protected areas. The 2019 IUCN Red List assessment estimated that *A. lemur* has experienced a population decline of 80–95% since 1998 (21 years/three generation lengths).

Panama

The majority of the species' current range occurs in Panama. Whilst the 2008 IUCN assessment for *A. lemur* noted the species was "reasonably common" in lower elevations in central and eastern Panama, the 2019 assessment noted that the species could only be found at a few sites in the west of the country, with occasional records from central Panama. Extensive declines have been recorded in western Panama, including from the Reserva Forestal Fortuna, Chiriquí, with no records from this site since 1999 and from El Copé, Coclé, where the species disappeared in 2010. The disappearance of *A. lemur* from Santa Fe National Park was reported in 2003, despite the species being "regularly encountered" in this location in the past.

Costa Rica

Currently only three localities in Costa Rica have confirmed the occurrence of *A. lemur*. The three remaining stable populations in Costa Rica are reported by IUCN to occur in privately owned or indigenous reserves. *Agalychnis lemur* was once considered common in the montane forests of Costa Rica's Talamanca, Tilarán, and Central mountain ranges, however, more recently it has been noted that the species appears to have been completely extirpated from these latter two ranges. A study conducted in 2021 suggested that *A. lemur* is experiencing a "stable recovery" across part of its historical range in the country but noted that *A. lemur* may remain "locally extinct in a large part of its range outside Talamanca". Former sites in Costa Rica where the species was known to occur and has since disappeared include Monteverde, San Ramón, Braulio Carrillo National Park, and Tapantí National Park.

Colombia

The status of subpopulations is unknown, with the species thought to be either rare, elusive, or to have a very small population.

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

International trade for pets has been identified to have had an impact on the species in the past. The *IUCN Red List assessment in 2019* states that “the species is commercially available in the international pet trade, presumably from captive-bred sources”. The assessment further states that it is not clear if individuals are still being collected from the wild (IUCN SSC Amphibian Specialist Group, 2020).

Trade data from LEMIS showed that between 2008 and 2020, all cleared imports of live individuals of *A. lemur* for commercial purposes to the USA were captive-bred. All of the 83 individuals were direct exports from Germany imported between 2016 and 2020. Only four wild-sourced specimens were cleared for import into the USA in this time period, all of which were direct exports from Panama for scientific purposes. There were no wild-sourced live individuals reported in imports, exports or re-exports from the USA between 2008 and 2020.

LEMIS data from 2008–2020 were additionally searched for evidence of trade under the synonyms *Hylomantis lemur* and *Phyllomedusa lemur*. All cleared imports of live individuals of *P. lemur* and *H. lemur* to the USA for commercial purposes were captive-bred, with a total of 19 *P. lemur* and 56 *H. lemur*. All the *P. lemur* individuals were directly imported from Canada in 2010. Most *H. lemur* were imported from Canada between 2010 and 2013, with 10 individuals also imported from Germany in 2010.

Only one wild-sourced specimen of *P. lemur*, originating in Panama, was reportedly re-exported by Argentina to the USA for scientific purposes in 2013. There were 21 wild-sourced specimens and 20 wild-sourced bodies of *H. lemur* imported by the USA for scientific purposes. Nine *H. lemur* scientific specimens were exported by Brazil in 2011, with most of these originating in Costa Rica and one originating in Panama. The remaining 12 *H. lemur* scientific specimens and all 20 bodies were reported in direct exports from Panama in 2008 and 2012.

Low numbers of captive-born and captive-bred individuals of this species (traded as *H. lemur*) were exported by the USA between 2014 and 2018.

Between 2000 and 2014, the USA imported 4,594 wild-sourced *Agalychnis* individuals where the species was not identified; 804 of these individuals came from *A. lemur* range States, the majority of which were exported from Panama in 2001 for commercial purposes (87%). No trade in individuals reported at the genus level (*Agalychnis* spp.) was reported after 2007 for the period spanning 2000–2014. Analysis of LEMIS data for imports since 2014 shows that there have been no imports of unidentified *Agalychnis* species into the USA between 2015 and 2020.

Online surveys demonstrate that live individuals of *A. lemur* are advertised for sale online as pets in small volumes, with online listings for specimens advertising captive-bred individuals priced between USD35–USD60 and EUR35–EUR60 (sites accessed 28th June 2021). A study commissioned by Germany’s CITES Management Authority screened six internet platforms and several Facebook groups between 2017 and 2018 and found 20 advertisements for *A. lemur* on German sites, making the species the second-most popular *Agalychnis* species advertised during the study period (Altherr et al., 2020). However, review of Altherr et al. (2020), undertaken in preparing the present analysis, did not find records for advertisements of *Agalychnis lemur* or its synonyms. A rapid survey of physical and online markets in Japan conducted between January 2020 and April 2021 identified *A. lemur* offered for sale, however, no details were available on the number of specimens observed and whether they were wild-sourced or captive-bred. A rapid search of online advertisements conducted by TRAFFIC in August 2022, found four online advertisements for the species, all from sellers in the USA. All four advertisements stated that the species was currently out of stock but reported that individuals are captive-bred with prices ranging from USD50–USD60 per individual.

Little evidence was found of illegal trade in *A. lemur*. One expert expressed doubts as to the legal acquisition of founder stocks for captive populations that are held outside of the species’ natural range, noting that specimens have been regularly advertised for sale as originating from founder stock exported during the 1980s. It was considered that the localities in which the species is found are relatively easy to access, leaving populations potentially vulnerable to illegal trade. There are no reports of seizures of *A. lemur*, or unidentified *Agalychnis* spp., in WiTIS between 2008 and 2020, or EU-TWIX in 2022.

Illegal trade has been reported in other species within the genus from at least one *A. lemur* range State. Between 1999 and 2008 the USA imported over 250 specimens of *Agalychnis* spp. from Costa Rica and Honduras, despite restrictions on the export of *Agalychnis* species reported to be in place in these countries (CoP15 Prop.13). An expert from the IUCN SSC Conservation Planning Specialist Group stated that they were not aware of any evidence of harvest of wild populations of *A. lemur* for international trade, although, despite strict trade laws in Costa Rica, it

is known that some species are still traded outside of the country (Matamoros, in litt., 2022). According to data in LEMIS, in 2015, 100 captive-bred live individuals of unidentified *Agalychnis* species were refused import and seized by the USA in a direct export from Mexico for commercial purposes. The reason for this seizure is not stated in LEMIS.

As outlined previously, the 2019 IUCN assessment noted that it remains unclear whether wild populations of *A. lemur* are currently being harvested for commercial trade. Data for imports to the USA and online advertisements show some demand for live *A. lemur* individuals as pets. Given the scale and rapidity of the species decline and the restriction of the species to a small number of highly fragmented populations, any collection of wild individuals for international trade would be detrimental to the survival of *A. lemur*. A total of 46,637 CITES-listed *Agalychnis* species have been reported in direct exports in the CITES Trade Database between 2010 and 2020. Almost all exports were of captive-bred live individuals of *Agalychnis callidryas*, originating in Nicaragua, and over 80% of these were imported by the USA. These CITES trade data indicate a degree of international demand for similar species.

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

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved
Populations in Costa Rica and Panama are genetically distinct; any trade could endanger these genetically distinct units.

Additional information

Threats

Declines in *A. lemur* populations are suspected to have been principally caused by chytridiomycosis (an amphibian infectious disease caused by the fungus *Batrachochytrium dendrobatidis*), as well as habitat loss, however the precise cause of the steep declines remains unknown. Habitat loss and fragmentation have the potential to have a large impact on *A. lemur* due to the species' preference for primary forest. Climate change was also reported to affect the species, however, the impact has yet to be quantified.

Deforestation by squatters was reported to impact one of the three remaining Costa Rican populations in Fila Asunción. Recent surveys conducted in the Costa Rican Veragua Rainforest Park and surroundings found that *A. lemur* had disappeared from certain areas after a period of intense wood extraction during 2013. It was noted that a lack of reproductive sites in forests, due to habitat degradation, promoted the species' use of less suitable breeding sites such as exposed flooding banks or small ponds at the forest edge, further increasing the vulnerability of the species.

In Colombia, deforestation and illegal mining were identified as having the greatest impact where the species is known to occur.

Conservation, management and legislation

Costa Rica

Wild species in Costa Rica are protected by the Wildlife Conservation Law No. 7317 of 1992 and its implementing Regulation 40548. Article 14 of the law prohibits the removal of endangered species from the wild for all purposes with the exception of sustainable captive breeding in facilities registered with the General Directorate of Wildlife of the Ministry of the Environment and Energy. Commercial trade is only allowed from the third generation for endangered species.

An *in-situ* conservation effort whereby tadpoles were introduced into artificial ponds at the Costa Rican Amphibian Research Centre in Guayacán has been in place since 2003, with individuals reported to have expanded to other nearby sites that are within the species' historical distribution. The British Zoological Society, in partnership with the Veragua Rainforest Foundation, has planned a habitat restoration and range determination project in Costa Rica.

Ongoing monitoring was reported to be taking place for *A. lemur* subpopulations in Costa Rica.

Panama

Article 15 of the Panama Wildlife Law (No. 24) prohibits the use and transport of wildlife unless prior authorisation is received from the National Directorate of Protected Areas and Wildlife. *A. lemur* is included as an endangered species in the most recent national list that could be located (Resolution No. DM-0657-2016 of 2016).

Agalychnis lemur was given high priority status in Panama's 2011 National Action Plan for Amphibians (Dirección de Áreas Protegidas y Vida Silvestre, 2011). The plan aims to ensure the conservation of amphibians in the country through (a) scientific surveys to update information on the population trends of species and to characterise better the causes of decline, (b) co-ordination and funding of conservation actions, including the identification of important protected areas for amphibians, and (c) the implementation of educational programmes that facilitate conservation activities (Dirección de Áreas Protegidas y Vida Silvestre, 2011).

Colombia

Export for commercial purposes of live *Agalychnis* species, including *A. lemur*, is prohibited.

A. lemur is not included in Colombia's "List of threatened wild species of Colombian biological diversity" in Resolución N° 192, 2014. No information on national management measures could be identified for Colombia.

Captive breeding

A study in 2015 surveyed Panamanian amphibian experts to determine the potential for Panamanian species to avoid extinctions resulting from chytridiomycosis using captive breeding programmes. The authors concluded that of the five genera identified as being most susceptible to chytridiomycosis, Agalychnis spp. were one of the genera with the best chances of establishing viable, long-term captive populations (Gratwicke et al., 2015).

A number of *ex-situ* breeding programmes are in place for *A. lemur*, which demonstrates that this species is capable of being bred in captivity. These include:

Panama

An *ex-situ* population of 60 individuals was present at the El Valle Amphibian Conservation Center (EVACC) in Panama in 2014. In 2018, EVACC reported that they held 105 *A. lemur* adults descended from 11 founders.

USA

The first captive breeding population was established in the Atlanta Botanical Garden in 2001, which as of 2014 held a captive population of 152 individuals. Frogs produced from this captive breeding project were also transferred to the Association of Zoos and Aquariums (AZA) in the USA, totalling 241 captive frogs across 19 AZA zoos.

UK and Sweden

Project Lemur Frog was established in 2012 as an international collaboration between institutions and individuals seeking to conserve the species through collaborative research, *in-situ* and *ex-situ* conservation, and public engagement and education. Through the project, an *ex-situ* *A. lemur* population, representing three distinct bloodlines, was transferred from Manchester Museum and Bristol Zoo to Nordens Ark, Sweden in 2016 with the aim of maintaining an assurance population. Manchester Museum, Bristol Zoo and Nordens Ark continue to maintain captive breeding facilities for *A. lemur*. The European Studbook for *A. lemur* is maintained and co-ordinated by Bristol Zoo.

Germany

Citizen Conservation, an organisation established in Germany in 2018 as a community initiative linking zoos, private breeders, and media professionals, states that this species has been successfully bred in captivity by private breeders (Citizen Conservation, 2022). The organisation has published husbandry guidelines for private breeders of A. lemur (Citizen Conservation, 2019). Germany is the country in which the highest volumes of captive-bred specimens are reported in recent commercial imports to the USA by LEMIS. There is no legal obligation to register captive-bred A. lemur individuals with German authorities (Citizen Conservation, 2019).

Implementation challenges (including similar species)

Agalychnis lemur was reported to be easily distinguishable from other Phyllomedusidae from Central and South America. The lack of webbing between the toes is considered to be a distinctive feature of *A. lemur*, in contrast to other *Agalychnis* species that possess "definite finger and toe webs".

To help distinguish between the five CITES Appendix II listed *Agalychnis* species and other treefrogs within the genus which are not covered by this listing, Mexico's Scientific Authority (CONABIO) produced an identification guide which details the morphological differences between each species.

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Inclusion of Laos Warty Newt *Laotriton laoensis* in Appendix II with a zero export quota for wild-taken specimens traded for commercial purposes

Proponent: European Union

Summary: The Laos Warty Newt *Laotriton laoensis* is a large strikingly marked newt endemic to Lao People's Democratic Republic (PDR), where it inhabits pools at the head of shallow streams at elevations of over 1,000 m. It has a restricted range with a maximum estimated extent of occurrence of 4,800 km², likely in relatively isolated subpopulations. Captive individuals take around four years to reach sexual maturity; this is predicted to take longer in the wild. The species was initially assigned to the genus *Paramesotriton* and named *Paramesotriton laoensis* before being transferred to the monotypic genus *Laotriton* in 2009. The genus *Paramesotriton* was listed in Appendix II in 2019 at CoP18. Commercial trade of *Laotriton laoensis* (as *Paramesotriton laoensis*) has been prohibited in Lao PDR since 2008.

The species was classified as Endangered on the IUCN Red List in 2013 due to its restricted range, continued declines in habitat quality, and extremely restricted population. It was inferred that the species had experienced a population decline of at least 50% in the ten years prior to 2013. There are no quantitative data available on the population size for the species across its distribution. The species is likely distributed in relatively isolated subpopulations as it only occurs in pools at the headwaters of streams, but it can be locally abundant in these; one available population study (2012) in a single area estimated 1,200 individuals in a stream of approximately 5 km.

Overharvesting, mainly for international trade, is stated to be the primary threat to the species. Other factors include habitat loss and fragmentation, and harvest of much smaller volumes for domestic consumption as food and medicine. No populations are known to occur in protected areas.

Live individuals are in demand for international trade, mainly as pets amongst hobbyists in countries including the USA, the UK, Germany, Japan, and Spain. This is thought to be the primary cause of harvest but there is evidence of international trade in dried specimens or specimens soaked in alcohol for use in traditional medicine. The species is vulnerable to overharvesting when populations accumulate in large numbers in accessible pools during the breeding season.

No global trade data exist. The EU recorded exports of 41 live individuals of unknown source in 2013 for commercial purposes from Germany to Japan. Wild-sourced individuals have been observed for sale at fairs in Germany. US data recorded imports of 252 live wild-sourced individuals for commercial purposes from Thailand in 2011. As the species is endemic to Lao PDR, it seems likely that these individuals were first exported (illegally) from Lao PDR into Thailand. There is anecdotal evidence of markets in Thailand where individuals are offered for sale and no evidence of captive breeding of the species in Thailand could be found. A seizure of 120 individuals dried for medicinal purposes from Lao PDR was made at an airport in the USA in 2005. Some exchange of scientific specimens has also been recorded by the USA.

Limited numbers of individuals have been observed offered for sale online (~20 adverts, minimum 65 individuals from 2011–2020). There were anecdotal reports from local Lao residents that over 400 individuals of the species had been offered for sale to collectors from Europe, Japan, and China in domestic markets in 2008 and 2009. More recently, in 2015, there were reports of traders placing orders with local residents for the purchase of the species in unknown quantities at another domestic market. Prices for live individuals vary from an average of USD100 in online advertisements to USD1 in markets in Lao PDR, although there is one instance of a captive-bred juvenile offered for sale online for USD250.

The species is captive-bred by zoos in Europe and North America and there is thought to be a

sufficient supply of offspring in Europe from private captive breeders. In Lao PDR, there are no captive breeding facilities.

Analysis: The endemic *Laotriton laoensis* is classified as Endangered due to its limited distribution and very restricted population, believed to have undergone a marked decline. Although evidence of international trade in wild-sourced individuals of this species is largely anecdotal, it is clear that there is international demand. Recent information on the impact of trade on the wild population is lacking; however, large population accumulations in the breeding season make them vulnerable to overharvesting and the species may already be close to meeting the biological criteria for inclusion in Appendix I. The species therefore appears to meet the criteria for inclusion in Appendix II under Criterion A of Annex 2a, in Res. Conf. 9.24 (Rev.CoP17). A zero-export quota of wild specimens for commercial purposes would afford the species equivalent protection to an Appendix I listing and reflect the prohibition of trade from Lao PDR.

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

Laotriton laoensis (Stuart & Papenfuss, 2002)

CITES Standard reference: Taxonomic Checklist of Amphibian Species listed in the CITES Appendices and the Annexes of EC Regulation 338/97. Species information extracted from Frost, D.R. (ed) (2015). Amphibian Species of the World: a taxonomic and geographic reference, an online reference, version 6.0 as of May 2015 with additional comments by the Nomenclature Specialist of the CITES Animals Committee. See Annex 5 of CoP17 Doc. 81.1 at <https://cites.org/sites/default/files/eng/cop/17/WorkingDocs/E-CoP17-81-01-A5.pdf>

Laotriton is a mono-specific genus, created to accommodate the morphologically unique and molecularly divergent *Laotriton laoensis*, which was originally described as *Paramesotriton laoensis*.

Trade regulation history

Since 2009, the species has been included in EU Annex D. *EU annexes A–C are the equivalent of CITES Appendices I–III, with Annex D for trade monitoring of species that may require listing in the future.* It was originally included under the whole genus *Paramesotriton*, then in 2012 it was listed at the species level as *Paramesotriton laoensis*. In 2013 this was updated to include the species under the updated recognised name of *Laotriton laoensis*.

The genus *Paramesotriton* was listed in CITES Appendix II in 2019 at CoP18 but *L. laoensis* has never been listed in CITES.

Range

Lao PDR

IUCN Global Category

Endangered B1ab (iii,v) (assessed 2013, ver. 3.1)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

Population

This species has a distribution range restricted to a small area in northern Lao PDR, which is not included under any protected area. Increased efforts to discover additional populations in northern, eastern and southern areas of the species' known range have been unsuccessful, providing further evidence of its restricted range and endemism. The current extent of occurrence is approximately 4,560–4,800 km². The 2014 IUCN Red List assessment stated that the species has an "extremely restricted population" (IUCN SSC Amphibian Specialist Group, 2014).

Phimmachak et al. (2012) conducted a mark-recapture study during the dry season and estimated a population size of 1,200 individuals in a 4.7 km stream transect. The researchers concluded that the species can be locally

abundant. One expert commented that although the species may be locally abundant, this is likely to be restricted to specific areas with suitable habitat only (Stuart, via SSC IUCN Amphibian specialist group in litt., 2022). The exact number of sites and an estimate of the entire population remains unknown. On the basis of its small distributional range, the decline in suitable habitat, and the offtake for medicine, food and international trade, the species was considered to have experienced a population decline of at least 50% in the last 10 years. *The Red List assessment states that there are continuing declines in habitat quality and the number of mature individuals (IUCN SSC Amphibian Specialist Group, 2014).*

Species distribution models predict that the available suitable habitat is scarce and most likely restricted to elevations of over 1,000 m above sea level (Chunco et al., 2013). The species is likely distributed in relatively isolated subpopulations, as it only occurs in pools at the headwaters of streams. It is unlikely that significant gene flow occurs between subpopulations.

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

International trade

Overharvesting of this species primarily for the international pet trade, and to a lesser extent for traditional medicines internationally, is said to be the primary threat to the species (IUCN SSC Amphibian Specialist Group, 2014). Laos Warty Newts are prone to harvesting due to having an easily accessible aquatic habitat, bright colouration, a large body size and breeding behaviour in which they accumulate in large numbers in pool sections of streams. *The species can be harvested quickly and easily in large numbers, with the potential for all mature individuals from a site to be harvested in a few days, which is thought to be why they are often sold in their hundreds (IUCN SSC Amphibian Specialist Group, 2014).*

The species was first described in 2002 and since then has been in high demand amongst hobbyists, with the description of the species inadvertently bringing the species and its locations to the attention of commercial traders (IUCN SSC Amphibian Specialist Group, 2014). In 2013, it was on the front cover of Reptiles magazine, which is said to be the world's largest herpetological hobbyist magazine (Stuart, via IUCN SSC Amphibian Specialist Group, in litt., 2022). This edition included a feature article on keeping them as pets (Stuart, via IUCN SSC Amphibian Specialist Group in litt., 2022). The species' large size, bright coloration and rarity are all thought to be factors contributing to its popularity: some hobbyists are especially attracted to rare or recently described species and are willing to pay high sums for such species. Shortly after the species was discovered and described in 2002, commercial collectors from Germany and Japan visited villages in Lao DPR to obtain individuals of the species for sale into the pet trade.

For use in traditional medicine to treat respiratory ailments and arthritis, the species may be boiled in water so that the skin's secretions can easily be scraped off, dried, or preserved in alcohol. Local residents have harvested the species in low numbers for local use in medicine and food for a long-time but it is thought some international trade is also for use as traditional medicine. *There is some evidence that international demand for the species for medicinal purposes may have increased based on interviews with a Vietnamese trader with hundreds of dried individuals in Lao PDR in 2011 (IUCN SSC Amphibian Specialist Group, 2014).*

The trade in this species is predominantly in live animals, with smaller volumes of bodies dried or soaked in alcohol. For traditional medicine these animals are collected for relatively low prices of less than USD1 per individual, but in the international trade rare and recently described species like the Laos Warty Newt can be sold for more than USD200.

Evidence from legal trade databases

*As the species is listed in EU Annex D, trade reported by EU Member States only is recorded in the CITES Trade Database. According to this database, imports of *L. laoensis* into the EU and the UK between 2010 and 2019 consisted solely of two shipments in 2013, in which a total of 41 live individuals of unknown source were imported directly by Germany from Japan for commercial purposes. No trade in this species by EU members has been reported in the CITES Trade Database since 2019.*

According to data from the LEMIS database presented by the proponents, imports of *L. laoensis* into the USA between 2012 and 2021 comprised a total of 21 specimens, all of which were wild-sourced and imported for scientific purposes. Exports of the species from the USA over this period comprised five live captive-bred individuals exported to the Republic of Korea for commercial purposes in 2013. Re-exports of the species from the USA over this period comprised 12 specimens (SPE) for scientific purposes, all of which originated in Lao PDR, and were re-exported to China (3) and Lao PDR (9).

An examination of cleared imports from LEMIS between 2010 and 2011 for this analysis shows that a total of 43 wild-sourced scientific specimens, all of which originated from Lao PDR, were imported into the US for scientific

purposes in 2010. An additional 252 wild-sourced live individuals, all reported in a direct export from Thailand, were imported into the US in 2011 for commercial purposes. This species is endemic to Lao PDR, so it seems likely that these wild-sourced individuals were first smuggled from Lao PDR into Thailand. The species has been protected in Lao PDR since 2008, so legal trade in wild-sourced specimens is assumed only to encompass trade prior to this date.

Evidence from online advertisements

The proponents present raw data as evidence of trade in the species collated from online pet shops and internet platforms in non-range States from September–November 2020. Further analysis of these data presented in Table 1 (Annex 1) by the proponents shows that there were 19 offers for sale for this species between 2012 and 2020, with most (74%) of these between 2017 and 2020. Most offers for sale (12) indicated that the individuals were captive-bred, with six advertisements not stating the source of individuals for sale. Only one advertisement in Germany in 2018 stated wild-caught individuals were for sale but did not give the volume available. Out of the 19 offers for sale, 11 stated the number of individuals available; a minimum of 65 individuals was offered for sale with 48 of these stated to be captive-bred and the remainder of an unknown source. Most advertisements (14) were in the USA with three in Germany and one each in the UK and Spain.

The proponents also collected 75 instances of “demand” for the species between 2011 and 2020. Potential buyers were predominantly in the USA and countries within Europe. The proponents do not clearly define what they categorised as “demand” but some examples of accompanying descriptions are given. The proponents note that one buyer was willing to “pay anything” and “beat other offers” to acquire the desired adults and juveniles, which can run out of stock in just one day. One potential buyer in the USA in 2017 stated they had a preference for wild-caught individuals and were “always looking for more” with “ample cash in hand”. There was also evidence of international demand, with one potential buyer interested in having *L. laoensis* shipped overseas to Argentina from Germany in 2017 and another in having individuals shipped to Japan from the USA in 2019. According to the data from this survey, there was a peak of instances of online demand in 2017.

Commercial prices are thought to depend on the life stage of the animals and the type of seller, with the price for adults higher than for juveniles and pet shops advertising higher prices than commercial private breeders. Reported prices for captive-bred juveniles from online advertisements in the proponents’ survey vary from a minimum of USD50 in 2017 to a maximum of USD250 in 2020, with an average of USD100 per individual for all online advertisements.

Evidence from physical markets

Evidence suggests that the levels of harvest in South-East Asian newts including *L. laoensis* is far higher than the limited number of trade statistics might suggest based on local reports of sales at markets in Lao PDR and Thailand. Anecdotal evidence also suggests that individuals are also regularly observed at fairs in Germany, with some offered for sale from 2015–2018 described as wild-caught.

Improved transportation infrastructure within Lao PDR now subjects the species to increased pressure from outside collectors within and outside the country. Reports on the sale of the species at markets in Lao PDR from local residents include 300–400 individuals per year sold to European or Japanese collectors in 2008 and “hundreds” of individuals to a Chinese collector in 2009. More recently, there were reports of individuals for sale at a market in 2015, with traders in Vientiane and adjacent areas within Lao PDR placing orders with local residents for the species. Prices reported from sales in 2015 were around USD1 per individual and USD18 per kg.

Individuals have also regularly been observed for sale as pets outside of their native range in Chatuchak market, Bangkok, Thailand. It is not clear in which years this has been observed, or how many individuals were offered for sale; more recent reports of trade in markets within Lao PDR were not available (Phimmachak, in litt., 2022).

Evidence from seizures

Around 120 *L. laoensis* individuals, dried for medicinal purposes, were confiscated in the USA in 2005 from a woman who smuggled them from Lao PDR for sale at a traditional medicine shop. There are no reports of seizures other than this event in the WiTIS database from 2008–2022 and EU -TWIX in 2022.

Additional information

Threats

Aside from overharvesting for international trade and a small distribution size, threats according to the IUCN Red List assessment for the species include habitat loss and degradation. Anthropogenic activities such as the conversion of land for agricultural practices and infrastructure development are negatively impacting sites across the species’ range. The species is also likely to be impacted by changes in stream water quality or flow and extensive burning in areas adjacent to streams. The species is also threatened by local consumption as both food and traditional medicine.

Conservation, management and legislation

The species is known to occur entirely outside of protected areas.

Since 2008, the commercial trade of *Laotriton laoensis* (as *Paramesotriton laoensis*) has been prohibited in Lao PDR as the species was listed as a Category I species under the Lao Wildlife and Aquatic Animal Law (Decree No. 81/PM 2008).

Captive breeding

There are reports of captive breeding and husbandry of the species in Europe.

The first captive breeding of the species in a public zoo or aquarium succeeded in 2011 and was repeated in 2012 and 2018 with the aim of contributing to *ex-situ* breeding programmes, or for display and education purposes. According to the Zoological Information Management System (ZIMS), a total of 46 Laos Warty Newt individuals are currently kept in seven facilities worldwide: one in Germany, one in Poland, two in the UK, one in Russia, and two in North America.

The species is also being bred by several private breeders, with the first documented success in 2006. *At the end of 2006 a register was established for the species by a group of newt and salamander enthusiasts named "AG Urodela" and findings from captive husbandry were made available through journal articles; in 2017 it was stated that there had been a "sufficient" supply of offspring of the species in Europe for several years (Bachhausen, 2017)*

No captive breeding has been documented in Lao PDR.

Implementation challenges (including similar species)

Laotriton laoensis has been previously described as *P. laoensis* and the two genera share some characteristic morphological traits. The genus *Paramesotriton* has been listed in CITES Appendix II since 2019 and a local expert stated that *L. laoensis* is easy to distinguish from species within the *Paramesotriton* genus (Phimmachak, in litt., 2022). A listing of *L. laoensis* in Appendix II is therefore unlikely to cause issues for enforcement and implementation in either genus.

Potential benefit(s) of listing for trade regulation

Laotriton laoensis may be a reservoir species for the chytrid fungus *Batrachochytrium*, which has been detected in species of the closely related genus *Paramesotriton* in 2017 (Yuan et al., 2018). The recent introduction of this fungus in Europe has been linked to trade in eastern Asian salamandrid species and now threatens the survival of a large proportion of western Palearctic urodeles.

One expert stressed that it is important to regulate the movement of species like *L. laoensis* that can spread pathogens through the pet trade. An Appendix II listing with a zero export quota for wild-caught specimens can help to ensure that any legal trade is based on captive-produced specimens and that additional controls are in place to curb the illegal harvest of wild specimens for the international pet trade. *Laotriton laoensis* is already banned from the state of North Carolina to avoid the potential introduction of the chytrid fungus *Batrachochytrium* (Stuart, via SSC IUCN Amphibian Specialist Group, in litt., 2022).

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Inclusion of Requiem Sharks (Family Carcharhinidae) in Appendix II

Proponents: Bangladesh, Colombia, Dominican Republic, Ecuador, El Salvador, European Union, Gabon, Israel, Maldives, Panama, Senegal, Seychelles, Sri Lanka, Syrian Arab Republic, United Kingdom of Great Britain and Northern Ireland

Summary: Two members of the family Carcharhinidae are currently included in Appendix II (*Carcharhinus longimanus*, included at CoP16 and *Carcharhinus falciformis* at CoP17). The remaining 54 members of the family (in twelve genera) are now proposed for inclusion in Appendix II; 19 in accordance with Article II, paragraph 2(a) of the Convention (from here onwards referred to as lead species), and all remaining species in accordance with Article II paragraph 2(b) (as lookalike species). The species included in this Proposal occur in tropical to temperate oceans and are found in coastal, pelagic, and riverine environments. Most of the lead species are found inshore along continental and insular shelves in relatively shallow water (< 150 m), although some (*C. obscurus* and *C. signatus*) are semi-oceanic and found at depths of 500–600 m.

All lead species proposed are known, or inferred, to have low productivity. Most species reach a maximum size of under 200 cm, although *C. obscurus* reaches 420 cm. The frequency of reproduction is typically annual to biennial, with most species producing between 2 and 10 pups per litter (*C. signatus* can produce up to 15 pups). The estimated three generation period is under 30 years for most of these species, although *C. plumbeus* (60–78 years) and *C. obscurus* (90–114 years) are exceptions.

All members of the family Carcharhinidae are subject to targeted and incidental catch by commercial, artisanal, and recreational fishers. Their fins and, increasingly their meat, feature prominently in international trade. Species in the family together comprise one-third of the annual global chondrichthyan (cartilaginous fish) catch and two-thirds of the shark fin trade of coastal sharks. One species alone (*Prionace glauca*, not a lead species) was estimated to make up one sixth of all shark landings in 2017 and may now dominate the international shark meat trade. Other products such as liver oil, skin, and jaws are also used. While some species are common in trade, other carcharhinids are extremely rare and have not been found in surveys of large fin trading hubs.

Three recent fin surveys in Hong Kong SAR and Guangzhou, China, confirmed the dominance of the family Carcharhinidae in the fin trade. *Prionace glauca*, *Carcharhinus falciformis*, and the Blacktip complex (*C. limbatus*, *C. amblyrhynchoides*, *C. leiodon*, and *C. tilstoni*) constituted the top three most-sampled species or species groups in Hong Kong SAR in 2014–2015. Another study in Hong Kong SAR and Guangzhou in 2015–2017, found that *P. glauca*, *C. falciformis*, and *Carcharhinus* spp. constituted the top three most-sampled species or species groups. A recent survey of small, low-value fins from 2018–2019 in Hong Kong SAR reported *Rhizoprionodon acutus*, *C. sorrah*, and the Blacktip complex as three of the top four most sampled species or species groups.

All 19 lead species are classified as either Endangered or Critically Endangered on the IUCN Red List. Twelve (*Carcharhinus acronotus*, *C. amblyrhynchoides*, *C. dussumieri*, *C. leiodon*, *C. obscurus*, *C. perezii*, *C. plumbeus*, *C. signatus*, *Lamiopsis temminckii*, *L. tephrodes*, *Nasolamia velox*, and *Negaprion acutidens*) are globally listed as Endangered due to steep recent population reductions (>50% over three generations) mainly due to overfishing. Seven species (*C. borneensis*, *C. cerdale*, *C. hemiodon*, *C. porosus*, *C. obsoletus*, *Glyphis gangeticus*, and *Isogomphodon oxyrinchus*) are globally listed as Critically Endangered resulting from recent population reductions of >80% over three generations inferred from overfishing and habitat loss. Four of these (*C. borneensis*, *C. hemiodon*, *C. obsoletus*, and *Glyphis gangeticus*) are thought to be locally, regionally or globally extinct throughout much or all of their range.

Species-specific reductions have been estimated from three main sources: fisheries agency stock assessments (*C. acronotus*, *C. obscurus*, *C. plumbeus*); Global FinPrint spatial depletion estimates on coral reef habitats (*C. amblyrhynchoides*, *C. perezii*); or catch landings and effort data (*C. cerdale*,

C. dussumieri, *C. leiodon*, *C. porosus*, *C. signatus*, *Isogomphodon oxyrinchus*, *Lamiopsis temminckii*, *L. tephrodes*, *Nasolamia velox*, and *Negaprion acutidens*).

Legislation and regulations are in place in some countries for some of the lead species in this proposal; particularly for *C. obscurus*, *Glyphis gangeticus*, and several species found in Brazilian waters (*C. perezi*, *C. plumbeus*, *C. signatus*, and *Isogomphodon oxyrinchus*). A recent study examined management risk for 18 carcharhinid species and reported that the best managed were *C. acronotus*, *C. porosus*, and *C. tilstonii* (not a lead species), while the most poorly managed were *C. leiodon*, *C. dussumieri*, and *C. melanopterus* (not a lead species). Beyond a limited set of management measures, it is assumed that most species in this proposal are unmanaged throughout their respective ranges. A recent study found that the family Carcharhinidae is inadequately managed worldwide, both by nations and Regional Fisheries Management Organizations (RFMOs), with only half of the necessary management in place.

While fins are the most conspicuous and widely recognised shark products in trade, other products are also used, including meat, oil, skins, and jaws. The meat trade is recognised as a growing threat to many shark and ray species, although there are limited data on species composition. Genetic identification is necessary to identify traded meat to species level, although this is not possible within the Blacktip Complex. While identification of attached fins is possible, dried fins (the form in which fins are traded) from different species resemble those of other species of shark. Trained personnel can identify many dried fins to species level; however, it can be difficult for untrained people to do this. Species included in this proposal were considered lookalikes based primarily on “similar species” that shared dried fin characteristics.

Analysis: The family Carcharhinidae dominates the global chondrichthyan catch and fin trade. Declines are consistent with the indicative guidelines for inclusion in Appendix II of commercially exploited low-productivity aquatic species suggested in the footnote to Annex 5 of Resolution Conf. 9.24 (Rev. CoP17) for 11 of the 19 lead species identified in the current proposal (*C. borneensis*, *C. cerdale*, *C. hemiodon*, *C. leiodon*, *C. obscurus*, *C. obsoletus*, *C. plumbeus*, *C. porosus*, *Glyphis gangeticus*, *Isogomphodon oxyrinchus*, and *Nasolamia velox*). While there was no evidence of international trade in fins of *C. borneensis*, *C. cerdale*, *C. hemiodon*, *C. obsoletus*, and *Isogomphodon oxyrinchus*, these are exceptionally rare and possibly extinct, but have the potential to enter international trade if caught and may already meet the criteria for inclusion in Appendix I.

Available information for four of the 19 species (*C. acronotus*, *C. amblyrhynchus*, *C. dussumieri*, and *Negaprion acutidens*), indicates that at present these do not meet the criteria in 2a A when following the footnote in Annex 5 of the Resolution. However, evidence of ongoing declines and the presence of fins in international trade of these and *Lamiopsis temminckii* implies that regulation is needed to ensure that the harvest is not reducing the wild populations of these to a level at which their survival may be threatened by continued harvesting or other influences, indicating that these species meet the criteria for inclusion in Appendix II in Annex 2aB of the Resolution.

Two species, *C. signatus* and *C. perezi*, did not appear to meet the Annex 2a criteria for inclusion in Appendix II as there was no evidence of these species in international trade and estimated declines did not meet the indicative guidelines for inclusion in Appendix II of commercially exploited low-productivity aquatic species suggested in the footnote to Annex 5. However, both species are declining globally. These species meet the criteria for listing in Annex 2bA, based on the difficulty of distinguishing their fins from at least one of the species that met Annex 2aA in this proposal, as well as two species that are already included in Appendix II (*Sphyrna mokarran* and *Carcharhinus falciformis*).

When considering all other (non-lead) species in the family Carcharhinidae, 27 species have been assessed for the IUCN Red list as Near Threatened or Vulnerable (declining in some or all areas of their range), and there was evidence of these species in international trade (these were: *Carcharhinus altimus*, *C. albigmarginatus*, *C. amblyrhynchoides*, *C. amboinensis*, *C. brachyurus*, *C. brevipinna*, *C. isodon*, *C. leucas*, *C. limbatus*, *C. macroti*, *C. melanopterus*, *C. sealei*, *C. sorrah*, *Glyphis garricki*, *Glyphis glyphis*, *Loxodon macrorhinus*, *Negaprion brevirostris*, *Prionace glauca*,

Rhizoprionodon acutus, *R. lalandii*, *R. longurio*, *R. oligolinx*, *R. porosus*, *R. taylori*, *Scoliodon laticaudus*, *S. macrorhynchus*, and *Triaenodon obesus*). These species may also meet the criteria for inclusion in Appendix II in Annex 2aB in that regulation is needed to ensure that the harvest of specimens from the wild is not reducing the wild populations to a level at which their survival may be threatened by continued harvesting or other influences.

Some 16 of these species (*Carcharhinus altimus*, *C. albimarginatus*, *C. amboinensis*, *C. brevipinna*, *C. leucas*, *C. limbatus*, *C. sorrah*, *Negaprion brevirostris*, *Prionace glauca*, *Rhizoprionodon acutus*, *R. lalandii*, *R. longurio*, *R. oligolinx*, *R. porosus*, *R. taylori*, and *Triaenodon obesus*) and two additional species (*C. galapagensis* and *Rhizoprionodon terraenovae*) also have fins that are difficult to distinguish from some of the lead species above as well as species already included in the Appendices (*Sphyrna mokarran* and *Carcharhinus falciformis*). These appear to meet the (lookalike) criteria for listing in Annex 2bA.

Seven species, *C. cautus*, *C. coatesi*, *C. fitzroyensis*, *C. humani*, *C. tjutjot*, *C. tilstoni*, and *Lamiopsis tephrodes* clearly did not meet criteria for listing in either Annex 2a or Annex 2b. *Lamiopsis tephrodes* (a lead species) is listed as Endangered, *C. cautus*, *C. coatesi*, *C. fitzroyensis*, and *C. tilstoni* were globally listed as Least Concern, *C. tjutjot* was globally listed as Vulnerable, and *C. humani* as Data Deficient. Of these only *C. coatesi*, *C. fitzroyensis*, and *C. tilstoni* have been detected in international trade but harvest for trade is not considered a major conservation concern.

In summary, the great majority of species in the family Carcharhinidae appear to meet the criteria for inclusion in Appendix II, either because regulation is needed to ensure that the harvest is not reducing the wild populations to a level at which their survival may be threatened by continued harvesting or other influences (Annex 2a of the Resolution) or as lookalikes (Annex 2b of the Resolution). Inclusion of the remaining seven species in the Appendices would facilitate compliance.

Summary Table

Table 1. Scientific and common names of the family Carcharhinidae. Criteria met in this Analysis of Resolution 9.24 (Rev. CoP17) are indicated in the three columns to the right.

| | Lead Species | | Analysis Criteria Met | | |
|-----------------------------|--------------------------------------|----------------------------|-----------------------|-----------|-----------|
| | Scientific name | Common name | Annex 2aA | Annex 2aB | Annex 2bA |
| 1 | <i>Carcharhinus acronotus</i> | Blacknose Shark | | ✓ | |
| 2 | <i>Carcharhinus amblyrhynchus</i> | Grey Reef Shark | | ✓ | ✓ |
| 3 | <i>Carcharhinus borneensis</i> | Borneo Shark | ✓ | | |
| 4 | <i>Carcharhinus cerdale</i> | Pacific Smalltail Shark | ✓ | | |
| 5 | <i>Carcharhinus dussumieri</i> | Whitecheek Shark | | ✓ | |
| 6 | <i>Carcharhinus hemiodon</i> | Pondicherry Shark | ✓ | | |
| 7 | <i>Carcharhinus leiodon</i> | Smoothtooth Blacktip Shark | ✓ | ✓ | |
| 8 | <i>Carcharhinus obscurus</i> | Dusky Shark | ✓ | ✓ | ✓ |
| 9 | <i>Carcharhinus obsoletus</i> | Lost Shark | ✓ | | |
| 10 | <i>Carcharhinus perezi</i> | Caribbean Reef Shark | | | ✓ |
| 11 | <i>Carcharhinus plumbeus</i> | Sandbar Shark | ✓ | ✓ | ✓ |
| 12 | <i>Carcharhinus porosus</i> | Smalltail Shark | ✓ | ✓ | |
| 13 | <i>Carcharhinus signatus</i> | Night Shark | | | ✓ |
| 14 | <i>Glyphis gangeticus</i> | Ganges Shark | ✓ | ✓ | |
| 15 | <i>Isogomphodon oxyrinchus</i> | Daggernose Shark | ✓ | | |
| 16 | <i>Lamiopsis temminckii</i> | Broadfin Shark | | ✓ | |
| 17 | <i>Lamiopsis tephrodes</i> | Borneo broadfin Shark | | | |
| 18 | <i>Nasolamia velox</i> | Whitenose Shark | ✓ | | |
| 19 | <i>Negaprion acutidens</i> | Sharptooth Lemon Shark | | ✓ | |
| Other Carcharhinidae | | | | | |
| | <i>Carcharhinus altimus</i> | Bignose Shark | | ✓ | ✓ |
| | <i>Carcharhinus albimarginatus</i> | Silvertip Shark | | ✓ | ✓ |
| | <i>Carcharhinus amblyrhynchoides</i> | Graceful Shark | | ✓ | |

| | | | | | |
|-----------------------------|-----------------------------------|----------------------------|--|---|---|
| | <i>Carcharhinus amboinensis</i> | Pigeye Shark | | ✓ | ✓ |
| | <i>Carcharhinus cautus</i> | Nervous Shark | | | |
| | <i>Carcharhinus brachyurus</i> | Bronze Whaler | | ✓ | |
| | <i>Carcharhinus brevipinna</i> | Spinner Shark | | ✓ | ✓ |
| | <i>Carcharhinus coatesi</i> | Coates's Shark | | | |
| | <i>Carcharhinus fitzroyensis</i> | Creek Whaler | | | |
| | <i>Carcharhinus galapagensis</i> | Galapagos Shark | | | ✓ |
| | <i>Carcharhinus humani</i> | Human's Whaler Shark | | | |
| | <i>Carcharhinus isodon</i> | Finetooth Shark | | ✓ | |
| | <i>Carcharhinus leucas</i> | Bull Shark | | ✓ | ✓ |
| | <i>Carcharhinus limbatus</i> | Blacktip Shark | | ✓ | ✓ |
| | <i>Carcharhinus macloti</i> | Hardnose Shark | | ✓ | |
| | <i>Carcharhinus melanopterus</i> | Blacktip Reef Shark | | ✓ | |
| | <i>Carcharhinus sealei</i> | Blackspot Shark | | ✓ | |
| | <i>Carcharhinus sorrah</i> | Spottail Shark | | ✓ | ✓ |
| Other Carcharhinidae | | | | | |
| | <i>Carcharhinus tilstoni</i> | Australian Blacktip Shark | | | |
| | <i>Carcharhinus tjutjot</i> | Indonesian Whaler | | | |
| | <i>Glyphis garricki</i> | New Guinea River Shark | | ✓ | |
| | <i>Glyphis glyphis</i> | Speartooth Shark | | ✓ | |
| | <i>Loxodon macrorhinus</i> | Sliteye Shark | | ✓ | |
| | <i>Negaprion brevirostris</i> | Lemon Shark | | ✓ | ✓ |
| | <i>Prionace glauca</i> | Blue Shark | | ✓ | ✓ |
| | <i>Rhizoprionodon acutus</i> | Milk Shark | | ✓ | ✓ |
| | <i>Rhizoprionodon landii</i> | Brazilian Sharpnose Shark | | | ✓ |
| | <i>Rhizoprionodon longurio</i> | Pacific Sharpnose Shark | | ✓ | ✓ |
| | <i>Rhizoprionodon oligolinx</i> | Grey Sharpnose Shark | | ✓ | ✓ |
| | <i>Rhizoprionodon porosus</i> | Caribbean Sharpnose Shark | | ✓ | ✓ |
| | <i>Rhizoprionodon taylori</i> | Australian Sharpnose Shark | | ✓ | ✓ |
| | <i>Rhizoprionodon terraenovae</i> | Atlantic Sharpnose Shark | | | ✓ |
| | <i>Scoliodon laticaudus</i> | Spadenose Shark | | ✓ | |
| | <i>Scoliodon macrorhynchus</i> | Pacific Spadenose Shark | | ✓ | |
| | <i>Triaenodon obesus</i> | Whitetip Reef Shark | | ✓ | ✓ |

Summary of Available Information

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

Approach to this analysis

The following detail focuses on the 19 lead species proposed for listing under the criteria in Annex 2a of Res. Conf. 9.24 (Rev. CoP17).

Given the large number of taxa they were considered in three stages. First, species were assessed against the criteria in Annex 2aA, with consideration to the indicative guidelines for inclusion in Appendix II of commercially exploited low productivity aquatic species suggested in the footnote to Annex 5 of Resolution 9.24 (Rev. CoP17). Next, species were assessed against the criteria in Annex 2aB of Res. Conf. 9.24 (Rev. CoP17) by examining the effect that trade regulation may have at controlling harvest of the wild population, as well as the responsiveness of the species to effective management. Lastly, species were assessed against the criteria for listing in Annex 2bA of Resolution Conf. 9.24 (Rev CoP17) based on the difficulty of distinguishing their fins from those that were assessed as meeting criteria in Annex 2aA in this proposal or species already listed in Appendix II.

A summary of the population trend information for the remaining species is included in Table 2, and these species are discussed in less detail in the section on Annex 2b.

IUCN Global Category and population trend information

Table 2. Family Carcharhinidae, IUCN Red List global category (all using IUCN Red List version 3.1), and key population trend information used in this analysis. Population trend information taken from the IUCN Red List for each species, unless otherwise noted.

| Species | IUCN Red List Global Category | Population Trend Information |
|-----------------------------------|--|---|
| Lead Species | | |
| <i>Carcharhinus acronotus</i> | Endangered A2bd (2019) (Carlson et al., 2021c) | Suspected reduction over three generation lengths (global): 50–79% Annual rates of reduction (U.S. South Atlantic and Gulf of Mexico): 1.8%, 2.1%, and 6.1%; annual rate of increase: 4.8%. Reduction over three generation lengths (Brazil): 44% |
| <i>Carcharhinus amblyrhynchos</i> | Endangered A2bcd (2020) (Simpfendorfer et al., 2020a) | Estimated reduction over three generation lengths (global): 59% Reduction over three generation lengths (some regions): 75% |
| <i>Carcharhinus borneensis</i> | Critically Endangered A2cd (2020) (Dulvy et al., 2021a) | Suspected reduction over three generation lengths (Global): >80% |
| <i>Carcharhinus cerdale</i> | Critically Endangered A2bcd (2019) (Pollom et al., 2020d) | Suspected reduction over three generation lengths (Global): >80% |
| <i>Carcharhinus dussumieri</i> | Endangered A2d+3d (2018) (Simpfendorfer et al., 2019) | Suspected reduction over three generation lengths (Global): 50–79% Abundant, common catch in some regions (Arabian Gulf and Sea of Oman); and suspected reduction over three generation lengths of at least 50–70% in India. |
| <i>Carcharhinus hemiodon</i> | Critically Endangered C2a(i) (2020) (Kyne et al., 2021a) | Lack of verifiable records since 1960 combined with evidence of high catch and trade of congeners leads to a weighted average probability of extinction is 0.44–0.45, which is <0.5 and therefore marginally outside the threshold for IUCN Critically Endangered (Possibly Extinct). Estimated (Global) population size now <250 individuals |
| <i>Carcharhinus leiodon</i> | Endangered A2cd+3cd (2017) (Simpfendorfer et al., 2017) | Suspected reduction over three generation lengths (Global): 50–79% Rediscovered in 2009, few specimens, likely mistaken for other Blacktip complex species, and therefore to have suffered similar reductions. |
| <i>Carcharhinus obscurus</i> | Endangered A2bd (2018) (Rigby et al., 2019a) | Estimated reduction over three generation lengths (global): 75.8% Annual rates of reductions: 2.6% (Northwest Atlantic); 0.9% (Western Indian Ocean). Recovered (Eastern Indian Ocean) |
| <i>Carcharhinus obsoletus</i> | Critically Endangered A2d; D (2020) (Dulvy et al., 2020) | Suspected reduction over three generation lengths (Global): >80% and the population size is now suspected to be fewer than 50 individuals. Three type specimens, last of which was reported in 1934. Considerable fishing pressure in the region and directed and passive surveys have not revealed this species since. |
| <i>Carcharhinus perezi</i> | Endangered A2bcd (2019) (Carlson et al., 2021a) | Estimated reduction over three generation lengths (global): 52.5% (standard error 40.4–64.5%) |
| <i>Carcharhinus plumbeus</i> | Endangered A2bd (2020) (Rigby et al., 2021b) | Suspected reduction over three generation lengths (Global): 50–79% |

| Species | IUCN Red List Global Category | Population Trend Information |
|---------------------------------------|---|---|
| | | Annual rates of reduction: 2.2% (United States) and 3.0% (South Africa) |
| <i>Carcharhinus porosus</i> | Critically Endangered A2d (2019) (Pollom et al., 2020a) | Suspected reduction over three generation lengths (Global): >80% Reduction over three generation lengths (northern Brazil): 85% |
| <i>Carcharhinus signatus</i> | Endangered A2bd (2019) (Carlson et al., 2021b) | Suspected reduction over three generation lengths (Global): 50–79% Annual rate of reduction (Northwest Atlantic): 2.4%, or 0.43% if corrected for possible reporting error. Annual rate of reduction (Brazil): 8.1% (includes <i>C. falciformis</i>) |
| <i>Glyphis gangeticus</i> | Critically Endangered A2cd;C2a(i) (2021) (Rigby et al., 2021a) | Suspected reduction over three generation lengths: >80% |
| <i>Isogomphodon oxyrinchus</i> | Critically Endangered A2bcd (2019) (Pollom et al., 2020b) | Inferred reduction over three generation lengths: >80%. Annual rate of reduction (northwestern Brazil): 18.4% |
| <i>Lamiopsis temminckii</i> | Endangered A2d (2020) (Dulvy et al., 2021c) | Suspected reduction over three generation lengths (Global): 50–79% Historically common, now rare or absent throughout its range. |
| <i>Lamiopsis tephrodes</i> | Endangered A2d (2020) (Dulvy et al., 2021b) | Suspected reduction over three generation lengths (Global): 50–79% |
| <i>Nasolamia velox</i> | Endangered A2cd (2019) (Pollom et al., 2020c) | Suspected reduction over three generation lengths (Global): 50–79% Historically common, now rare throughout its range. |
| <i>Negaprion acutidens</i> | Endangered A2bd (2020) (Simpfendorfer et al., 2021a) | Suspected reduction over three generation lengths (Global): 50–79% Stable or common in some regions (oceanic islands in Western Pacific and Indian Oceans), heavily depleted in other regions (Red Sea, Pakistan). |
| Other Carcharhinidae | | |
| <i>Carcharhinus altimus</i> | Near Threatened (2020) (Rigby et al., 2020a) | Suspected reduction over three generation lengths (Global): 20–29% |
| <i>Carcharhinus albimarginatus</i> | Vulnerable (2015) (Espinoza et al., 2021) | Suspected reduction over three generation lengths (Global): >30% |
| <i>Carcharhinus amblyrhynchoideus</i> | Vulnerable (2020) (Simpfendorfer et al., 2021b) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Carcharhinus amboinensis</i> | Vulnerable (2020) (Simpfendorfer et al., 2021c) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Carcharhinus caudatus</i> | Least Concern (2020) (Morgan et al., 2020) | Not suspected to be close to reaching the population reduction threshold |
| <i>Carcharhinus brachyurus</i> | Vulnerable (2020) (Huveneers et al., 2020) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Carcharhinus brevipinna</i> | Vulnerable (2020) (Rigby et al., 2020b) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Carcharhinus coatesi</i> | Least Concern (2018) (Baje, 2019) | There is nothing to infer or suspect a population reduction. |
| <i>Carcharhinus fitzroyensis</i> | Least Concern (2018) (Harry et al., 2019) | Relatively productive and is likely able to tolerate the current and projected level of [fishing] pressure. |
| <i>Carcharhinus galapagensis</i> | Least Concern (2018) (Kyne et al., 2019) | The global population is not suspected to have undergone a population reduction reaching a level to suggest Near Threatened. |
| <i>Carcharhinus humani</i> | Data Deficient (2018) (Pollom et al., 2019) | Lack of population trend data across its range. |

| Species | IUCN Red List Global Category | Population Trend Information |
|-----------------------------------|---|--|
| <i>Carcharhinus isodon</i> | Near Threatened (2019) (Carlson et al., 2021d) | Suspected reduction over three generation lengths (Global): 20–29% |
| <i>Carcharhinus leucas</i> | Vulnerable (2020) (Rigby et al., 2021c) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Carcharhinus limbatus</i> | Vulnerable (2020) (Rigby et al., 2021d) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Carcharhinus macroti</i> | Near Threatened (2020) (Rigby et al., 2021e) | Suspected reduction over three generation lengths (Global): 20–29% |
| <i>Carcharhinus melanopterus</i> | Vulnerable (2020) (Simpfendorfer et al., 2020b) | Estimated reduction over three generation lengths (Global): 30–49% |
| <i>Carcharhinus sealei</i> | Vulnerable (2020) (Dulvy et al., 2021d) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Carcharhinus sorrah</i> | Near Threatened (2020) (Simpfendorfer et al., 2021d) | Suspected almost to meet reduction criteria for Vulnerable. |
| <i>Carcharhinus tilstoni</i> | Least Concern (2018) (Johnson et al., 2019) | Documented recovery and the implementation of management measures. |
| <i>Carcharhinus tjtjt</i> | Vulnerable (2019) (Rigby and Kyne, 2019) | Suspected reduction over three generation lengths (Global): >30% |
| <i>Glyphis garricki</i> | Vulnerable (2021) (Kyne et al., 2021b) | Projected reduction over the next three generation lengths: >10% |
| <i>Glyphis glyphis</i> | Vulnerable (2021) (Kyne et al., 2021c) | Projected reduction over the next three generation lengths: >10% |
| <i>Loxodon macrorhinus</i> | Near Threatened (2021) (Rigby et al., 2021f) | Suspected reduction over three generation lengths (Global): 20–29% |
| <i>Negaprion brevirostris</i> | Vulnerable (2020) (Carlson et al., 2021e) | Suspected reduction over three generation lengths (Global): 30–49% |
| Other Carcharhinidae | | |
| <i>Prionace glauca</i> | Near Threatened (2018) (Rigby et al., 2019b) | Estimated reduction over three generation lengths (Global): 20–29% |
| <i>Rhizoprionodon acutus</i> | Vulnerable (2020) (Rigby et al., 2020c) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Rhizoprionodon landii</i> | Vulnerable (2019) (Pollom et al., 2020e) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Rhizoprionodon longurio</i> | Vulnerable (2019) (Pollom et al., 2020f) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Rhizoprionodon oligolinx</i> | Near Threatened (2021) (Rigby et al., 2021g) | Suspected reduction over three generation lengths (Global): 20–29% |
| <i>Rhizoprionodon porosus</i> | Vulnerable (2019) (Carlson et al., 2021f) | Suspected reduction over three generation lengths (Global): 30–49% |
| <i>Rhizoprionodon taylori</i> | Least Concern (2019) (Baje and Simpfendorfer, 2019) | Limited catches, high productivity, and high rates of population increase. |
| <i>Rhizoprionodon terraenovae</i> | Least Concern (2019) (Carlson et al., 2021g) | No evidence of population reduction, not suspected to be close to reaching the population reduction threshold. |
| <i>Scoliodon laticaudus</i> | Near Threatened (2020) (Dulvy et al., 2021e) | Suspected reduction over three generation lengths (Global): 20–29% |
| <i>Scoliodon macrorhynchus</i> | Near Threatened (2020) (Rigby et al., 2020d) | Suspected to be close to reaching the population reduction threshold. |
| <i>Triaenodon obesus</i> | Vulnerable (2020) (Simpfendorfer et al., 2020c) | Estimated reduction over three generation lengths (Global): 30–49% |

Range**Blacknose Shark *Carcharhinus acronotus***

Endangered A2bd (assessed 2019, ver 3.1) (Carlson et al., 2021c)

Found in the Western Central and Southwest Atlantic Oceans from North Carolina to Brazil, including the Gulf of Mexico and Caribbean Sea. *Inhabits continental and insular shelves at depths of 18–64 m* (Ebert et al., 2013). *Carcharhinus acronotus* is found in the following FAO Fishing Areas: 31, 41. Extant range countries and territories are: Anguilla; Antigua and Barbuda; Aruba; Bahamas; Barbados; Belize; Bermuda; Bonaire, Sint Eustatius and Saba; Brazil; Cayman Islands; Colombia; Costa Rica; Cuba; Curaçao; Dominica; Dominican Republic; French Guiana; Grenada; Guadeloupe; Guatemala; Guyana; Haiti; Honduras; Jamaica; Martinique; Mexico (Veracruz, Tamaulipas,

Tabasco, Quintana Roo, Yucatán, Campeche); Montserrat; Nicaragua; Panama; Puerto Rico (Puerto Rico (main island)); Saint Barthélemy; Saint Kitts and Nevis; Saint Lucia; Saint Martin (French part); Saint Pierre and Miquelon; Saint Vincent and the Grenadines; Sint Maarten (Dutch part); Suriname; Trinidad and Tobago; Turks and Caicos Islands; United States (Louisiana, Alabama, Texas, Mississippi, Georgia, Florida, North Carolina, South Carolina); Uruguay; Bolivarian Republic of Venezuela (henceforth Venezuela) (Venezuelan Antilles); British Virgin Islands; US Virgin Islands.

Grey Reef Shark *Carcharhinus amblyrhynchos*

Endangered A2bcd (assessed 2020, ver. 3.1) (Simpfendorfer et al., 2020a)

Medium-sized coastal shark that is found in clear tropical waters from the surface to depths of around 280 m and common around coral reefs, particularly near drop-offs and fringing coral reefs. Its distribution is patchy in continental shelf waters. *Carcharhinus amblyrhynchos* is found in the following FAO Fishing Areas: 51, 57, 77, 61, 81, 71). Range countries and territories are: American Samoa; Australia; Bahrain; Bangladesh; British Indian Ocean Territory (Chagos Archipelago); Cambodia; China; Christmas Island; Cocos (Keeling) Islands; Comoros; Cook Islands; Disputed Territory (Spratly Is., Paracel Is.); Djibouti; Ecuador (Galápagos); Egypt; Eritrea; Federated States of Micronesia; Fiji; French Polynesia; Guam; Hong Kong SAR; India; Indonesia; Islamic Republic of Iran (henceforth Iran); Iraq; Japan; Kenya; Kiribati; Kuwait; Madagascar; Malaysia (Peninsular Malaysia, Sarawak, Sabah); Maldives; Marshall Islands; Mayotte; Mozambique; Myanmar; Nauru; New Caledonia; Niue; Northern Mariana Islands; Oman; Pakistan; Palau; Papua New Guinea; Philippines; Qatar; Réunion; Samoa; Saudi Arabia; Singapore; Solomon Islands; Somalia; South Africa (KwaZulu-Natal); Sri Lanka; Sudan; Taiwan POC; Thailand; Timor-Leste; Tokelau; Tonga; Tuvalu; United Arab Emirates; United Republic of Tanzania (henceforth Tanzania); United States Minor Outlying Islands; Vanuatu; Viet Nam; Wallis and Futuna; Yemen

Borneo Shark *Carcharhinus borneensis*

Critically Endangered A2cd (assessed 2020, ver 3.1) (Dulvy et al., 2021a)

Inhabits coastal bays and estuaries, although depth range is unknown as it has only been collected from fish markets. *Carcharhinus borneensis* is found in the following FAO Fishing Areas: 61, 71. Extant range countries and territories are: Indonesia (Kalimantan); Malaysia (Sarawak).

Pacific Smalltail Shark *Carcharhinus cerdale*

Critically Endangered A2bcd (assessed 2019, ver 3.1) (Pollom et al., 2020d)

Occurs in the Eastern Central and Southeast Pacific from the Gulf of California, Mexico to Peru (Castro 2011) and inhabits coastal areas and estuaries in the Eastern Central and Southeast Pacific from the Gulf of California to Peru from inshore to a depth of 40 m. *Carcharhinus cerdale* is found in the following FAO Fishing Areas: 87, 77. Extant range countries and territories are: Colombia; Costa Rica; Ecuador; El Salvador; Guatemala; Honduras; Mexico; Nicaragua; Panama; Peru.

Whitecheek Shark *Carcharhinus dussumieri*

Endangered A2d+3d (assessed 2018, ver 3.1) (Simpfendorfer et al., 2019)

Occurs primarily in the Western Indian Ocean from at least the Arabian/Persian Gulf to the southeastern coast of India, is common in inshore waters over soft substrates at depths of 0–100 m (Weigmann, 2016). *Carcharhinus dussumieri* is found in the following FAO Fishing Areas: 51, 57. Extant range countries and territories are: Bahrain; India; Iran; Iraq; Kuwait; Oman; Pakistan; Qatar; Saudi Arabia; Sri Lanka; United Arab Emirates.

Pondicherry Shark *Carcharhinus hemiodon*

Critically Endangered C2a(i) (assessed 2020, ver 3.1) (Kyne et al., 2021a)

This species has a wide historic range from Oman to southern China but known records are scattered, and it has only been reliably verified from a handful of countries. It appears to occur in shallow coastal waters from 10–150 m depth, and has also been reported to enter rivers, although this has not been verified. *Carcharhinus hemiodon* is found in the following FAO Fishing Areas: 57, 57, 61, 71. Possibly extant range countries and territories are: China; India; Indonesia (Kalimantan, Jawa); Malaysia; Oman; Pakistan.

Smoothtooth Blacktip Shark *Carcharhinus leiodon*

Endangered A2cd+3cd (assessed 2017, ver 3.1) (Simpfendorfer et al., 2017)

Endemic to the Arabian Seas region and was only rediscovered in 2009, is coastally distributed. Further records of the Smoothtooth Blacktip Shark from the western Arabian Sea indicate that adults are present in this region

throughout the year. *Carcharhinus leiodon* is found in the following FAO Fishing Areas: 51. Extant range countries and territories are: Bahrain; Kuwait; Oman; Qatar; United Arab Emirates; Yemen (South Yemen).

Dusky Shark *Carcharhinus obscurus*

Endangered A2bd (assessed 2018, ver 3.1) (Rigby et al., 2019a)

Coastal and pelagic shark with a patchy distribution in tropical and warm temperate seas from the surface down to depths of 500 m. Found on continental and insular shelves, from the shoreline to the outer reaches of the continental shelf and adjacent oceanic waters, where it is generally a mid-level to bottom feeder. It occurs from the surf zone to well offshore. *Carcharhinus obscurus* is found in the following FAO Fishing Areas: 31, 87, 61, 77, 51, 71, 34, 81, 57, 41, 47, 21, 27, 37. Extant range countries and territories are: Algeria; Australia; Bahamas; Belize; Brazil; Cabo Verde; China; Colombia; Costa Rica; Cuba; El Salvador; Eritrea; French Guiana; Guatemala; Guyana; Haiti; Honduras; Italy; Japan; Libya; Madagascar; Mexico; Mozambique; New Caledonia; New Zealand; Nicaragua; Panama; Senegal; South Africa; Spain (Canary Is., Spain (mainland)); Sudan; Suriname; United States; Uruguay; Venezuela; Viet Nam; Western Sahara; Yemen.

Lost Shark *Carcharhinus obsoletus*

Critically Endangered A2d; D (assessed 2020, ver 3.1) (Dulvy et al., 2020)

Small requiem shark from the southern South China Sea (Gulf of Thailand, Viet Nam, and Sarawak in Malaysian Borneo) in the Western Central Pacific, but it may have had a wider historic distribution in the southern South China Sea. Habitat and depth range are unknown—it is thought likely to be extinct. *Carcharhinus obsoletus* was found in the following FAO Fishing Areas: 71. The species is Possibly Extinct in Malaysia (Sarawak); Thailand; Viet Nam.

Caribbean Reef Shark *Carcharhinus perez*

Endangered A2bcd (assessed 2019, ver 3.1) (Carlson et al., 2021a)

Tropical inshore shark found mainly on coral reefs on continental and insular shelves from the surface to a depth of 378 m. *Carcharhinus perez* is found in the following FAO Fishing Areas: 31, 41. Extant range countries and territories are: Anguilla; Antigua and Barbuda; Aruba; Bahamas; Barbados; Belize; Bermuda; Bonaire, Sint Eustatius and Saba; Brazil; Cayman Islands; Colombia; Costa Rica; Cuba; Curaçao; Dominica; Dominican Republic; French Guiana; Grenada; Guadeloupe; Guatemala; Guyana; Haiti; Honduras; Jamaica; Mexico; Montserrat; Nicaragua; Panama; Puerto Rico; Saint Barthélemy; Saint Kitts and Nevis; Saint Lucia; Saint Martin (French part); Saint Vincent and the Grenadines; Sint Maarten (Dutch part); Suriname; Trinidad and Tobago; Turks and Caicos Islands; United States; Venezuela; British Virgin Islands; US Virgin Islands.

Sandbar Shark *Carcharhinus plumbeus*

Endangered A2bd (assessed 2020, ver 3.1) (Rigby et al., 2021b)

Demersal and pelagic environments in tropical and temperate seas on the continental shelf from close inshore to a depth of 280 m. It occurs in shallow waters associated with bays, estuaries and harbours and offshore on oceanic banks. Some stocks make extensive seasonal migrations, such as those in the Northwest Atlantic and South Africa. *Carcharhinus plumbeus* is found in the following FAO Fishing Areas: 31, 61, 77, 37, 51, 71, 34, 81, 57, 41, 47, 21, 27. Extant range countries and territories are: Albania; Algeria; Anguilla; Antigua and Barbuda; Argentina; Aruba; Australia; Bahamas; Bahrain; Barbados; Belize; Benin; Bonaire, Sint Eustatius and Saba; Bosnia and Herzegovina; Brazil; Brunei Darussalam; Cabo Verde; Cameroon; China; Colombia; Congo; Costa Rica; Croatia; Cuba; Curaçao; Cyprus; Côte d'Ivoire; Democratic People's Republic of Korea; Djibouti; Dominica; Dominican Republic; Egypt; Equatorial Guinea; Eritrea; France (Corsica); French Guiana; Gabon; Gambia; Greece; Grenada; Guadeloupe; Guatemala; Guinea; Guinea-Bissau; Guyana; Haiti; Honduras; India; Indonesia; Iran; Iraq; Israel; Italy; Jamaica; Japan; Kuwait; Lebanon; Libya; Madagascar; Malaysia (Sabah, Sarawak); Malta; Martinique; Mauritius; Mexico; Montenegro; Montserrat; Morocco; Mozambique; Myanmar; New Caledonia; Nicaragua; Nigeria; Norfolk Island; Oman; Palestine, State of; Panama; Papua New Guinea; Portugal; Puerto Rico; Qatar; Republic of Korea; Saint Kitts and Nevis; Saint Lucia; Saint Martin (French part); Saint Vincent and the Grenadines; Sao Tome and Principe; Saudi Arabia; Senegal; Seychelles; Sint Maarten (Dutch part); Slovenia; Solomon Islands; South Africa; Spain (Canary Is.); Sri Lanka; Sudan; Suriname; Syrian Arab Republic; Taiwan, Province of China; Tanzania; Thailand; Togo; Trinidad and Tobago; Tunisia; Turkey; Turks and Caicos Islands; United Arab Emirates; United States (Hawaiian Is.); Uruguay; Venezuela; Viet Nam; British Virgin Islands; US Virgin Islands.; Western Sahara; Yemen.

Smalltail Shark *Carcharhinus porosus*

Critically Endangered A2d (assessed 2019, ver 3.1) (Pollom et al., 2020a)

Central and South American coastal requiem shark that inhabits muddy inshore areas and estuaries down to a depth of 84 m. The species is strongly associated with mangrove forests, which can be considered as essential habitat for the species on the basis of probability of occurrence and patterns of habitat use. *Carcharhinus porosus* is found in the following FAO Fishing Areas: 31, 41. Range countries and territories are: Belize; Brazil (Amapá, Maranhão, Pará); Colombia; Costa Rica; French Guiana; Guatemala; Guyana; Honduras; Mexico; Nicaragua; Panama; Suriname; United States; Venezuela.

Night Shark *Carcharhinus signatus*

Endangered A2bd (assessed 2019, ver 3.1) (Carlson et al., 2021b)

Commonly caught in pelagic fisheries. Occurs in the Northwest, Western Central, and Southwest Atlantic from New York, USA to Río Negro, Argentina, including the Gulf of Mexico and Caribbean Islands and in the Eastern Central and Southeast Atlantic from Senegal to Namibia. *Pelagic and semi-oceanic on the outer continental shelf to a depth of 600 m but mostly, specimens have been collected from 26–365 m* (Carlson et al., 2008, Castro 2011). *Carcharhinus signatus* is found in the following FAO Fishing Areas: 31, 47, 41, 34, 21. Extant range countries and territories are: Angola; Anguilla; Antigua and Barbuda; Argentina; Aruba; Bahamas; Barbados; Belize; Benin; Bermuda; Bonaire, Sint Eustatius and Saba; Brazil; Cameroon; Cayman Islands; Colombia; Colombia (Colombia (mainland), Colombian Caribbean Is.); Costa Rica; Cuba; Curaçao; Côte d'Ivoire; Democratic Republic of the Congo; Dominica; Dominican Republic; French Guiana; Gabon; Gambia; Ghana; Grenada; Guadeloupe; Guatemala; Guinea; Guinea-Bissau; Guyana; Haiti; Jamaica; Liberia; Martinique; Mexico; Montserrat; Nicaragua; Nigeria; Puerto Rico; Saint Barthélemy; Saint Kitts and Nevis; Saint Lucia; Saint Martin (French part); Saint Vincent and the Grenadines; Senegal; Sierra Leone; Sint Maarten (Dutch part); Suriname; Togo; Trinidad and Tobago; Turks and Caicos Islands; United States; Uruguay; Venezuela; British Virgin Islands; US Virgin Islands.

Ganges Shark *Glyphis gangeticus*

Critically Endangered A2cd; C2a(i) (assessed 2021, ver 3.1) (Rigby et al., 2021a)

Restricted to freshwater, estuarine and occasionally adjacent nearshore systems in Australasia and South and South-East Asia, at depths of 0–50 m. Restricted to turbid waters in large rivers, estuaries; also adjacent coastal areas during the monsoon, when salinity is reduced. *Juvenile and subadult individuals generally occur in rivers while adults are generally coastal and marine.* *Glyphis gangeticus* is found in the following FAO Fishing Areas: 51, 57, 71. Extant range countries and territories are: Bangladesh; India; Malaysia.

Daggernose Shark *Isogomphodon oxyrinchus*

Critically Endangered A2bcd (assessed 2019, ver 3.1) (Pollom et al., 2020b)

Occurs in the Western Central and Southwest Atlantic from Trinidad and Tobago and eastern Venezuela to the Brazilian Amazon coast. Inhabits inshore waters in turbid estuaries, river mouths, and shallow banks at depths of 4–40 m, and has also been recently recorded in freshwater. *Isogomphodon oxyrinchus* is found in the following FAO Fishing Areas: 31, 41. Range countries and territories are: Brazil; French Guiana; Guyana; Suriname; Trinidad and Tobago; Venezuela.

Broadfin Shark *Lamiopsis temminckii*

Endangered A2d (assessed 2020, ver 3.1) (Dulvy et al., 2021c)

Ranges from Pakistan to Thailand in the northern Indian Ocean. It is found inshore on the continental shelf in depths of less than 50 m. *Lamiopsis temminckii* is found in the following FAO Fishing Areas: 51, 57. Extant range countries and territories are: Bangladesh; India; Myanmar; Pakistan.

Borneo Broadfin Shark *Lamiopsis tephrodes*

Endangered A2d (assessed 2020, ver 3.1) (Dulvy et al., 2021b)

Found inshore on the continental shelf in depths of less than 50 m, associated with turbid estuarine waters in the Western Central and Northwest Pacific. *Lamiopsis tephrodes* is found in the following FAO Fishing Areas: 71, 61. Extant range countries and territories are: Cambodia; China; Indonesia; Malaysia; Thailand; Viet Nam.

Whitenose Shark *Nasolamia velox*

Endangered A2cd (assessed 2019, ver 3.1) (Pollom et al., 2020c)

This species occurs in the Eastern Central and Southeast Pacific from Baja California, Mexico, to Peru in estuaries and over the continental shelf to a depth of 192 m. *It is typically encountered in waters 15–24 m deep* (Ebert et al., 2013, Weigmann 2016). *Nasolamia velox* is found in the following FAO Fishing Areas: 87, 77. Extant

range countries and territories are: Colombia; Costa Rica; Ecuador (Ecuador (mainland), Galápagos); El Salvador; Guatemala; Honduras; Mexico; Nicaragua; Panama; Peru.

Sharptooth Lemon Shark *Negaprion acutidens*

Endangered A2bd (assessed 2020, ver 3.1) (Simpfendorfer et al., 2021a)

Coastal shark that is widespread throughout the Indo-West and Central Pacific. It is demersal in shallow inshore and offshore waters to at least 90 m depth and is often found on and around coral reefs and on sandy plateaus near coral. It is also known to occur around and within the mangrove forests in certain areas, which may be used as nursery grounds (Bonfil, 2003). *Negaprion acutidens* is found in the following FAO Fishing Areas: 51, 57, 77, 61, 71. Extant range countries and territories are: American Samoa; Australia; Bahrain; Bangladesh; British Indian Ocean Territory; Brunei Darussalam; Cambodia; China; Comoros; Cook Islands; Disputed Territory; Djibouti; Egypt; Eritrea; Federated States of Micronesia; Fiji; French Polynesia; Guam; Hong Kong SAR; India; Indonesia; Iran; Japan; Kenya; Kiribati; Kuwait; Madagascar; Malaysia; Maldives; Marshall Islands; Mayotte; Mozambique; Myanmar; Nauru; New Caledonia; Niue; Northern Mariana Islands; Oman; Pakistan; Palau; Papua New Guinea; Philippines; Qatar; Réunion; Samoa; Saudi Arabia; Seychelles; Singapore; Solomon Islands; Somalia; South Africa; Sri Lanka; Sudan; Taiwan POC; Tanzania; Thailand; Timor-Leste; Tokelau; Tonga; Tuvalu; United Arab Emirates; Vanuatu; Viet Nam; Yemen.

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

Biology

All species included in this proposal are members of the family Carcharhinidae (requiem sharks), which currently includes 56 species. Most are viviparous with a yolk sac placenta; litters range in size from just one or two pups to (rarely) over 100. They are active, strong swimmers. Some species are “ram-ventilators” needing to swim continually to oxygenate their gills, while others are capable of resting motionless for extended periods on the bottom. Many are more active at night, or dawn and dusk, than during the daytime. Some are solitary or socialise in small groups, and some are social schooling species.

A simple but robust estimator of natural mortality (M) for sharks and rays is the reciprocal of average lifespan ($M=1/w$) where average lifespan ‘ w ’ is the generation length of the species (Pardo and Dulvy, 2016; Pardo et al., 2016). Hence, these CITES mortality thresholds can be back converted to calculate the corresponding generation length productivity thresholds ($GL = M^{-1}$) and these are as follows: high productivity (natural mortality > 0.5 or a generation length of < 2 years), medium productivity (natural mortality $0.2-0.5$ or generation lengths $2-5$ years), and low productivity (natural mortality < 0.2 or generation length > 5 years).

Table 3. Life history characteristics of the lead species. Estimated generation length is from the IUCN Red List assessment for each species.

| Species | Maximum Size (cm) | Size of maturity (cm) | Litter size (pups) | Frequency of reproduction /gestation period | Estimated generation length (years) | Productivity |
|-------------------------|-------------------|--------------------------------|--------------------|---|-------------------------------------|--------------|
| <i>C. acronotus</i> | 137 TL | M – 97–110 TL, F – 101–120 TL | 1–6 | Annual to Biennial | 8.5–10.5 | Low |
| <i>C. amblyrhynchos</i> | 265 | M – 130–145 TL, F 120–142 TL | 1–6 | Biennial | 14.5 | Low |
| <i>C. borneensis</i> | 70 TL | M – 59–62 TL, F – 61–65 TL | 2–9 | Biennial, 1 year gestation | 9 | Low |
| <i>C. cerdale</i> | 140 TL | M 100 TL | | | 9 | Low |
| <i>C. dussumieri</i> | 100 TL | M – 72 TL, F 80 TL | 2–5 | Annual | 4 | Medium |
| <i>C. hemiodon</i> | 102 TL | | | | 9 | Low |
| <i>C. leiodon</i> | 165 TL | F – 131 TL | 4–6 | | 8.25 | Low |
| <i>C. obscurus</i> | 420 | M – 265–280, F– 257–310 | 7 | Biennial, 18–24 month gestation | 29.8–38 | Low |
| <i>C. obsoletus</i> | 100 TL | | | Likely biennial | 9 | Low |
| <i>C. perezii</i> | 295 TL | M – 150–179 TL, F – 180–190 TL | 3–6 | Biennial | 9.6 | Low |
| <i>C. plumbeus</i> | 240–300 TL | M – 123–180 TL, F – 129–190 TL | 1–14 (5–12 common) | Biennial to triennial (varies regionally) | 20–26 | Low |

| Species | Maximum Size (cm) | Size of maturity (cm) | Litter size (pups) | Frequency of reproduction /gestation period | Estimated generation length (years) | Productivity |
|--------------------------------|-------------------------|--------------------------------|--------------------|---|-------------------------------------|---------------|
| <i>C. porosus</i> | 150 | M– 70 TL, F– 71 TL | 2–7 | Biennial, 12–month gestation | 9 | Low |
| <i>C. signatus</i> | 276 TL | M – 185–190 TL, F – 200–205 TL | 4–15 | Annual | 16.5 | Low |
| <i>G. gangeticus</i> | 275 cm, possibly larger | M – 178 | | | 18 | Low |
| <i>Isogomphodon oxyrinchus</i> | 160 TL | M – 103 TL, F – 115 TL | 3–8 | Biennial | 9 | Low |
| <i>Lamiopsis temminckii</i> | 178 TL | M – 137 TL, F – 143 TL | 4–8 | | 6.5 | Medium-to-Low |
| <i>L. tephrodes</i> | 157 TL | M – 114 TL, F – 130 TL | 4–8 | | 6.5 | Medium-to-Low |
| <i>Nasolamia velox</i> | 150 TL | | 5 | | 9 | Low |
| <i>Negaprion acutidens</i> | 340 TL | M and F – 220–240 TL | 6–12 | Biennial | 16.5 | Low |

Population size

Global population sizes are unknown. The IUCN Red List assessment for *Glypis gangeticus* estimated the number of mature individuals as very small (< 250) (Rigby et al., 2021a). The assessment for *C. hemiodon* also estimated that it was very small (<250 with no subpopulation >50 mature individuals) (Kyne et al., 2021a). The remaining population size for *C. obsoletus* is suspected to be fewer than 50 individuals and is inferred to be continuing to reduce due to actual or potential fishing levels (Dulvy et al., 2020).

Population trends

Members of the family *Carcharhinidae* were recently assessed on the IUCN Red List from 2017 to 2021.

Carcharhinus acronotus

Trawl survey using multiple gear (1989–2014) exhibited a steady annual rate of reduction of 6.1% (potential population reduction of 82.2% over 3 generations). Stock assessments from the U.S. South Atlantic and Gulf of Mexico had annual rates of reduction of 1.8% and 2.1% to 2009. An increasing trend of 4.8% per year was found from a fishery independent bottom longline survey covering the same geographic area from 1995–2018. Taken together, *C. acronotus* is estimated to have decreased by 2.8% per year over 3 generation lengths (26 years). The corresponding probabilities for population reduction fall between 57% probability of 50–79% and 43% probability of 30–49%. *In the Southwest Atlantic there are no population trend estimates for this species.* In Venezuela, it is suspected that unmanaged fisheries have caused a reduction in population size. In the Guianas and northern Brazil, it is suspected that reductions of this species have occurred. For example, this species was known to be much more common in the State of Maranhão a decade ago. In the State of Pará, this species was formerly very common, but today there are very few adults encountered in landings. Demographic analysis from Pernambuco State indicates a 44% reduction over three generation lengths (26 years) due to mortality in the gillnet fishery, and that fishery now lands mostly juveniles. In Ceará State, landings of this species have decreased by 64% between 1998–1999 and by 2015–2016 had decreased by 78% since 1998–1999. It is uncommon in Central America, Cuba, and the Eastern Caribbean.

It is suspected that this species has undergone a population reduction of 50–79% over the last three generation lengths (26 years) due to levels of exploitation. *It is assessed as Endangered A2bd. Overall trends suggest a depletion to 19–62% over the past three generation lengths (26 years), or a tripling of the population over the next three generation lengths, based on an annual increase of 4.8%. If this reduction was projected into the near future (10 years), it suggests a depletion to 10–52% for this low-productivity species (Carlson et al., 2021c).*

Carcharhinus amblyrhynchus

The Global FinPrint project sampled in countries containing 88.6% of the coral reefs within the species' global historic range, creating the largest and most recent dataset available to assess the status of this species. Reef-level depletion estimates were aggregated, weighted by jurisdictional coral reef area (relative to global coral reef area), to produce an estimate of global depletion. This research concluded that the grey reef shark has undergone a global population reduction of 59% (standard error 51.9–66.5%) in the last three generation lengths (44 years). *As such it is concluded, based on the Global Finprint data, that C. amblyrhynchus has undergone a*

population reduction of 50–79% in the last three generation lengths (44 years) owing to levels of exploitation and reduction in habitat quality. It is assessed as Endangered A2bcd (Simpfendorfer et al., 2020a).

There has been mixed evidence of the severity of reduction in Chagos Archipelago: a decrease to 10% of 1975 levels over 30 years was reported by Graham et al. (2010). However this information has been contested and it is believed to have decreased to 79% of its original abundance (Ferretti et al., 2018). On Australia's Great Barrier Reef there are varying reports of population decreases and recovery. Robbins et al. (2006) reported that *C. amblyrhynchos* was suffering from ongoing collapse, with annual reductions of between 7 and 17%. However, subsequent research has shown that populations have not decreased as dramatically as first estimated (Heupel et al., 2009), and Espinoza et al. (2014) demonstrating that rezoning of the reef had led to increasing abundance in areas protected from fishing.

Considered together, these data suggest a depletion to 41% globally over the past three generation lengths (44 years). If this reduction was projected into the near future (10 years), it suggests a depletion to 33% for this low-productivity species. In some regions (Viet Nam, Tanzania, Sri Lanka, Qatar, Japan, Indonesia, India, Taiwan POC, Guam, the Philippines, Malaysia, Saudi Arabia, and Vanuatu), the species appears to have decreased to 25% over the past three generation lengths (44 years) and could decrease further to 18% in the near future (10 years).

Carcharhinus borneensis

This species was historically known only from five specimens, the last of which was collected in 1937 and the species was presumed extinct until recently rediscovered in 2004 with numerous specimens collected from Mukah, Sarawak (Malaysian Borneo). Overall, *C. borneensis* is suspected to have undergone a population reduction of >80% over the past three generation lengths (24 years) due to a reduction in habitat quality and actual or potential fishing levels. It is assessed as Critically Endangered A2cd (Dulvy et al., 2021a).

Carcharhinus cerdale

In the Mexican Pacific, this species was relatively common in ichthyological collections until the 1980s. There were very few records in the late 1990s and early 2000s, and there are no records since a 2001 fishing survey. Targeted fisheries for sharks including this species off Mazatlán were in operation in the early 1960s, and this species was one of the most regularly caught sharks. Overall, in Mexico, records of this species were relatively common from the 1950s to the 1980s, then became increasingly rare through the 1990s and early 2000s, and the last confirmed record is from 2001. There was a decrease in the relative abundance of *C. cerdale* in the industrial shrimp trawl fishery between 1995 (0.24 individuals/hour) and 2004 (0.02 individuals/hour), the equivalent of a >99% population reduction over three generations (24 years). There was also a decrease in the average size of individuals caught from 41 cm TL in 1995 to 38 cm TL in 2004. This species is still recorded in Pacific Colombian artisanal fisheries but with less frequency than in the past.

Castro (2011) resurrected *C. cerdale*, considering it distinct from *C. porosus*. The Pacific Smalltail Shark is inferred to have undergone a population reduction of more than 80% over the past three generations (27 years) based on levels of exploitation. It is assessed as Critically Endangered A2bcd (Pollom et al., 2020d).

Carcharhinus dussumieri

To date, there have been no dedicated surveys or population estimates for this species. A 1937–1938 survey recorded *C. dussumieri* as by far the most common species of shark in the Arabian/Persian Gulf (mostly Iranian coast). It was reported as common around Bahrain in surveys carried out from 1974–1978 and represented almost 3% of elasmobranchs landed by number in more recent surveys. This species is commonly caught in Iran and is considered the most abundant species. Overall, it makes up about 60% of shark landings by number in Iran (Arabian/Persian Gulf and Sea of Oman) and continues to be an important part of the catch. In Kuwait, this species also represented 22% (in 2008) and 20% (in 2011) of elasmobranch landings by number. In Qatar, it comprises 26% of elasmobranch landings by number. In the United Arab Emirates this species accounts for 4.5% of shark landings by number and <1% of sharks traded from Oman to the UAE. However, in Oman, there have only been nine specimens of this species landed from data collected over five years of landing site surveys along the coast. In Pakistan, this species was among the most common species caught in gillnet fisheries in the 1980s but is currently uncommon. *C. dussumieri* is one of the major species contributing to Indian fisheries. In 2003–2004, 58 t were recorded from gillnet fisheries along the southwest coast. However, *C. dussumieri* was not observed in surveys of landings in Porbandar (Gujarat), Sasoan Dock (Mumbai), and Malvan (Maharashtra) undertaken in 2014–2015. This species was also reported as one of the dominant species landed by various fishing gears along the coast of Tamil Nadu (southeast coast of India in the 1980s and 1990s). However, more recent landing site surveys (2002–2006) along the eastern Indian coast (Chennai) failed to record it. A reduction in abundance of over 50–70% based on catch levels has occurred in India. This species has not been reported from over a year of landing site surveys along the coast of Sri Lanka.

The abundance of this species appears to have followed two different trends. In the Arabian/Persian Gulf, it remains an abundant species dominating landings of many countries. However, in the Arabian Sea greater fishing

intensity has resulted in suspected reductions of at least 50–70% in some areas. With increasing fishing pressure across the Arabian Seas region, it is suspected that this species has decreased by at least 50–70% over the past three generations (~12 years) and that these reductions are likely to be ongoing. Overall, a suspected population decrease of at least 50–70% over the past three generations (12 years) and further population reduction is suspected over the future three generation lengths (2018–2030) based on current levels of exploitation, as such this species is assessed as *Endangered A2d+3d* (Simpfendorfer et al., 2019).

Carcharhinus hemiodon

There are no verified records of this species since 1960 despite extensive surveys across its range, likely due to intensive and unregulated coastal fisheries. Given a lack of verifiable records since 1960, it is considered that the major population reduction would have occurred prior to the last three generation period (estimated at 24 years from a congener), therefore an assessment under criterion A is not appropriate. However, given the lack of records, the number of mature individuals is assumed to be <250 with no subpopulation >50 mature individuals, and the species is assessed as *Critically Endangered (C2a(i))* (Kyne et al., 2021a).

Carcharhinus leiodon

Endemic to the Arabian Seas region and only rediscovered in 2009 in Kuwait landings, with subsequent confirmed records across the Gulf and in Oman (Moore et al., 2013). Overall, there is a limited number of specimens reported. Therefore, although the status of *C. leiodon* remains unknown, based on the significant decrease in other similar species in the region, a population reduction of 50–80% is suspected over a period of three generation spans (~25 years). A further population reduction is suspected over three generation lengths (2017–2042) based on current levels of exploitation. As such it is assessed as *Endangered under criterion A2cd+3cd* (Simpfendorfer et al., 2017).

Carcharhinus leiodon is morphologically very similar to *C. limbatus*, *C. sorrah*, and *C. amblyrhynchoides* and there is likely to have been confusion in species identification across its potential range. Its recent rediscovery and redescription means that historically it has likely been under-recorded.

Carcharhinus obscurus

In the Northwest Atlantic a stock assessment considered overfishing has been occurring since the mid-1980s. Prohibition in catches in 2000 has reduced, but not ceased, overfishing. The trend analysis of the Northwest Atlantic relative biomass for 1960–2015 (56 years) revealed annual rates of reduction of 2.6%, consistent with an estimated median reduction of 89.9% over three generation lengths (89.4 years). In the Eastern Atlantic, this species is known from Cabo Verde, off the coast of West Africa, where it was one of the most commonly captured species on longline surveys in 1982, but by the late 2000s was infrequently caught (Stobberup, 2005; Diop and Dossa, 2011). *C. obscurus* could also be expected to be found further along the West African coastline (Ebert et al., 2013). Given the intense coastal shark fisheries in this region there is concern that this species may have disappeared from this large part of its Eastern Atlantic distribution.

The trend analysis of the Western Indian Ocean catch-per-unit-effort (CPUE) for 1978–2003 (25 years) revealed annual rates of reduction of 0.9%, consistent with an estimated median reduction of 60.9% over three generation lengths (114 years), with the highest probability of 50–79% reduction over three generation lengths. In the Eastern Indian Ocean, the stock was previously subject to overfishing and is now considered recovered, as confirmed by a stock assessment (in preparation) which indicates that the stock should recover under the current level of fishing mortality (SAFS, 2018; Braccini and O'Malley, 2018). Trend analysis of CPUE for 1975–2015 (41 years) revealed annual rates of reduction of 3.8%, consistent with an estimated median reduction of 98.7% over three generation lengths (114 years), with the highest probability of >80% reduction over three generation lengths. This probability of a high level of reduction is over a very long period of three generations; it has incorporated the reductions from historically higher abundances and projected an estimated trend based on those reductions for a considerable period beyond that of the time-series and is thus indicative of historic declines rather than the stable and increasing trend since 2006 (Rigby et al., 2019a).

The global estimated median reduction was 75.8%, with the highest probability of >80% reduction over three generation lengths (89.4–114 years). Overall, expert judgement elicitation inferred global population reduction was 50–79% over three generations (89.4–114 years) based on slow recovery in the Eastern Indian Ocean, prohibition in the Northwest Atlantic that is reducing catches, and actual levels of exploitation that likely result in steep suspected reduction where the species is subject to unmanaged fisheries. Therefore, *C. obscurus* is assessed as *Endangered A2bd* (Rigby et al., 2019a).

Considered together, these data suggest a global depletion to 24% globally over the past three generation lengths (89.4–114 years). If this decrease was projected into the near future (10 years), it suggests a depletion to 21% for this low-productivity species. While there are indications of recovery in some regions, the areas of the three regional data sets (Northwest Atlantic, Eastern and Western Indian Ocean) are relatively small compared to the global distribution, and abundance trends are unknown in parts of its distribution (Rigby et al., 2019a).

Carcharhinus obsoletus

This species is known only from three type specimens recorded from fish landing sites and markets, the last of which was collected in 1934. *Carcharhinus obsoletus* is suspected to have undergone a population reduction of >80% over the past three generation lengths (24 years) and the remaining population size is suspected to be fewer than 50 individuals and is inferred to be continuing to decrease due to actual or potential fishing levels. The weighted probability of extinction of both the threats and records and surveys models combined is 0.77–0.78 and hence the lost shark species is suspected to be Critically Endangered (Possibly Extinct) (Dulvy et al., 2020).

Carcharhinus perezii

Globally, a population reduction of 52.5% (standard error 40.4–64.5%) is estimated assuming that this depletion occurred over the past three generation lengths (29 years), *meeting Endangered A2b* (Carlson et al., 2021a). In Belize relative abundance time-series data based on baited remote underwater video stations (BRUVs) from 2009–2018 indicated that annual abundance decreased by 15.4%, consistent with an estimated median reduction of 99.2% over the past three generation lengths (29 years). *However this species appears to have initially had a stable population up to 2013 based on longline catches* (Bond et al., 2017). Longline catch data from the Bahamas from 1979–1984 and 2011–2013 suggest that the population has been relatively stable, increasing annually by 0.8% over the past three generation lengths (29 years). However, these two short time-series with a 30-year gap should be interpreted with caution; the stability and slow increase may be due to an actual population increase but may also reflect a habitat shift from areas of higher human activity to the sampled area that has less human activity. In Caribbean Colombia, this species is common and increasing. In Venezuela *there are no data but* this species is caught in high numbers. Notable reductions in landings in the State of Maranhão and the Trindade and Martin Vaz archipelago, with a suspected reduction in population size of 30%. This species is suspected to have been lost from coastal Brazil and is now likely only found off *three archipelagos* (Noronhua, Abrolhos, Trindade), as the last confirmed record was from Ceará State in 1987. It was formerly common in places such as São Paulo.

Carcharhinus plumbeus

The stock assessment for the United States Atlantic, Gulf of Mexico, and Caribbean estimate that the spawning stock fecundity had most likely reduced by 66% from virgin levels, *and while the stock was overfished it was not currently experiencing overfishing* (SEDAR, 2017). The trend analysis of the spawning stock fecundity for 1960–2015 (56 years) revealed annual rates of reduction of 2.2%, consistent with an estimated median reduction of 74.5% over three generation lengths (60 years). *The low biological productivity of the species and vulnerability of non-mature animals to fishing gears suggest a stock that cannot support a high level of exploitation, however, the strict limitation on catches in recent years has halted overfishing* (SEDAR, 2017).

In South Africa, trend analysis of the bathy protection programme's CPUE for 1981–2019 (39 years) revealed annual rates of reduction of 3.0%, consistent with an estimated median reduction of 88.9% over three generation lengths (78 years). *It is likely that the declines are influenced by the heavy fishing pressure in the adjacent waters of Mozambique and Tanzania. Over the past decade CPUE has stabilised or increased, although this may have been driven by changes in effort.*

In Australia, the Western Australia is recovering while the stock in east Australia is undefined due to insufficient available information to determine status. *In northwest Australia, fishery independent surveys from 2003 to 2017 showed that catch rates increased from 2008–2017 suggesting the population is beginning to recover, although this area was in an area closed to commercial fishing in 1993* (Braccini et al., 2020).

In the Mediterranean Sea, catches of C. plumbeus have declined significantly. The species was common along the Levantine coast until the 1980s. Historically, the species was regularly observed in fish markets of southern Sicily and was recorded in most coastal areas of the Mediterranean Sea. However, recent records in those markets and areas are sporadic with no observations of pregnant females. The population is suspected to have declined by 50–79% in the Mediterranean Sea over the past three generation lengths (69 years) (Ferretti et al., 2016).

In Taiwan POC, C. plumbeus was historically one of the most abundant species in the commercial shark fishery off northeast waters, where it represented 10% of the annual total shark catch in the 1990s. However, the catches have since declined due to high fishing mortality with a substantial decrease in average sizes also noted from 1991 to 2002 (Joung et al., 2004).

This species is considered uncommon in the Arabian Seas region. There has been a significant increase in coastal fishing effort in some parts of the Arabian Seas region leading to a reduction in the number of shark catches with a suspected population decline of 50–79% for C. plumbeus over the past three generation lengths (78 years) (Jabado et al., 2017). *Its presence was recently confirmed in India and Sri Lanka. Across the regions, C. plumbeus*

is estimated and suspected to have decreased by 50–79% over the past three generation lengths (60–78 years) due to fishing pressure and it is assessed as Endangered A2bd (Rigby et al., 2021b).

While this species has responded well to effective management in countries like Australia and the United States—a relatively small portion of its range—it is clearly susceptible to population reduction given its low productivity and evidence of declines in unmanaged regions. Available information (from the United States and South Africa annual rates of reduction) suggests a depletion to 9–26% in the absence of management over the past three generation lengths (60–78 years). If this reduction was projected into the near future (10 years), it suggests a depletion to 7–21% for this low-productivity species.

Carcharhinus porosus

The probability of catching this shark over time has decreased in all parts of its range from 1970–2015, with particularly drastic reductions in the Gulf of Mexico and South America. In the southern Gulf of Mexico, landings became sparser from the 1980s/1990s to the 2000s, and the area where landings were reported had significantly reduced (Pérez-Jiménez et al., 2012). During the 1980s, this species comprised up to 70% of the total catch weight in the artisanal gillnet fisheries in northern Brazil. Catch rates have decreased from 2.87 kg per hour to 0.43 kg per hour in the 2000s, this is equivalent to a population reduction of 85% over the equivalent of three generation lengths (27 years) (Santana et al., 2020). This shark has undergone a three-fold decrease in catch probability over 30 years due to shrimp trawling and gillnetting in northern Brazil (Feitosa et al., 2020). This species is subjected to intense and largely unmanaged fishing pressure across its range. It is inferred that the Smalltail Shark has undergone a population reduction of >80% over the past three generation lengths (24 years) due to levels of exploitation, and *C. porosus* is assessed as Critically Endangered A2d (Pollom et al., 2020a).

This species is rare throughout its range, both in areas where no baseline information exists and in areas where it was historically common. Based on catch decreases from northern Brazil, this species has been depleted to 16% over the past three generation lengths (24 years). If this rate is projected into the near future (10 years), it suggests a depletion to 8% for this low productivity species.

Carcharhinus signatus

In the Northwest Atlantic, *C. signatus* makes up a significant proportion of the shark bycatch in the pelagic longline fishery and the data from the US pelagic longline fishery spanning from Canada to the Caribbean Sea suggests either very steep or relatively minor decreases depending on the inclusion of the first years of data. US pelagic longline fishery. This analysis for 1995–2018 (24 years) revealed annual rates of reduction of 2.4%, consistent with an estimated median reduction of 79.1% over three generation lengths (50 years). However, there were very few *C. signatus* in the first four years (1992–1995) of the data series, but many more between 1996–2000 (Beerkircher et al., 2002). It seems likely that more *C. signatus* were caught during 1992–1995 but were reported mostly as “unidentified sharks” or were misidentified as other species in the genus *Carcharhinus* (Beerkircher et al., 2002). Nevertheless, starting the analysis in 1996 still suggests a steady reduction in *C. signatus* abundance (9.4% from 1996–2017), which suggests the species is declining despite being prohibited from commercial harvest. In Brazil, while there are no time series specific to this species, reported landings of “Machote”, which includes Silky Shark, decreased by 77% between 2001–2009 and the government subsequently assessed them as regionally vulnerable due to a suspected reduction in population size of >30%. According to the study of demography of the species in Brazil, the population of Night Shark has an annual decrease of 8.1%, with a generation time of 12.1 years. Taking into account the data from this study, in three generations (36 years), the population reduction would be 94.7%.

Overall, the combination of high fishing mortality throughout its range, estimated reduction in the US pelagic longline fishery, and suspected reduction in the Southwestern Atlantic Ocean, *C. signatus* is suspected to have undergone a population reduction of 50–79% over the past three generation lengths (50 years). It is assessed as Endangered A2bd (Carlson et al., 2021b). The United States time series are ambiguous, estimating a depletion to 30–81% of baseline over the past three generation lengths (50 years). The Brazil data suggest depletion to 5% of baseline over the past three generations (36 years), or a recent rate of decrease that would deplete the population to <1% of baseline over the next 10 years, although these data also include Silky Shark.

Glyphis gangeticus

Records of *G. gangeticus* are sparse, and the species is considered extremely rare. It is suspected that *G. gangeticus* has undergone a population reduction of >80% over the past three generation lengths (54 years) due to levels of exploitation and given the rarity of contemporary records, it is estimated that the number of mature individuals of *G. gangeticus* is very small (< 250) with small numbers (<50) of mature adults in each subpopulation with an inferred continuing reduction due to ongoing intensive and unmanaged fishing pressure and habitat degradation across its entire range. The species is considered possibly locally extinct in Pakistan as no evidence of the species has been recorded since 2001–2002 despite extensive searches. Similarly, the species is also considered possibly locally extinct in Sabah (Borneo) and Myanmar. In India, landing site surveys have failed to record this species, although one recent record was confirmed in west Indian waters (Jabado et al., 2018).

Surveys of Bangladeshi fisheries and markets in 2016–2017 identified three records of *G. gangeticus*; one from a landing site and two from fins at shark processing centres. *It is suspected that G. gangeticus has undergone a population reduction of >80% over the past three generation lengths (54 years) due to levels of exploitation and given the rarity of contemporary records, it is estimated that the number of mature individuals of G. gangeticus is very small (< 250) with an inferred continuing decrease due to ongoing intensive and unmanaged fishing pressure and habitat degradation across its entire range. It is assessed as Critically Endangered A2cd; C2a(i) (Rigby et al., 2021a).*

Isogomphodon oxyrinchus

In Trinidad and Tobago, annual shark landings between 1972–1993 were variable but showed a general pattern of reduction over time, which is suspected to have continued to the present day. In northern Brazil, the reduction of this species is well-documented, with individuals having been commonly encountered in landings in the 1980s but becoming increasingly rare up to the present. Although the species is still present there, the few recent records lead to inference of a drastic population reduction. A neonate was captured in late 2016, and a further four specimens were recorded between 2018–2020 (Lessa and Feitosa, 2021). Furthermore, demographic analysis revealed that between 1992–2002 the population decreased at an average rate of 18.4% per year, which is equivalent to a >99% population reduction if scaled over three generations (27 years). *It is assessed as Critically Endangered A2bcd (Pollom et al., 2020b).*

Lamiopsis temminckii

This is a rare and poorly known species. It is considered rare throughout most of its Indian range and rarely observed or reported from commercial fish catches along the Indian coast, except from the northwest of India (Maharashtra region) where it was once considered to be common. Most of the available records are from Mumbai, India; it was once known to be common in this area, but has drastically decreased in the past two decades. In 2003–2004, landings of 513 t of this species were reported from Gujarat to Mumbai. Evidence of catch decreases are now reported with catches reaching 82 t in Mumbai in 2016. Additionally, landing surveys from 2013–2014 in Gujarat and Mumbai only recorded seven specimens. In Pakistan, this species used to be caught as bycatch of trawl fisheries that operated on the inner continental, however, it is now seldom caught and has almost disappeared from commercial catches. In Bangladesh, this species was historically present but only 14 specimens have been collected in recent landings surveys; they were captured in large mesh gillnets at 40–75 m depth.

Reconstructed catches of mainly carcharhinids for the western and northern Indian Ocean infer reductions of 67% when scaled to three generation lengths (20 years) and it is therefore assessed as *Endangered A2d (Dulvy et al., 2021c) for this medium-to-low productivity species. While there are no time series for this species in any part of its range, this species was once common throughout many regions where it is now rarely encountered.*

Lamiopsis tephrodes

This species was recently resurrected and hence there is little information on former and current catches. Reconstructed catches of carcharhinids and other elasmobranchs in the Gulf of Thailand, Indonesia, Malaysia (Peninsular and Sarawak), and China were used to infer population reductions of 76%, 28%, 72%, and 29%, respectively, when they are scaled to the suspected three generation lengths of *L. tephrodes* (20 years). Therefore, it is suspected that *L. tephrodes* has undergone a population reduction of 50–79% over the past three generation lengths (20 years) due to actual or potential fishing levels and it is therefore assessed as *Endangered A2d (Dulvy et al., 2021b).*

Inferred population reductions suggest an average depletion to 49% (ranging from 28–72%) over the past three generation lengths (20 years) for this medium-to-low productivity species. If this reduction was projected into the near future (10 years), depletion could fall to an average of 30% (ranging from 13–62%) from baseline for this species. These data are not species-specific and relate to general reductions of carcharhinid sharks throughout the range of this species.

Nasolamia velox

Landing records of *N. velox* from artisanal fisheries in the Gulf of California indicate a peak catch of about 500 t in 1969 and another smaller peak of over 300 t in the late 1970s and early 1980s. Catch was lower over the following decades, remaining at about 100 t annually through the 1990s and early 2000s. At the end of the time-series, *N. velox* catches increased to around 200 t in 2014. Targeted fisheries for sharks including this species off Mazatlán were already in operation in the early 1960s. In the Gulf of Tehuantepec, Mexico, this species made up a small portion of the catch in artisanal fisheries but was the fourth most captured shark between 1996 and 2003.

In the 1990s, this species was the second-most abundant shark in artisanal fisheries in Guatemala, representing about 12% of the catch. In 2006–2007 landings surveys only 29 individuals were reported. Two years of surveys from 2017–2018 only recorded two individuals. In Costa Rica, 346 trawls between 2008–2012 failed to record

the species even though it was present in the 1980s. A few individuals were recorded in Peruvian landings in the late 1990s, but it has not been recorded since. In Colombia, this species was relatively common in the 1990s, but it has rarely been recorded since.

Overall, due to the level of intense and unmanaged fisheries across its range, combined with an increasing rarity of records, it is suspected that this shark has undergone a population reduction of 50–79% over the past three generations (27 years). It is assessed as *Endangered A2cd* (Pollom et al., 2020c).

Negaprion acutidens

The species was not observed in sufficient numbers in BRUV surveys on coral reefs across its range to estimate levels of population reduction quantitatively. Survey results indicate that the species has been reduced substantially along the continental margins of Asia and Africa and some parts of the Pacific. However, it was regularly observed in Australia, and some island nations in the Western Pacific (e.g., French Polynesia, Palau) and Indian (e.g., Seychelles) Oceans (Simpfendorfer et al., 2021a) and appears to have remained stable. This species is rarely encountered in landing site surveys in Indonesia, the Philippines, and Thailand, and has not been observed in recent surveys in Bangladesh, Myanmar, and Malaysia. In the Arabian Seas region, the species represented 0.33% of shark landings by number in the United Arab Emirates (UAE), and 0.6% of sharks by number traded through the UAE from Oman. This species was recorded as being one of the most commonly landed species in Somalia shark fisheries. However, the species was not recorded from landing site surveys in Kuwait, Bahrain, and Qatar, and only a single historical record was reported in Saudi Arabia. Divers in the Red Sea have reported significant reductions over the past 30–40 years (Simpfendorfer et al., 2021a). It is reportedly uncommon in fisheries in Oman, India and the Maldives with no catch data available (Anderson and Ahmed, 1993, Henderson et al., 2007, Jabado et al., 2015, Simpfendorfer et al., 2021a). In Pakistan, this species used to be caught in large quantities using live baits, however, there has been a ~90% decrease in catches in recent years. It is estimated that *N. acutidens* has undergone a population reduction of 50–79% over the last three generation lengths (50 years) due to levels of exploitation and reductions in habitat quality. It is assessed as *Endangered A2bcd* (Simpfendorfer et al., 2021a).

The interpretation of the level of global population reduction is complicated by the divergent state of this species between island (limited population reduction) and continental margin (large population reductions) portions of its range. This species appears heavily depleted or rare in some areas of its range, and stable or abundant in others.

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

Requiem sharks comprise one-third of the annual global chondrichthyan catch and two-thirds of the shark fin trade of coastal sharks (Sherman et al., 2022), with *Prionace glauca* alone making up 16% of global shark landings in 2017. *Prionace glauca* may now be dominating the meat trade via international commerce to meat markets in Japan, Spain, Taiwan POC, Uruguay, and Brazil (L. Feitosa, in litt., 2022). Some species of carcharhinid are incredibly rare and were not present in surveys of large fin trading hubs in Hong Kong SAR. However, due to their Critically Endangered status and limited geographical range, any unregulated trade in their products is an acute conservation concern.

Three recent fin surveys in international fin trade hubs confirmed the dominance of the family Carcharhinidae in the trade. *P. glauca* (34% of samples), *C. falciformis* (10.06% of samples), and the Blacktip complex (*C. limbatus*, *C. amblyrhynchoides*, *C. leiodon*, and *C. tilstoni*) (4.13% of samples) constituted the top three most-sampled species or species groups in Hong Kong SAR from 2014–2015 (Fields et al., 2018). Another study in Hong Kong SAR and Guangzhou from 2015–2017 found that *P. glauca* (39.01% of samples), *C. falciformis* (12.74% of samples), and *Carcharhinus* spp. (4.40% of samples) constituted the top three most-sampled species or species groups (Cardeñosa et al., 2020a). A recent survey of small, low-value fins from 2018–2019 in Hong Kong SAR found *Rhizoprionodon acutus* (25.3% of samples), *C. sorrah* (10.9% of samples), and the Blacktip complex (10.7% of samples) as three of the top four most sampled species or species groups (Cardeñosa et al., 2020b).

Table 4. Species in the family Carcharhinidae reported in fin trimming surveys in Hong Kong SAR from 2014–2015 (Fields et al., 2018); processed fin trimmings collected from 2015–2017 in Guangzhou, China (GZ) Hong Kong Special SAR (HK) (Cardeñosa et al., 2020a); small, low-value fin samples collected from 2018–2019 in Hong Kong SAR (Cardeñosa et al., 2020b). An * indicates one of the 19 “lead” species in the Proposal. Shaded rows indicate that the species was absent from samples collected in the three surveys. Sources: ⁽¹⁾ Fields et al., 2018; ⁽²⁾ Cardeñosa et al., 2020a; ⁽³⁾ Cardeñosa et al., 2020b.

| Scientific Name | Presence in Fin Trade |
|-------------------------------------|---|
| <i>Carcharhinus acronotus</i> * | 0.19% of samples ¹ ; 0.12% GZ and 0.21% HK samples ² |
| <i>Carcharhinus amblyrhynchos</i> * | 0.31% of samples ¹ ; 0.06% GZ and 0.31% HK samples ² ; 0.2% of samples ³ |
| <i>Carcharhinus borneensis</i> * | |
| <i>Carcharhinus cerdale</i> * | |
| <i>Carcharhinus dussumieri</i> * | 0.31% of samples ¹ ; 0.06% GZ and 0.28% HK samples ² ; 2.3% of samples ³ |
| <i>Carcharhinus hemiodon</i> * | |

| Scientific Name | Presence in Fin Trade |
|--------------------------------------|--|
| <i>Carcharhinus leiodon</i> * | 4.13% of samples defined as Blacktip complex ¹ ; 2.17% GZ and 4.66% HK samples defined as Blacktip complex ² ; 10.7% of samples defined as Blacktip complex ³ |
| <i>Carcharhinus porosus</i> * | 0.04% of samples ¹ ; 0.06% GZ and 0.05% HK species-specific samples ² ; 0.6% of samples ³ |
| <i>Carcharhinus obsoletus</i> * | |
| <i>Carcharhinus obscurus</i> * | 0.88% of samples combined with Galapagos Shark ¹ ; 0.18% GZ and 0.09% HK samples ² |
| <i>Carcharhinus perezi</i> * | |
| <i>Carcharhinus plumbeus</i> * | 0.23% of samples combined with Bignose Shark ¹ ; 0.35% GZ and 0.67% HK samples combined with Bignose Shark ² |
| <i>Carcharhinus signatus</i> * | |
| <i>Glyphis gangeticus</i> * | 0.04% of samples defined as <i>Glyphis</i> spp. ¹ ; 0.05% HK samples defined as <i>Glyphis</i> spp. ² |
| <i>Isogomphodon oxyrhynchus</i> * | |
| <i>Lamiopsis temmincki</i> * | 0.08% of samples ¹ ; 0.07% HK samples ² |
| <i>Lamiopsis tephrodes</i> * | |
| <i>Nasolamia velox</i> * | |
| <i>Negaprion acutidens</i> * | 0.60% of samples ¹ ; 0.46% HK samples ² |
| <i>Carcharhinus altimus</i> | 0.23% of samples combined with Sandbar Shark ¹ |
| <i>Carcharhinus albimarginatus</i> | 0.10% of samples ¹ ; 0.35% GZ and 0.13% HK samples ² |
| <i>Carcharhinus amblyrhynchoides</i> | 4.13% of samples defined as Blacktip complex included this species ¹ ; 0.13% of samples were species-specific ¹ ; 0.08% of HK species-specific samples ² ; 2.16% GZ and 4.66% HK samples defined as Blacktip complex ² ; 10.7% of samples defined as Blacktip complex ³ |
| <i>Carcharhinus amboinensis</i> | 1.13% of samples ¹ ; 0.82% GZ and 1.48% HK samples ² ; 0.4% of samples ³ |
| <i>Carcharhinus cautus</i> | |
| <i>Carcharhinus brachyurus</i> | 1.15% of species-specific samples ¹ ; 0.27% of samples combined with Spinner Shark ¹ ; 0.12% GZ and 0.21% HK species-specific samples ² ; 0.04% GZ samples combined with Spinner Shark ² |
| <i>Carcharhinus brevipinna</i> | 1.15% of species-specific samples ¹ ; 0.04% of samples combined with Bronze Whaler Shark ¹ ; 0.59% GZ and 1.34% HK species-specific samples ² ; 0.04% GZ samples combined with Bronze Whaler Shark ² ; 1.5% of samples ³ |
| <i>Carcharhinus coatesi</i> | 5.3% of samples combined with Blackspot Shark ³ |
| <i>Carcharhinus fitzroyensis</i> | 0.4% of samples ³ |
| <i>Carcharhinus galapagensis</i> | 0.88% of samples combined with Dusky Shark ¹ ; 0.12% GZ and 0.76% HK samples combined with Dusky Shark ² |
| <i>Carcharhinus humani</i> | |
| <i>Carcharhinus isodon</i> | 0.13% of samples ¹ ; 0.09% HK samples ² ; 0.2% of samples ³ |
| <i>Carcharhinus leucas</i> | 1.81% of samples ¹ ; 1.29% GZ and 1.69% HK samples ² ; 1.1% of samples ³ |
| <i>Carcharhinus limbatus</i> | 4.13% of samples defined as Blacktip complex ¹ ; 0.44% of species-specific samples ¹ ; 0.28% species-specific HK samples ² ; 2.17% GZ and 4.66% HK samples defined as Blacktip complex ² ; 10.7% of samples defined as Blacktip complex ³ |
| <i>Carcharhinus macroti</i> | 0.13% of samples ¹ ; 0.08% HK samples ² ; 2.5% of samples ³ |
| <i>Carcharhinus melanopterus</i> | 0.04% of samples ¹ ; 0.18% GZ and 0.09% HK ² ; 0.6% of samples ³ |
| <i>Carcharhinus sealei</i> | 5.3% of samples combined with Australian Blackspot Shark ³ |
| <i>Carcharhinus sorrah</i> | 1.04% of samples ¹ ; 1.58% HK and 0.97% GZ samples ² ; 10.9% of samples ³ |
| <i>Carcharhinus tilstoni</i> | 4.13% of samples defined as Blacktip complex included this species ¹ ; 2.17% GZ and 4.66% HK samples defined as Blacktip complex ² ; 10.7% of samples defined as Blacktip complex ³ |
| <i>Carcharhinus tjtjt</i> | |
| <i>Glyphis garricki</i> | 0.04% of samples defined as <i>Glyphis</i> spp. ¹ ; 0.05% HK samples defined as <i>Glyphis</i> spp. ² ; 0.2% of species-specific samples ³ |
| <i>Glyphis glyphis</i> | 0.04% of samples defined as <i>Glyphis</i> spp. ¹ ; 0.05% HK samples defined as <i>Glyphis</i> spp. and <i>Glyphis glyphis</i> ² |
| <i>Loxodon macrorhinus</i> | 0.04% of samples ¹ ; 0.06% GZ 0.03% HK samples ² ; 0.6% of samples ³ |
| <i>Negaprion brevirostris</i> | 0.10% of samples ¹ ; 0.12% GZ 0.07% HK samples ² |
| <i>Prionace glauca</i> | 34% of samples ¹ ; 36.11% GZ 39.01% HK samples ² |
| <i>Rhizoprionodon acutus</i> | 1.38% of samples ¹ ; 1.82% GZ 0.98% HK samples ² ; 25.3% of samples ³ |
| <i>Rhizoprionodon landii</i> | |

| Scientific Name | Presence in Fin Trade |
|-----------------------------------|--|
| <i>Rhizoprionodon longurio</i> | 0.15% of samples ¹ ; 0.09% HK samples ² |
| <i>Rhizoprionodon oligolinx</i> | 0.14% HK samples ² ; 0.2% of samples ³ |
| <i>Rhizoprionodon porosus</i> | 0.35% of samples combined with Atlantic Sharpnose Shark ¹ ; 0.04% of species-specific samples ¹ ; 0.12% GZ 0.28% HK samples combined with Atlantic Sharpnose Shark ² ; 1.7% of samples ³ |
| <i>Rhizoprionodon taylori</i> | 0.50% of samples ¹ ; 0.06% GZ 0.45% HK samples ² ; 0.4% of samples ³ |
| <i>Rhizoprionodon terraenovae</i> | 0.35% of samples combined with Caribbean Sharpnose Shark ¹ ; 0.04% of species-specific samples ¹ ; 0.12% GZ 0.28% HK samples combined with Caribbean Sharpnose Shark ² |
| <i>Scoliodon laticaudus</i> | 0.08% of samples ¹ ; 0.05% HK samples ² |
| <i>Scoliodon macrorhynchus</i> | 0.12% GZ samples ² |
| <i>Triaenodon obesus</i> | 0.02% of samples ¹ ; 0.12% GZ 0.01% HK samples ² |

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

Fins

Depending on the type of product, identification is not always possible to the species level, with multiple lookalike species within the wider family Carcharhinidae. Lookalikes in the fin trade were determined using a variety of resources that were produced to aid in the implementation of Appendix II listings for sharks. Species were considered "lookalikes" if they were included as "similar species" in identification guides.

Table 5. Species in the family Carcharhinidae that meet the criteria for listing in Annex 2b A of Resolution Conf. 9.24 (Rev CoP17) based on the difficulty of distinguishing their fins from: (a) one of the species already listed in CITES Appendix II or (b) at least one of the species that met Annex 2aA or 2aB of Res. Conf. 9.24 (Rev. CoP17) in this Proposal. Species with * were considered to meet the criteria for Annex2aA; those with ** were considered in this Analysis to meet the criteria for Annex 2aB. Species with † were deemed in this analysis not to meet Annex 2aA or Annex 2aB but were proposed as meeting these criteria in the Supporting Statement. Shaded rows indicate that no "similar species" were included in identification guides for these species. Sources: ⁽¹⁾ FAO, 2016; ⁽²⁾ Abercrombie et al., 2013; ⁽³⁾ Abercrombie and Hernandez, 2017; ⁽⁴⁾ Abercrombie and Jabado, 2022; ⁽⁵⁾ J. Carlson, in litt., 2022; ⁽⁶⁾ D. Grubbs, in litt., 2022; ⁽⁷⁾ L. Feitosa, in litt., 2022.

| Scientific Name | Annex 2b Appendix II listed Species | Annex 2b New proposal Annex 2aA or 2aB |
|--------------------------------------|--|---|
| <i>Carcharhinus acronotus</i> ** | | |
| <i>Carcharhinus amblyrhynchus</i> ** | <i>Carcharhinus longimanus</i> (Oceanic Whitetip Shark) ¹ | <i>Carcharhinus obscurus</i> (Dusky Shark) ¹ |
| <i>Carcharhinus borneensis</i> * | | |
| <i>Carcharhinus cerdale</i> * | | |
| <i>Carcharhinus dussumieri</i> ** | | |
| <i>Carcharhinus hemiodon</i> * | | |
| <i>Carcharhinus leiodon</i> ** | | |
| <i>Carcharhinus porosus</i> * | | |
| <i>Carcharhinus obsoletus</i> * | | |
| <i>Carcharhinus obscurus</i> * | <i>Carcharodon carcharias</i> (Great White Shark) ^{1,2} <i>Sphyrna mokarran</i> (Great Hammerhead) ² ; <i>Carcharhinus falciformis</i> (Silky shark) ^{1,2,3,4} | <i>Carcharhinus plumbeus</i> (Sandbar Shark) ^{1,2} ; <i>Carcharhinus perezii</i> (Caribbean Reef Shark) ^{2,4} ; <i>Carcharhinus signatus</i> (Night Shark) ^{2,4} |
| <i>Carcharhinus perezii</i> † | <i>Sphyrna mokarran</i> (Great Hammerhead) ² ; <i>Carcharhinus falciformis</i> (Silky shark) ^{2,3,4} | <i>Carcharhinus signatus</i> (Night Shark) ^{2,4} ; <i>Carcharhinus obscurus</i> (Dusky Shark) ^{2,4} |
| <i>Carcharhinus plumbeus</i> * | <i>Carcharhinus falciformis</i> (Silky shark) ^{1,2} ; <i>Carcharodon carcharias</i> (Great White Shark) ² | <i>Carcharhinus obscurus</i> (Dusky Shark) ^{1,2} ; <i>Carcharhinus perezii</i> (Caribbean Reef Shark) ² |

| Scientific Name | Annex 2b Appendix II listed Species | Annex 2b New proposal Annex 2aA or 2aB |
|---|--|---|
| | <i>Sphyrna mokarran</i> (Great Hammerhead) ² | |
| <i>Carcharhinus signatus</i> [†] | <i>Sphyrna mokarran</i> (Great Hammerhead) ² <i>Carcharhinus falciformis</i> (Silky Shark) ^{2,3,4,6} | <i>Prionace glauca</i> (Blue Shark) ² ; <i>Carcharhinus obscurus</i> (Dusky Shark) ^{2,4} ; <i>Carcharhinus plumbeus</i> (Sandbar Shark) ² ; <i>Carcharhinus perezii</i> (Caribbean Reef Shark) ^{2,4} |
| <i>Glyphis gangeticus</i> [*] | | |
| <i>Isogomphodon oxyrinchus</i> [*] | | |
| <i>Lamiopsis temminckii</i> [*] | | |
| <i>Lamiopsis tephrodes</i> [†] | | |
| <i>Nasolamia velox</i> [*] | | |
| <i>Negaprion acutidens</i> ^{**} | | |
| <i>Carcharhinus altimus</i> | <i>Carcharhinus falciformis</i> (Silky shark) ² | <i>Carcharhinus signatus</i> ⁶ ; <i>Carcharhinus plumbeus</i> ⁶ |
| <i>Carcharhinus albimarginatus</i> | <i>Carcharhinus falciformis</i> (Silky shark) ¹ <i>Carcharhinus longimanus</i> (Oceanic Whitetip Shark) ¹ | |
| <i>Carcharhinus amblyrhynchoides</i> | | |
| <i>Carcharhinus amboinensis</i> | <i>Carcharodon carcharias</i> (Great White Shark) ¹ | <i>Carcharhinus obscurus</i> (Dusky Shark) ¹ ; <i>Carcharhinus plumbeus</i> (Sandbar Shark) ¹ |
| <i>Carcharhinus cautus</i> | | |
| <i>Carcharhinus brachyurus</i> | | |
| <i>Carcharhinus brevipinna</i> | <i>Sphyrna lewini</i> (Scalloped Hammerhead) ² ; <i>Carcharodon carcharias</i> (Great White Shark) ² | |
| <i>Carcharhinus coatesi</i> | | |
| <i>Carcharhinus fitzroyensis</i> | | |
| <i>Carcharhinus galapagensis</i> | | <i>Carcharhinus obscurus</i> (Dusky Shark) ⁵ |
| <i>Carcharhinus humani</i> | | |
| <i>Carcharhinus isodon</i> | | |
| <i>Carcharhinus leucas</i> | <i>Carcharodon carcharias</i> (Great White Shark) ¹ ; <i>Sphyrna mokarran</i> (Great Hammerhead) ² ; <i>Carcharhinus falciformis</i> (Silky Shark) ^{2,3,4} | <i>Carcharhinus plumbeus</i> (Sandbar Shark) ² ; <i>Carcharhinus obscurus</i> (Dusky Shark) ^{1,2,4} ; <i>Carcharhinus signatus</i> (Night Shark) ^{2,4} ; <i>Carcharhinus perezii</i> (Caribbean Reef Shark) ^{2,4} |
| <i>Carcharhinus limbatus</i> | <i>Sphyrna lewini</i> (Scalloped Hammerhead) ^{2,3,4} <i>Sphyrna zygaena</i> (Smooth Hammerhead) ² ; <i>Alopias</i> spp (Thresher sharks) ² ; <i>Isurus</i> spp (Mako sharks) ² ; <i>Carcharodon carcharias</i> (Great White Shark) ² | <i>Carcharhinus plumbeus</i> (Sandbar Shark) ¹ ; <i>Carcharhinus porosus</i> (Smalltail Shark) ⁷ |
| <i>Carcharhinus macrotis</i> | | |
| <i>Carcharhinus melanopterus</i> | | |
| <i>Carcharhinus sealei</i> | | |
| <i>Carcharhinus sorrah</i> | <i>Carcharhinus falciformis</i> (Silky Shark) ¹ | |
| <i>Carcharhinus tilstoni</i> | | |
| <i>Carcharhinus tjtutjot</i> | | |
| <i>Glyphis garricki</i> | | |
| <i>Glyphis glyphis</i> | | |
| <i>Loxodon macrorhinus</i> | | |
| <i>Negaprion brevirostris</i> | <i>Sphyrna mokarran</i> (Great Hammerhead) ² ; <i>Carcharhinus falciformis</i> (Silky Shark) ² ; | <i>Carcharhinus obscurus</i> (Dusky Shark) ² ; <i>Carcharhinus perezii</i> (Caribbean Reef Shark) ² |

| Scientific Name | Annex 2b Appendix II listed Species | Annex 2b New proposal Annex 2aA or 2aB |
|-----------------------------------|---|--|
| <i>Prionace glauca</i> | <i>Carcharhinus falciformis</i> (Silky Shark) ^{1,2,3} ; <i>Alopias superciliosus</i> (Bigeye Thresher) ³ ; <i>Isurus oxyrinchus</i> (Shortfin Mako Shark) ^{2,3} ; <i>Isurus paucus</i> (Longfin Mako shark) ⁴ ; <i>Lamna nasus</i> (Porbeagle) ⁴ | <i>Carcharhinus signatus</i> (Night Shark) ² |
| <i>Rhizoprionodon acutus</i> | | <i>Carcharhinus porosus</i> (Smalltail Shark) ⁷ |
| <i>Rhizoprionodon landii</i> | | <i>Carcharhinus porosus</i> (Smalltail Shark) ⁷ |
| <i>Rhizoprionodon longurio</i> | | <i>Carcharhinus porosus</i> (Smalltail Shark) ⁷ |
| <i>Rhizoprionodon oligolinx</i> | | <i>Carcharhinus porosus</i> (Smalltail Shark) ⁷ |
| <i>Rhizoprionodon porosus</i> | | <i>Carcharhinus porosus</i> (Smalltail Shark) ⁷ |
| <i>Rhizoprionodon taylori</i> | | <i>Carcharhinus porosus</i> (Smalltail Shark) ⁷ |
| <i>Rhizoprionodon terraenovae</i> | | <i>Carcharhinus porosus</i> (Smalltail Shark) ⁷ |
| <i>Scoliodon laticaudus</i> | | |
| <i>Scoliodon macrorhynchus</i> | | |
| <i>Triaenodon obesus</i> | <i>Carcharhinus longimanus</i> (Oceanic Whitetip Shark) ¹ | |

Meat

The shark meat trade is also increasingly recognised to be a contributory threat to many shark and ray species, although there are limited data on the species composition in the trade. Genetic identification or a comprehensive traceability system would be needed to identify traded meat in those higher capacity countries that trade large quantities of meat, and that is simplest when conducted to the family level.

Other Products

Oil, skins, jaws, and other secondary products are likely being utilised (albeit mostly in domestic markets). Cartilage is also a notable product and is traded internationally. Use and trade of secondary products is common among a range of species and different countries, for example *C. perezi*, products include leather (skin), oil (livers) and fishmeal (from carcasses). In Colombia, *C. perezi* jaws and livers are used for ornaments and oil, respectively, while the meat is only occasionally used as it is not easily marketed. *Carcharhinidae plumbeus* is used to a lesser extent for its skin and liver oil, relative to its fins and meat. *Carcharhinidae signatus* is used for its meat, fins, liver oil, and skin.

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Some species of *Carcharhinidae* have proven to be responsive to effective fisheries management, notably *C. plumbeus* and *C. obscurus* (Rigby et al., 2019a; 2021b).

Additional information

Conservation, management and legislation

Australia and the USA have implemented fishery management measures aimed specifically at reducing *C. obscurus* mortality, and US commercial and recreational fishers are prohibited from retaining the species. South Africa has imposed a recreational bag limit for *C. obscurus*.

In India, *Glyphis gangeticus* is one of 10 species of chondrichthyans protected under Schedule I, Part II A of the Indian Wildlife (Protection) Act, 1972. However, the effectiveness of this measure is unknown, with ongoing issues in enforcement and compliance. In Bangladesh, *G. gangeticus* has been protected since 2012 under Schedule I of the Wildlife (Conservation and Security) Act, 2012, however the effectiveness of this measure is limited due to a general lack of awareness of the protection among fishers and traders. To conserve the population and to permit recovery, a suite of measures will be required that may include species protection, spatial management, bycatch mitigation, and harvest and trade management measures (including international trade measures).

In Brazil, there is specific legislation in place for *Isogomphodon oxyrinchus*, *C. obscurus*, *C. perezi*, *C. plumbeus*, *C. porosus*, *C. signatus*, and *N. brevirostris* which restricts harvest and trade (MMA Ordinance No. 148, of 7th June 2022).

Outside of this limited range of management measures, it is assumed that Grey Reef Shark *C. amblyrhynchus*, River sharks *Glyphis* spp., Dusky Shark *C. obscurus*, Smalltail Shark *C. porosus*, Sandbar Shark *C. plumbeus*, Borneo Shark *C. borneensis*, Pondicherry Shark *C. hemiodon*, Smoothtooth Blacktip Shark *C. leiodon*, Sharptooth Lemon Shark *Negaprion acutidens*, Caribbean Reef Shark *C. perezi*, Daggernose Shark *Isogomphodon oxyrinchus*, Night Shark *C. signatus*, Whitenose Shark *Nasolamia velox*, Blacknose Shark *C. acronotus*, Whitecheek Shark *C. dussumieri*, Lost Shark *C. obsoletus*, Pacific Smalltail Shark *C. cerdale*, Borneo Broadfin Shark *Lamiopsis tephrodes*, and Broadfin Shark *Lamiopsis temminckii* are largely unmanaged throughout their range.

A recent survey assessed management risk of overexploitation (the shortfall in national and international management) for 18 species of *Carcharhinus* spp. This survey found that the species with the highest mean management score (i.e., the best managed) were *C. acronotus*: 61.7%, *C. porosus*: 58.8%, and *C. tilstonii* 58.0%. The lowest scoring species (i.e., the least managed) were *C. leiodon*: 39.5%, *C. melanopterus* 40.1%, and *C. dussumieri*: 40.9%. On average management risk was high (49.2%) with only half of the necessary management in place for requiem sharks (Sherman et al., 2022).

For certain species of *Carcharhinidae* in particular, *P. glauca* stocks are currently assessed and managed by the Indian Ocean Tuna Commission (IOTC; Indian Ocean), International Convention for the Conservation of Atlantic Tunas (ICCAT; North and South Atlantic), and the Western Pacific Fishery Council (WCPFC; North Pacific). All *P. glauca* stocks are regularly assessed and the most recent assessments have determined stocks not to be overfished and not experiencing overfishing (ICCAT, 2015; WCPFC, 2017; IOTC, 2021). The first catch limits for any shark species managed by a RFMO were put into place for *P. glauca* by ICCAT in 2019 (ICCAT, 2019).

Potential benefit(s) of listing for trade regulation

A recent study noted that global shark catches are dominated by members of the family *Carcharhinidae*, with *Prionace glauca* alone making up 16% of global shark landings in 2017, with the study noting that *P. glauca* may now be dominating the meat trade via international commerce to meat markets in Japan, Spain, Taiwan POC, Uruguay, and Brazil (L. Feitosa, in litt., 2022). A listing would likely reduce fishing pressure driven by international trade for some species, and would help harmonise assessment, enforcement, and compliance, and could potentially strengthen existing national measures. For *P. glauca*, in theory a listing could complement existing management measures that are already in place. Excluding only *P. glauca* from an otherwise family-wide listing could create a loophole that might allow for the laundering of similar species.

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Inclusion of Hammerhead Sharks Family Sphyrnidae in Appendix II

Proponents: Brazil, Colombia, Ecuador, European Union, Panama

Summary: Three members of the family Sphyrnidae were listed in Appendix II at CoP16. The remaining six members of the family are now proposed for inclusion in Appendix II; *Sphyrna tiburo* in accordance with Article II, paragraph 2(a) of the Convention, and all remaining species (five are named by the proponents), as well as any other yet to be identified species of the Family Sphyrnidae, as lookalikes. The species included in this proposal occur in coastal warm temperate and tropical continental shelves between 42°N and 34°S to depths of 100 m. *Sphyrna tiburo* and *S. media* are found in both the western Atlantic and eastern Pacific Oceans. *Sphyrna tudes* occurs only in the western central and southwestern Atlantic and *S. corona* occurs only in the Eastern Central and Southeast Pacific. *Sphyrna gilberti* is known to occur in the northwestern Atlantic and possibly through the western central and southwestern Atlantic, although its distribution is uncertain. *Eusphyra blochii* is an Indo-West Pacific species that ranges from the Arabian Gulf through southern Asia to northern Australia and Papua New Guinea.

Sphyrna tiburo is one of the most productive shark species; maximum age, age-at-maturity, size-at-maturity, and offspring size increase with increasing latitude. Males mature at 2 and 4 years and females between 2–3 and 6–7 years, both reaching a maximum age of 5–6 and 6–18 years, with the older ages at higher latitudes. This species has a litter size of 4–16 pups following one of the shortest known gestation periods observed in sharks lasting 4.5–5 months. *Sphyrna media*, *S. tudes*, and *S. corona* are suspected to have similar life history parameters to *S. tiburo* given relatively similar geographic distribution and maximum size, although *S. tudes* has a 10-month gestation period (producing 5–12 pups per litter) and *S. corona* produces only two pups per litter. *Eusphyra blochii* has much lower productivity. The maximum age is estimated to be 21 years with males maturing at 5.5 years and females maturing at 7.2 years. The gestation period is 7–11 months with a litter size of 6–25 pups. When considering the footnote on commercially exploited marine species in Annex 5 of Res. Conf. 9.24 (Rev. CoP17), all Sphyrnidae can be regarded as having low productivity.

All members of the family Sphyrnidae are subject to targeted and incidental catch by commercial, artisanal, and recreational fishers. Due to the coastal nature of *Sphyrna tiburo* and *Eusphyra blochii*, *S. tudes*, *S. corona*, *S. media*, and (probably) *S. gilberti*, these species are often caught in gillnets, demersal trawls, and longlines with no deepwater refuge. They are used for their meat, which is typically consumed domestically, and their fins that are traded internationally. Fin-trimming surveys in Hong Kong SAR from 2014–2015 found that *S. tiburo* and *E. blochii* comprised 0.06% and 0.02% of samples respectively. Another study in Hong Kong SAR and Guangzhou found that *S. tiburo* and *S. tudes* comprised 0.04% and 0.03% of samples respectively; *E. blochii* comprised 0.01% in Hong Kong SAR and 0.06% of samples in Guangzhou. A recent study in Hong Kong SAR that surveyed retail markets for small, low-value fins in 2018 and 2019 reported that *S. tiburo* and *E. blochii* each comprised 0.4% of samples. None of these studies reported *S. corona*, *S. media*, or *S. gilberti* in samples, but fins from these species may also be traded internationally.

The most detailed information on population trends is available for *Sphyrna tiburo*. In the Northwest Atlantic (Atlantic Ocean, Gulf of Mexico, and US Caribbean Sea), the stock was last assessed in 2013 as not overfished and not experiencing overfishing having recovered due to management; the current status of the stock in the region is unknown. Since the previous stock assessment, two distinct stocks have been identified in US waters (Gulf of Mexico and Atlantic) and the next stock assessment will examine each separately. Further south in the western central and southwestern Atlantic *S. tiburo* is among the most caught sharks in Quintana Roo (Mexico), Panama, and Venezuela. It has experienced a substantial population reduction in Caribbean Colombia and Brazil (where it is now considered regionally extinct off Rio de Janeiro). In the eastern central and southeastern Pacific, this formerly abundant species has experienced significant reductions and is now considered locally extinct in the Gulf of California. It has not been observed since the 1980s in Central America, and is

rare in South America.

Less information is available for *Sphyrna media*, *S. tudes*, *S. corona*, and *Eusphyra blochii*, and no species-specific data are available for *S. gilberti*. *Sphyrna media* was historically abundant in Mexico, but it is now considered locally extinct in the Gulf of California and possibly in Pacific Central America (last recorded in the 1980s). In the remainder of its eastern central and southeastern Pacific range *S. media* is absent or exceptionally rare. In Atlantic South America, *S. media* was formerly common or even abundant in the 1970s but is now rarely encountered. Little species-specific information is available for *S. tudes*, but intensive fisheries throughout its range suggest population reductions. Similarly, intensive, and largely unmanaged fisheries exist throughout the range of *S. corona*, although the species remains relatively common in Pacific Colombia. There is no species-specific information available for *E. blochii*, except for Australian gillnet fisheries where the species is rarely caught and believed to be relatively stable. In South-East Asia and elsewhere in the Indo-West Pacific, *E. blochii* is inferred to have experienced population reductions due to intensive and largely unmanaged fisheries, and evidence of significant reductions in other shark and ray populations throughout the region.

Sphyrna tiburo and *Eusphyra blochii* are listed globally as Endangered on the IUCN Red List. *Sphyrna media*, *S. tudes*, and *S. corona* are all globally listed as Critically Endangered, having undergone population reductions assessed as over 80% over three generations, inferred on the basis of overfishing and habitat degradation. *S. gilberti* has been assessed as Data Deficient.

In the US, *Sphyrna tiburo* is managed under the Consolidated Atlantic Highly Migratory Species Federal Management Plan, which was initially developed in 2006 and amended in 2021. Management measures in the plan include seasonal closures and quotas. In addition to species-specific management in the United States, state gillnet bans probably also provide protection for this species. Atlantic Mexico has month-long shark fishery closed seasons applicable to *S. tiburo*. Several South American and Caribbean countries have general shark finning prohibitions, closed seasons for shark fisheries or are shark sanctuaries. There are no species-specific management measures in place for *S. gilberti* and *Eusphyra blochii* although both species may indirectly benefit from other more general management measures for fisheries in their respective ranges.

Within the family Sphyrnidae identification of the principal products in trade (dried fins) is challenging given the similarities of small-to-moderate sized dorsal and pectoral fins.

Analysis: Overexploitation has led to significant population reductions for all species within the family Sphyrnidae. Declines observed for *Sphyrna tiburo* overall are not consistent with the indicative guidelines for inclusion in Appendix II of commercially exploited medium-to-low productivity aquatic species suggested in the footnote to Annex 5 of Res. Conf. 9.24 (Rev. CoP17). However, the widespread disappearance of this species from significant parts of its range in the Pacific and South Atlantic portions of its range, combined with evidence that fins are traded internationally, imply that regulation of trade is needed to ensure that the harvest of specimens is not reducing the wild population of *S. tiburo* to a level at which its survival might be threatened by continued harvesting or other influences. This indicates that the species meets the criteria for inclusion in Appendix II in Annex 2aB of the Resolution. The recovery of the species in the Northwest Atlantic portion of its range indicates that *S. tiburo* is very responsive to management, which has had a significant effect on the health of its populations.

The Critically Endangered *Sphyrna corona*, *S. media*, and *S. tudes* likely meet or are near to meeting the criteria for inclusion in Appendix II in Annex 2aA of Res. Conf. 9.24 (Rev. CoP17), taking into account the footnote to Annex 5 for medium-to-low productivity (*S. media* and *S. tudes*) and low productivity (*S. corona*) commercially exploited aquatic species. While severe reductions in landings have been reported in some parts of the range of *Eusphyra blochii*, in other parts population reductions are assumed to be minimal. It is unclear therefore whether this species does or does not meet the criteria for inclusion in Appendix II in Annex 2a.

Many species in the family Sphyrnidae are harvested for international trade of their fins, including *Sphyrna tiburo*, *Eusphyra blochii*, and *S. tudes* and are present in international shark fin trade hubs.

Although data are lacking it is suspected that fins of *S. corona*, *S. media*, and *S. gilberti* are also traded internationally.

Small-to-moderate sized dorsal and pectoral fins of all members of the family Sphyrnidae resemble each other. *Sphyrna tiburo*, *S. tudes*, *S. corona*, *S. media*, and *S. gilberti* meet the criteria for inclusion in Appendix II in Annex 2 bA of Res. Conf. 9.24 (Rev. CoP17) based on the difficulty of distinguishing their fins from those of juvenile Appendix II listed *S. lewini* and *S. zygaena*. There is difficulty distinguishing the fins of *Eusphyra blochii* from those of juvenile Appendix II listed *S. mokarran* and *S. zygaena*, indicating that this species too meets the criteria for inclusion in Appendix II.

It appears therefore that all species in the Family Sphyrnidae meet the criteria for inclusion in Appendix II either under Annex 2a or 2b of the Resolution.

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 utilises CITES standard reference for fishes: Taxonomic Checklist of Fish species listed in the CITES Appendices and the Annexes of EC Regulation 338/97 (Elasmobranchii, Actinopteri, Coelacanthi, and Dipneusti, except the genus Hippocampus). Information extracted from Eschmeyer, W.N. & Fricke, R. (Eds.): Catalog of Fishes, an online reference, version update from 3 February 2015.

Sphyrna tiburo (Linnaeus, 1758)
Sphyrna media (Springer, 1940)
Sphyrna tudes (Valenciennes, 1822)
Sphyrna corona (Springer, 1940)
Eusphyra blochii (Cuvier, 1816)
Sphyrna gilberti (Quattro, Driggers, Grady, Ulrich and Roberts, 2013)

Range and IUCN Global Category

Sphyrna tiburo Bonnethead shark
 Endangered A2bcd (assessed 2019, ver 3.1). (Pollom et al., 2021)

This species occurs in the both the western Atlantic and eastern Pacific Oceans, inhabiting continental and insular shelves over mud and sand, seagrass, coral reefs, estuaries, shallow bays, and channels from shore to 90 m depth (Ebert et al., 2013, Weigmann, 2016). Sphyrna tiburo is found in the following Food and Agriculture Organization (FAO) Fishing Areas: 21, 31, 41, 77, 87. Range countries and territories are: Aruba; Bahamas; Belize; Bermuda; Bonaire, Sint Eustatius and Saba; Brazil; Colombia; Costa Rica; Cuba; Curaçao; Ecuador; El Salvador; French Guiana; Guatemala; Guyana; Honduras; Mexico; Nicaragua; Panama; Peru; Suriname; Trinidad and Tobago; United States; Uruguay; Bolivarian Republic of Venezuela (henceforth Venezuela).

Sphyrna media Scoophead shark

Critically Endangered A2bcd (assessed 2010, ver 3.1) (Pollom et al., 2020a)

Occurs in the Eastern Central and Southeast Pacific from the Gulf of California, Mexico to northern Peru and in the Western Central and Southwest Atlantic from Panama to southern Brazil. It inhabits waters over continental shelves from inshore to 100 m depth. It is captured in commercial and artisanal longlines and gillnets, which are typically unmanaged and operate throughout its range. Sphyrna media is found in the following FAO Fishing Areas: 31, 41, 77, 87. Range countries and territories are Aruba; Bonaire, Sint Eustatius and Saba; Brazil; Colombia; Costa Rica; Curaçao; Ecuador; El Salvador; French Guiana; Guatemala; Guyana; Honduras; Mexico; Nicaragua; Panama; Peru; Suriname; Trinidad and Tobago; Venezuela.

Sphyrna tudes Smalleye hammerhead

Critically Endangered A2bd (assessed 2020, ver 3.1) (Pollom et al., 2020b)

This species occurs in the Western Central and Southwest Atlantic from Colombia to the Rio de La Plata, Argentina. It inhabits inshore waters over the continental shelf at depths of 5–80 m. It is captured in intense and largely unmanaged commercial and artisanal beach seines, gillnets, longlines, and trawls throughout its geographic range. Sphyrna tudes is found in the following FAO Fishing Areas: 31, 41, 77, 87. Range countries and

territories are: Argentina; Aruba; Bonaire, Sint Eustatius and Saba; Brazil; Colombia; Curaçao; French Guiana; Grenada; Guyana; Suriname; Trinidad and Tobago; Uruguay; Venezuela.

Sphyrna corona Scalloped bonnethead

Critically Endangered A2bcd (assessed 2019, ver 3.1) (Pollom et al., 2020c)

This species inhabits the continental shelf from inshore to 100 m depth (Weigmann, 2016). It occurs in the Eastern Central and Southeast Pacific from the Gulf of California, Mexico to Peru (Ebert et al., 2013). It now appears to be absent from Mexico (Balart et al., 1996, Pérez-Jiménez, 2014). *Sphyrna corona* is found in the following FAO Fishing Areas: 77, 87. Range countries and territories are: Colombia; Costa Rica; Ecuador; El Salvador; Guatemala; Honduras; Nicaragua; Panama; Peru; Possibly Extinct in Mexico.

Eusphyra blochii Winghead shark

Endangered A2d+3d (2019, ver 3.1) (Smart and Simpfendorfer, 2016)

This species occurs on and near continental shelf waters of the Indo-West Pacific from the Arabian/Persian Gulf through southern Asia to northern Australia and Papua New Guinea (Last and Stevens, 2009). It occurs mainly in coastal nearshore waters, including muddy river deltas and estuaries (Ebert et al., 2013). In eastern Australia, this species is mainly encountered in concentrated areas of less than 50 km² (Smart and Simpfendorfer, 2016). *Eusphyra blochii* is found in the following FAO Fishing Areas: 51, 57, 61, 71. Range countries and territories are: Australia; Bangladesh; Brunei Darussalam; Cambodia; China; India; Indonesia; Islamic Republic of Iran (henceforth Iran); Kuwait; Malaysia; Myanmar; Oman; Pakistan; Papua New Guinea; Philippines; Saudi Arabia; Sri Lanka; Taiwan POC; Thailand; United Arab Emirates; Viet Nam.

Sphyrna gilberti Carolina hammerhead

Global – Data Deficient (assessed 2020, ver 3.1) (VanderWright et al., 2020)

This species is found in coastal waters of the Northwest Atlantic Ocean where it is known from South Carolina in the USA (Quattro et al., 2013). There is evidence of a cryptic lineage of hammerhead shark that may also be this species suggesting it also possibly occurs in Panama in the Western Central Atlantic and in Brazil in the Southwest Atlantic (Quattro et al., 2006, Pinhal et al., 2012, Quattro et al., 2013). *Sphyrna gilberti* is found in the following FAO Fishing Areas: 31, and likely in 21 and 41. Range countries and territories are: the United States (South Carolina, Florida, Georgia, North Carolina), and its presence is uncertain in Brazil (Santa Catarina, São Paulo) and Panama.

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

Biology

Species productivity was estimated using generation length as reported by the IUCN Red List assessments for each species. Productivity estimates are important for determining if species meet Annex 2aA of Res. Conf. 9.24 (Rev. CoP17), taking into account the footnote to Annex 5. A simple but robust estimator of natural mortality (M) for sharks and rays is the reciprocal of average lifespan ($M=1/w$) where average lifespan “ w ” is the generation length of the species (Pardo and Dulvy, 2016; Pardo et al., 2016). Hence, these CITES mortality thresholds can be back converted to calculate the corresponding generation length productivity thresholds ($GL = M^{-1}$) and these are as follows: high productivity (natural mortality > 0.5 or a generation length of < 2 years), medium productivity (natural mortality $0.2-0.5$ or generation lengths $2-5$ years), and low productivity (natural mortality < 0.2 or generation length > 5 years). Hence, any shark or ray with a generation length of 5 years or more will be classified as having “low productivity” according to this translation of generation length into mortality thresholds in the CITES criteria.

Sphyrna tiburo

Sphyrna tiburo typically give birth annually, reproducing by placental viviparity. Brood size is significantly correlated with maternal size and ranges from 4–16 pups with one of the shortest known gestation periods in sharks, lasting ~4.5–5 months. In the Gulf of Mexico, males mature at 2 years and reach a maximum age of 5–6 years and females generally mature between 2–3 years and reach a maximum age of 5–6 years (Lombardi-Carlson et al., 2003). However, maximum age, size-at-maturity, and offspring size increases with increasing latitude, with females reaching a maximum age of 17.9 years in the Southeast Atlantic (Frazier et al., 2014). Overall, the generation length of *S. tiburo* is estimated to be 12 years.

Demographic studies however indicate that *S. tiburo* exhibits a very high rate of population growth (mean = 1.304 per yr; 95% confidence interval = 1.150–1.165 per year) relative to other sharks, making them one of the most productive species of elasmobranch. If managed correctly, their relatively high intrinsic rate of increase should allow *S. tiburo* to withstand higher fishing mortality compared with other shark species.

Sphyrna media

This species reaches a maximum size of 150 cm total length (TL); females mature at 100–133 cm TL and males at 90–100 cm TL; reproduction is placental viviparous and females give birth to young that are 34 cm TL (Ebert et al., 2013). Generation length is estimated based on *S. tiburo* (12 years). Like *S. tiburo*, this species probably has medium to low productivity, but very little is known about its life history (Brennan, 2020).

Sphyrna tudes

Reaches a maximum size of 150 cm TL; females reach maturity at 98 cm TL and males at 80 cm TL. Reproduction is placental viviparous, and females give birth after 10 months of gestation to 5–12 pups per litter (or up to 19; Lessa et al., 2018; Stride et al., 1992) that are 30 cm TL at birth (Ebert et al., 2013). Generation length is estimated based on *S. tiburo* (12.3 years). Like *S. tiburo*, this species probably has medium-to-low productivity.

Sphyrna corona

Reaches a maximum size of 92 cm TL; female maturity is unknown, but males mature at 67 cm TL; reproduction is placental viviparous, and it is suspected that females give birth to two pups per litter that are 23 cm TL (Ebert et al., 2013). Generation length is suspected to be about 8 years, based on available data for the larger *S. tiburo*. However, unlike *S. tiburo*, it is widely accepted that *S. corona* has a low fecundity (Brennan, 2020).

Eusphyra blochii

This is a slow growing species that reaches maturity at 7.2 years for females and 5.5 years for males (Stevens and Lyle, 1989, Smart et al., 2013). It reaches a maximum size of 186 cm TL, a maximum age of 21 years and has a generation length of 14 years.

Sphyrna gilberti

Reproduction is presumably placental viviparous based on neonates with an umbilicus (Quattro et al., 2013). Size at birth is thought to be near 39 cm TL based on sampled neonates with an open umbilicus (Quattro et al., 2013). Nothing else is known of its biology.

Population size

Global population size of any of the species is unknown.

Population trends

Sphyrna tiburo

Global

The global population trend for *S. tiburo* was estimated from a compilation of three estimates of the species' population trends across three populations: the Southwest Atlantic, Eastern and Central Pacific, and the Northwest Atlantic. Overall, the combination of low fishing mortality throughout the USA, decreases in Atlantic Central America combined with widespread disappearance in the Southwest Atlantic and Pacific means this species is inferred to have undergone a population reduction of 50–79% over three generation lengths (36 years). This is due to actual and potential levels of fishing and habitat loss and degradation from coastal development. It is currently assessed as Endangered A2bcd and was previously assessed as Endangered in 2020, and Least Concern prior to that in 2005 and 2016.

When the declines were scaled to the percent of distribution covered (Table 1), the estimated decline ranged from 52–68% from baseline over three generation lengths (36 years). If these declines were projected over the next ten years, declines range to 69–76% from baseline.

Table 1. *Sphyrna tiburo* population trends.

| Region | Estimated percent of distribution covered (%) | Population trend and extent of change | Time period | Source |
|---|---|---------------------------------------|---------------------------------|----------------------------------|
| The Northwest and Western Central Atlantic (Northwest Atlantic portion) | 20–30 | Increasing (40%) | 3 generation lengths (36 years) | Pollom et al., 2021; SEDAR, 2013 |
| The Northwest and Western Central Atlantic (Western Central Atlantic portion), Caribbean Sea and Southwest Atlantic (Caribbean Sea portion) | 20–30 | Decreasing (20–25%) | 3 generation lengths (36 years) | Pollom et al., 2021 |
| Caribbean Sea and Southwest Atlantic (Atlantic South American portion) | 20 | Decreasing (>80%) | 3 generation lengths (36 years) | Pollom et al., 2021 |
| Eastern Central and Southeast Pacific | 30 | Decreasing (>80%) | 3 generation lengths (36 years) | Pollom et al., 2021 |

Northwest and Western Central Atlantic

The Northwest and Western Central Atlantic spatial unit covers the USA (including the US Gulf of Mexico, and US Caribbean Sea), the Bahamas, and Mexico. According to the most recent US stock assessment, the species was considered not overfished and not experiencing overfishing (SEDAR, 2013). However, reviewers disagreed with the decision to conduct the assessment with combined Gulf of Mexico and United States South Atlantic stocks, and the stock status is currently considered unknown in the region. Since the previous stock assessment in 2013, two distinct stocks have been identified in US waters (Gulf of Mexico and Atlantic) and the next stock assessment will examine each separately (NOAA, 2021). In Mexico, *S. tiburo* are harvested via a directed fishery in Quintana Roo, where the species is the third most important catch. In the western Yucatan Peninsula, a mean catch-per-unit effort of 1.94 individuals per fishing day (± 2.36 standard deviations) has been estimated in areas with high environmental suitability indices for this species (Chi Chan et al., 2021).

The weighted overall population trend for the Northwest Atlantic estimated a 40% increase over three generation lengths (36 years). However, outside the US (where quota management is used), Mexico (where a closed season is implemented), and the Bahamas (where there is no shark harvest), the species is captured in artisanal fisheries and as bycatch in shrimp trawl and lobster fisheries in areas where management measures may not exist. Expert elicitation concluded that, given the uncertainty regarding the levels of harvest and their sustainability, the species should be categorised as Near Threatened. Mexico and the Bahamas were considered part of the Western Central Atlantic and Caribbean portion in Table 1, which was considered Near Threatened, based on the IUCN Red List.

Southwest Atlantic and Caribbean

The Caribbean Sea and Southwest Atlantic spatial unit covers Belize south to Brazil, including Panama and Trinidad and Tobago (Castro, 2011, Ebert et al., 2013). In Panama, *S. tiburo* is the third most abundant species in the nearshore gillnet fishery that is largely unregulated. In Venezuela, *S. tiburo* remains the fourth most caught shark, but is subjected to heavy and unmanaged exploitation and populations are suspected to have decreased as a result. This situation is expected to be reflected across The Guianas. This species is present in landings in Cuba, Belize (mostly adults), and Trinidad. In Colombia, *S. tiburo* was commonly encountered in the 1980s off the Caribbean coast but is now very rare.

In Brazil, *S. tiburo* was regionally assessed as Extinct in the state of Rio de Janeiro, collapsed in São Paulo, and Vulnerable in Espírito Santo State due to overfishing. There have been very sparse records of this species in Bahia state although several individuals were recorded there in a 2012–2013 study. The only other record since the 1980s is one from 1995, and fishers describe this species as being depleted. The Atlantic South American portion of this species' population has undergone a population reduction of >80% over the past three generation lengths (36 years) (Pollom et al., 2021).

The status of the species is less clear throughout Central America and the Caribbean Sea where *S. tiburo* is among the most caught shark species in Panama and Venezuela, but rare in Colombia. The Caribbean Sea was considered part of the Western Central Atlantic and Caribbean portion in Table 1, which was considered Near Threatened.

Eastern Central and Southeast Pacific

The spatial unit in the Eastern Pacific covers Southern California, USA to Ecuador (Compagno, 1984; Castro, 2011). *Sphyrna tiburo* was formerly abundant in the Gulf of California and the Pacific coast of Mexico. The species was initially caught as bycatch in coastal fisheries, with records becoming increasingly rare from the 1980s onwards. In the 1990s, after large-bodied sharks in the area were mostly fished out, artisanal fisheries shifted to targeting smaller coastal sharks, including *S. tiburo*. The species is no longer present in the Gulf of California, and despite extensive landings surveys and fishery-independent research, the last *S. tiburo* record in Mexico was in 2006 in Oaxaca. The species also appears to be extinct along Mexico's Pacific coastline.

Sphyrna tiburo had not been encountered in Central America since the 1980s. There were records of the species in the industrial trawl fishery in Colombia in the 1990s, but surveys of this fishery in 2007 did not record any *Sphyrna* species. A recent study in Pacific Panama found that this species made up 3% of the landings in an artisanal shark fishery between 2007–2009 (Guzman et al., 2020). *Sphyrna tiburo* is also now rare in gillnet and beach seine catches in Ecuador today despite a historical presence.

Sphyrna tiburo was classified as Critically Endangered in the Eastern Central and Southeast Pacific. The species is estimated to have undergone a population reduction of >80% over the past three generation lengths (36 years) due to heavy and largely unmanaged fishing pressure across the species' range, as well as the recent absence of records of *S. tiburo* from large parts of Mexico and Central America, and a dearth of contemporary records.

Sphyrna media

The declining numbers of records over recent decades and range contraction in some areas indicate that this shark has undergone population reductions in both the Pacific and the Atlantic. In the Eastern Central Pacific, this

shark was formerly abundant in the Gulf of California and off the Pacific coast of Mexico (Hernández-Carvalho, 1967). Records became increasingly rare from the 1980s onwards, and it is no longer present in the Gulf of California; there were only three records from Mexico in the two decades leading up to 2014, all of which were restricted to southern Mexico (Pérez-Jiménez, 2014, Saldaña-Ruiz et al., 2017). Despite extensive landings surveys, fishery-independent research surveys, and research on museum specimens, the last record in Mexico was in 2006 in Oaxaca (Pérez-Jiménez, 2014, Pollom et al., 2020a). This shark has not been encountered in Pacific Central America since the 1980s (apart from a single individual caught by artisanal shark fishers in Panama sometime between 2007 and 2009; Guzman et al., 2020). There are more recent records in Colombia and Ecuador, but they are rare (Pérez-Jiménez, 2014). There were records in the industrial trawl fishery in Colombia in the 1990s, but surveys of this fishery in 2007 did not record any *Sphyrna* species (Navia and Mejía-Falla, 2016). The Atlantic South American portion of the population has also been reduced in size substantially. Off Caribbean Colombia, this species is considered rare. In Venezuela, it is subjected to intense and unmanaged fishing pressure, and it is suspected to have undergone population reduction there as a result of levels of exploitation. This situation is expected to be similar across the Guianas. There have been very sparse records in Bahia state, Brazil; records of sphyrnid sharks there have been declining in number since the 1990s and since 2000 have only occurred very rarely. There are no recent records from southern Brazil. Overall, this shark was formerly common or even abundant in the 1970s, but has been and still is subjected to intense and largely unmanaged fishing pressure, as well as the degradation of mangrove habitats, and has undergone range retractions. There have been very few recent records and it is inferred that *S. media* has undergone a population reduction of >80% based on levels of exploitation, and it is assessed as Critically Endangered A2bcd.

This historically abundant and medium-to-low productivity species is considered locally extinct in the Gulf of California and possibly the remainder of Mexico and Pacific Central America. It is considered rare in the northern section of Pacific South America. Similarly, this species is considered rare in Atlantic Central and South America. While no data on population reduction are available, it seems likely that this species has declined to 10–20% from a historical baseline, given its widespread rarity and probable local extinction in some areas.

Sphyrna tudes

There are few data on population reduction, but intensive unmanaged fisheries are suspected to have caused reductions and possibly local extinctions throughout this species' range. For example, in Brazil, this hammerhead has not been recorded in 35 years from Ceará state and it is considered by local fishers to be depleted in Bahia state. Although strictly protected in Brazil, it is clear that it is still landed and traded in various states. *There are intense and unmanaged artisanal fisheries in southern Brazil, which are suspected to have reduced the population substantially. In Trinidad and Tobago, this species had already undergone a notable reduction in landings in the inshore artisanal fishery prior to 2006 (Shing, 2006).* Overall, due to intense and largely unmanaged fisheries across its range, lack of refuge at depth, suspected population reductions in many areas and local extinctions suspected from an absence of records (despite continued sampling and observation), and its relatively unproductive life history, it is suspected that *S. tudes* has undergone a population reduction of >80% over the past three generations (36 years), and it is assessed as Critically Endangered A2bd.

This medium-to-low productivity species appears to be locally extinct in Ceará, Brazil (northern Brazil), is depleted in Bahia (south-central Brazil), and was reported to have decreased in landings in Trinidad and Tobago by 2006. While there are very few data available, it seems possible that this species has declined to 10–20% from a historical baseline.

Sphyrna corona

Records of the *S. corona* have become increasingly rare since the 1950s in the northern part of its range. *This species is thought to be locally extinct in the Gulf of California and is likely overfished further south (Pérez-Jiménez, 2014; Saldaña-Ruiz et al., 2017).* There were nine records from Mexico between 1978 and 1994 and there have been none since. The situation is somewhat different in Colombia, where the species persists and is caught relatively frequently in artisanal catches (Orozco-Guarín, 2015; Galindo-Arana, 2016). Similarly, the species was relatively common in artisanal shark fisheries in Pacific Panama, representing 7% of the artisanal catch (Guzman et al., 2020), and another study reported it to be the third most caught species in artisanal shark fisheries in Pacific Panama (Rodríguez Arriatti et al., 2021). Fishing pressure is high in Colombia and it is suspected to have undergone a population reduction, although not as severe as that seen in Mexico. Overall, due to its slow life history, the known sensitivity of hammerhead sharks to overfishing, degradation of mangrove habitats, the level of intense and unmanaged fisheries across its range, its lack of refuge at depth, and the lack of recent records in Mexico (a large proportion of its range) despite fisheries and independent surveys, it is inferred that this hammerhead has undergone a population reduction of >80% over the past three generations (24 years) and it is assessed as Critically Endangered A2bcd.

While this low-productivity species appears to be locally extinct in the Gulf of California and the rest of Mexico, it is still frequently caught in artisanal catches in Colombia and Panama. It is likely that this species has declined to 15–20% from a historical baseline.

Eusphyra blochii

There are no species-specific data available on population numbers and how they have changed over time for any part of this species' range. Throughout the majority of its range, in particular Asia, fishing effort is concentrated in coastal regions, is intense and is generally unregulated; *E. blochii* is inferred to have been heavily exploited. This species is now rarely encountered in either India or Indonesia where it has previously been reported, and the absence of the species from fish markets and landing surveys in these countries is likely accurately to reflect the situation more broadly across the majority of its range. Only one individual was seen in market surveys in Indonesia during which approximately 20,000 sharks were recorded. It is therefore suspected to be severely overfished in this country as most of Indonesia's fishing effort is focused on coastal nearshore areas (Smart and Simpfendorfer, 2016). Recent catch data from India identifies sharks to species level and has no mention of *E. blochii* as a bycatch or byproduct species (e.g., Varghese et al., 2013). Within Australia, *E. blochii* is lightly exploited in several net fisheries. It makes up a very small proportion of catches in tropical gillnet fisheries (Harry et al., 2011) and its population is not believed to have been reduced substantially. While there are no species-scientific data on its status, the population is inferred to have decreased by at least 50% within the equivalent of three generations (42 years) and hence it is assessed as Endangered globally based on heavy exploitation levels.

The low productivity of this species, combined with the severe reductions in landings reported in India and Indonesia, suggest that this species has likely experienced substantial population reductions. However, this species is likely not to have been reduced substantially in Australian waters.

Sphyrna gilberti

There is currently inadequate information available to assess the species beyond Data Deficient (VanderWright et al., 2020).

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

Fins

Hammerheads are among the main shark species in the fin trade and are among the preferred species for shark fin soup. Like other hammerheads, *S. tiburo* is utilised for its meat and fins. Meat is typically consumed domestically, and fins are traded internationally. Fin-trimming surveys in Hong Kong SAR from 2014–2015 found that *S. tiburo* and *E. blochii* comprised 0.06% and 0.02% of samples respectively (Fields et al., 2018). Another study in Hong Kong SAR and Guangzhou detected *S. tiburo* (0.04% of samples), *S. tudes* (0.03% of samples), and *E. blochii* (0.01% and 0.06% of samples in two locations) in samples collected from 2015–2017 (Cardeñosa et al., 2020a). A more recent survey in Hong Kong SAR specifically of small, low-value fins in the shark fin retail markets, reported that *S. tiburo* and *E. blochii* each comprised 0.4% of samples (Cardeñosa et al., 2020b). None of these studies reported *S. corona*, *S. media*, or *S. gilberti* in samples, but fins from these species may be traded internationally. This seems possible for *S. corona*, as fins were removed from carcasses upon landing (Guzman et al., 2020).

Meat and Other Products

The meat, liver oil, skin, cartilage, and jaws may also be used. In Colombia, meat from *S. tudes* is consumed by Indigenous communities.

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

All hammerhead species have a characteristic fin shape that distinguishes them from all other shark species; however, visually distinguishing species of hammerhead based on their fins is much more challenging as *small-to-moderate sized dorsal and pectoral fins of all members of the family Sphyrnidae resemble each other* (Table 2). In their analysis of the proposal to include *S. lewini*, *S. mokarran* and *S. zygaena* in CITES Appendix II (CoP16 Prop. 43), FAO (2013) noted that it was not clear why other species in the family Sphyrnidae were not proposed to be listed as lookalikes, noting that some panel members considered that visual identification of dried fins by non-experts (such as enforcement officers) would be challenging. A similar conclusion was reached by an evaluation of implementation issues relevant to the three CITES-listed hammerhead species in 2018 (AC30 Inf. 14). At the time of the proposal to include *S. lewini*, *S. mokarran*, and *S. zygaena* in CITES Appendix II at CoP16, it was believed that only these three Sphyrnidae species were in international trade, due to their superior size. However, since CoP16, improved research of the species composition of fins sold at international trade hubs has demonstrated that smaller-bodied hammerhead species are also being internationally traded. This exposes the three large-bodied hammerhead species that are already included in the Appendices (*S. lewini*, *S. mokarran*, and *S. zygaena*) to illegal trade, because their fins may be hidden in shipments of lookalike species from the wider family. Given the latest conservation status of *S. lewini* and *S. mokarran* (both species are now assessed as Critically Endangered globally), there is an urgent need to ensure that loopholes do not exist that allow for widespread illegal trade in the listed hammerhead species.

Table 2. Species in the family Sphyrnidae that meet the criteria for listing in Annex 2b A of Resolution Conf. 9.24 (Rev CoP15) based on the difficulty of distinguishing their fins from one of the species already listed in CITES Appendix II.

| Species | Annex 2b Appendix II-listed Species | Annex 2b New proposal Annex 2aA or 2aB |
|--------------------|--|---|
| <i>S. tiburo</i> | Small-to-moderate sized dorsal fins, pectoral fins: <i>S. lewini</i> and <i>S. zygaena</i> ² | Dorsal fin: <i>S. gilberti</i> , <i>S. media</i> ³ Pectoral fin: <i>S. corona</i> , <i>S. media</i> ³ Caudal fin: <i>E. blochii</i> , <i>S. corona</i> , <i>S. gilberti</i> , <i>S. media</i> ³ |
| <i>S. gilberti</i> | Pectoral fins: <i>S. lewini</i> (Scalloped Hammerhead) ² ; <i>S. zygaena</i> ⁴ | Dorsal fin: <i>S. media</i> , <i>S. tiburo</i> ³ Caudal fin: <i>E. blochii</i> , <i>S. corona</i> , <i>S. media</i> , <i>S. tiburo</i> ³ |
| <i>S. media</i> | Small-to-moderate sized dorsal fins, pectoral fins: <i>S. lewini</i> and <i>S. zygaena</i> ² | Dorsal fin: <i>S. gilberti</i> , <i>S. tiburo</i> ³ Pectoral fin: <i>S. corona</i> , <i>S. tiburo</i> ³ Caudal fin: <i>E. blochii</i> , <i>S. corona</i> , <i>S. gilberti</i> , <i>S. tiburo</i> ³ |
| <i>S. corona</i> | Small-to-moderate sized dorsal fins: <i>S. lewini</i> and <i>S. zygaena</i> ^{2,3} | Dorsal fin: <i>S. tudes</i> ³ Pectoral fin: <i>S. media</i> , <i>S. tiburo</i> ³ Caudal fin: <i>E. blochii</i> , <i>S. gilberti</i> , <i>S. media</i> , <i>S. tiburo</i> ³ |
| <i>S. tudes</i> | Small-to-moderate sized dorsal fins: <i>S. lewini</i> and <i>S. zygaena</i> ^{2,3} ; Caudal fin: <i>S. lewini</i> ³ | Dorsal fin: <i>S. corona</i> ³ |
| <i>E. blochii</i> | Dorsal fin, Pectoral fin: <i>Sphyrna mokarran</i> (Great Hammerhead) ^{1,2,3} ; Caudal fin: <i>Sphyrna mokarran</i> , <i>S. zygaena</i> ² | Caudal fin: <i>S. corona</i> , <i>S. gilberti</i> , <i>S. media</i> , <i>S. tiburo</i> ³ |

Sources: ⁽¹⁾ FAO, 2016; ⁽²⁾ Abercrombie and Jabado, 2022; ⁽³⁾ R.W. Jabado, *in litt.*, 2022. ⁽⁴⁾ VanderWright, et al., 2020.

Additional information

Threats

All members of the family Sphyrnidae are subject to targeted and incidental catch by commercial, artisanal, and recreational fishers. Due to the coastal nature of *S. tiburo* and *E. blochii*, *S. tudes*, *S. corona*, *S. media*, and (likely) *S. gilberti* are often caught in gillnets, demersal trawls, shrimp trawls, and longlines with no deepwater refuge. As these fisheries are coastal, all of which typically fall outside of the mandate of Regional Fishery Management Organization (RFMO) management. Population reductions have been exacerbated by habitat degradation and loss, including conversion of mangrove forest to shrimp aquaculture.

Conservation, management and legislation

National

In the US, *S. tiburo* is managed under the Consolidated Atlantic Highly Migratory Species Federal Management Plan, which was initially developed in 2006 and amended in 2021. Management measures in the plan include seasonal closures and quotas. In addition to species-specific management in the US, state gillnet bans likely also provide protection for this species.

Atlantic Mexico has a shark fishery closed season in May and June in the states of Tamaulipas, Veracruz, and Quintana Roo, and 15 May–15 June, and 1–29 August in the states of Tabasco, Campeche and Yucatán, which would include *S. tiburo*. Several South American and Caribbean countries have shark finning prohibitions (all Central American countries (from Belize to Panama, including the Dominican Republic), closed seasons for shark fisheries (Guatemalan Caribbean), or are shark sanctuaries (Honduras, Bahamas, and the Caribbean Netherlands). These measures would be applicable to *S. tiburo*, possibly *S. gilberti*, and the measures in Panama may also benefit the northern range of *S. media*.

Colombia prohibited all shark and ray fishing in 2021, which is applicable to *S. tiburo*, *S. media*, *S. tudes*, and *S. corona*. Pacific Mexico has an annual closed season for targeted shark and ray fishing from 1 May–31 July, applicable to *S. tiburo*, *S. media*, and *S. corona*. The Brazilian Ordinance of Ministry of the Environment N° 445 listed *S. tiburo*, *S. media* and *S. tudes* restricting harvest and trade, which was updated in 2022. There are no species-specific management measures in place for *S. gilberti* and *E. blochii*, although both species may indirectly benefit from other more general management measures for fisheries in their respective ranges.

International

There are currently no species-specific international protections afforded to *S. tiburo* and *E. blochii*, *S. tudes*, *S. corona*, *S. media*, or *S. gilberti*. These species are distributed in nearshore coastal habitats and captured by artisanal or small-scale fisheries that generally fall outside of the scope of instruments adopted by RFMOs.

Implementation challenges (including similar species)

Some species appear to be consumed or sold locally when captured with little evidence of international trade in fins (i.e., *S. corona*, *S. media*, and *S. gilberti*).

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Inclusion of the following species of Freshwater Stingray in Appendix II: *Potamotrygon wallacei*, *P. leopoldi*, *P. albimaculata*, *P. henlei*, *P. jabuti*, *P. marquesi*, and *P. signata*

Proponent: Brazil

Summary: *Potamotrygon* is a genus of South American freshwater stingrays with complex taxonomy, currently recognised as comprising around 30 species. The Brazilian populations of *Potamotrygon* spp. and eight Colombian species were listed in Appendix III in 2017. *Potamotrygon wallacei* and *P. leopoldi* are now proposed for inclusion in Appendix II in accordance with Article II, paragraph 2(a) of Res. Conf. 9.24 (Rev. CoP17), and *P. albimaculata*, *P. henlei*, *P. jabuti*, *P. marquesi*, and *P. signata* in accordance with Article II paragraph 2(b) as lookalikes.

The species proposed are neotropical freshwater stingrays that are endemic to the Amazon and Parnaíba River Basins and are predominantly found in water shallower than 3 m. *Potamotrygon leopoldi* is endemic to the Xingu River drainage in the lower Amazon Basin, in Mato Gross and Pará states, Brazil. *Potamotrygon wallacei* is endemic to the middle Rio Negro drainage in Amazonas State, Brazil. *Potamotrygon leopoldi* and *P. wallacei* are considered to have low productivity. Age at maturity is not known for all species, however the youngest age reported for females is two years for *P. wallacei*, while the oldest is six to seven years for *P. leopoldi* females, while *P. leopoldi* males mature at four to five years. Female *P. wallacei* produce two embryos while *P. leopoldi* produce an average of 4.84 pups. Gestation length ranges from three to four months. *Potamotrygon leopoldi* and *P. wallacei* have annual reproductive cycles, often corresponding with cyclic hydrologic conditions. Maximum age is estimated to be five years for *P. wallacei*, ten years for *P. albimaculata* and *P. jabuti*, and 14 years for *P. leopoldi*.

Freshwater stingrays, including other *Potamotrygon* species not covered by this proposal, have been in demand for the ornamental aquarium trade since the late 1970s. The black rays (typically species with white or pale markings on a black background here represented by *P. leopoldi*, *P. albimaculata*, and *P. henlei*, and sometimes *P. jabuti*, a variable species) are the most sought-after ornamental freshwater stingrays in all ornamental markets (Asia, Europe, and North America). *Potamotrygon leopoldi* is the most valued species exported and the most popular stingray in Asian countries, followed by *P. jabuti*. *Potamotrygon leopoldi* is also the most popular stingray in the United States of America (USA) and Canada. This species was intensively fished for the international ornamental aquarium trade from the 1990s to 2006. Adult *P. leopoldi* were first captured for breeding stock in Asia, the European Union, and North America in the early 2000s, with captures intensifying between 2005 and 2011. Hybridisation involving these and other *Potamotrygon* species (mostly *P. albimaculata* and *P. jabuti*) began in 2000 by Asian breeders mainly in Thailand. Some of these hybrids are considered more attractive and of higher value than wild-caught individuals, resulting in a reduction in demand for the latter. Trade data from the USA (2011–2020) show that most imports (94%) were captive-bred, mainly from Thailand (80%). CITES data since the Appendix III listing show similar patterns. There is also evidence of some illegal trade. *Potamotrygon leopoldi* is the most encountered species in Brazil's official record of stingray seizures. Over half of all freshwater stingray seizures between 2002 and 2018, comprised *P. leopoldi* (55%), although actual quantities are unknown. Additionally, 30% of stingray individuals exported from Amazonas State and reported as *P. motoro* (an Appendix III listed species) were *P. leopoldi*. No recent seizure records were reported for *P. wallacei*.

Threats include capture for ornamental trade, habitat loss and degradation caused by fire in flooded-forest habitat, anthropogenic development such as agricultural expansion, ranching, mining, hydroelectric powerplant development, as well as persecution and local consumption by humans. Fishing mortality combined with other anthropogenic activities has led to *P. leopoldi* population reductions. Overall, international demand for wild-caught individuals for the ornamental trade has declined due to captive breeding facilities supplying the aquarium market. *P. wallacei* is said to be

more difficult to breed in captivity and therefore the shift in supply for the aquarium market seen for *P. leopoldi* may not be the same for *P. wallacei*. However, demand for this species is apparently lower than for other species and quotas set are not being fully used according to export records.

IUCN Red List assessments for *P. wallacei* and *P. leopoldi* are currently being completed and are pending review. Both are provisionally assessed as Vulnerable with estimated population reductions of 33% and 30–49% over the past three generations, respectively. Population trend data for these species vary. One study of *P. leopoldi* in a harvested area detected a marked decrease in catch rates between 2004/5 and 2021; other surveys have detected no clear trends. Eleven out of 14 separate subpopulations of *P. wallacei* in the Rio Negro region were assessed as stable. *Potamotrygon albimaculata*, *P. jabuti*, *P. marquesi*, *P. signata*, and *P. henlei* are currently undergoing assessment.

P. signata is believed to have a restricted distribution; the other species are relatively widespread in different river systems in the Amazon basin.

Since 2017, all Brazilian populations of *Potamotrygon* spp. have been listed in CITES Appendix III. Within Brazil, six species of freshwater stingray (*Potamotrygon henlei*, *P. motoro*, *P. orbignyi*, *P. leopoldi*, *P. schroederi*, and *P. wallacei*, (as *P. hystrix*, taxonomy now superseded)) are regulated under an export quota, which also specifies a species-specific maximum disc width export size. The Brazilian export quota system was established in 2003. However, since 2019 the Brazilian Management Authority has not authorized exports of these freshwater stingray species. In the years leading up to this, numbers of legally exported wild specimens were well below the export quotas.

All species of the genus *Potamotrygon* exhibit polychromatism, making identification to species level based on colour patterns challenging. However, the species in this proposal can be divided into two general groupings: the black rays (*P. leopoldi*, *P. albimaculata*, *P. henlei* (and sometimes *P. jabuti*)); and the brown rays (*P. wallacei*, *P. signata*, and *P. marquesi*).

Analysis: The seven Brazilian *Potamotrygon* species proposed are freshwater rays endemic to river systems in the Amazon basin. Most are relatively widespread; one (*P. signata*) has a restricted distribution. In the past some species, particularly *P. leopoldi* but also *P. wallacei* under its earlier name of *P. hystrix*, have been exported in significant numbers for the aquarium fish trade. Population decreases have been detected in both these species and may be ongoing, although population trend data are often conflicting and it is unclear whether decreases can be ascribed to harvesting for export or other factors such as habitat degradation. Since 2000 *P. leopoldi* has been hybridised, principally in Thailand, with other species to produce forms that are sometimes more desirable than wild-caught forms and command premium prices. This has reduced demand for wild-caught individuals. From 2008, Brazilian authorities set export quotas for six native *Potamotrygon* spp., but declared exports remained far below these limits. Since 2021 the Brazilian Management Authority has not authorized exports of these freshwater stingray species so that there is currently no legal export trade. There are indications of some illegal export, including reports of seizures, but there are no indications of large scale illegal trade and species other than *P. leopoldi* and its hybrids rarely appear on the open market (*P. signata* has not been detected in trade recently). It seems unlikely that harvest for such trade has a significant impact on *Potamotrygon* populations.

On this basis, it does not appear that regulation of trade will contribute to threat reduction for these species, so they do not appear to meet the criteria for listing in Appendix II in Annex 2a of Res. Conf. 9.24 (Rev. CoP17).

Of the species proposed, the black rays (*P. albimaculata*, *P. henlei*, *P. leopoldi* and *P. jabuti*) may resemble each other but should be relatively easy to distinguish from the brown rays (*P. wallacei*, *P. signata*, and *P. marquesi*).

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

Proposed as meeting Annex 2a criteria:

Potamotrygon leopoldi.

Potamotrygon wallacei. *P. wallacei* was previously called *P. hystrix* (*P. hystrix* is endemic to the Paraná-Paraguay River basin).

FishBase (www.fishbase.org, accessed 2nd Sept 2022) currently recognises 30 species of *Potamotrygon*.

Taxonomy of the group is complex.

Proposed as lookalikes:

P. albimaculata, *P. henlei*, *P. jabuti*, *P. marquesi*, and *P. signata*

The current CITES Standard Reference for *Potamotrygon* spp. is Eschmeyer's Catalog of Fishes as of 12 May 2017 (AC29 Doc. 35).

IUCN Global Category

No recent IUCN assessments for these proposed species are currently published on the Red List. However, preliminary assessments for *P. leopoldi* and *P. wallacei* are being reviewed and if accepted may be published in the Red List update in December 2022. Assessments for the other species are in progress but have not been completed or sent out for review. Here information from the preliminary assessments is presented that may or may not be included in the final assessment depending on their progress through the IUCN Red List system.

Table 1. Species included in this proposal and the associated IUCN Preliminary Red List global category (all using IUCN Red List version 3.1). IUCN Red List assessments for all species are currently in draft form and have not yet been accepted or submitted for publication in the December 2022 Red List update.

| Species | IUCN Preliminary Global Category |
|----------------------------------|---|
| <i>Potamotrygon leopoldi</i> | Vulnerable A2bcd (Charvet, in litt., 2022) |
| <i>Potamotrygon wallacei</i> | Vulnerable A2cd (Araújo, Torres, Sayer, in litt., 2022) |
| <i>Potamotrygon albimaculata</i> | In progress |
| <i>Potamotrygon henlei</i> | In progress |
| <i>Potamotrygon jabuti</i> | In progress |
| <i>Potamotrygon marquesi</i> | In progress |
| <i>Potamotrygon signata</i> | In progress |

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

A simple but robust estimator of natural mortality (M) for sharks and rays is the reciprocal of average lifespan ($M=1/w$) where average lifespan " w " is the generation length of the species (Pardo and Dulvy, 2016; Pardo et al., 2016). Hence, these CITES mortality thresholds can be back converted to calculate the corresponding generation length productivity thresholds ($GL = M^{-1}$) and these are as follows: high productivity (natural mortality >0.5 or a generation length of <2 years), medium productivity (natural mortality $0.2-0.5$ or generation lengths $2-5$ years), and low productivity (natural mortality <0.2 or generation length >5 years).

Potamotrygon leopoldi Xingu Freshwater Stingray

Preliminary Assessment: Vulnerable A2bcd (ver 3.1. (Charvet, in litt., not yet accepted for publication in the December 2022 Red List update).

Range

A Neotropical freshwater stingray species endemic to the Xingu River drainage in the lower part of the Amazon Basin, including the Iriri, Curuá, and Fresco (Rosa 1985) rivers, in Pará State, and some of the main tributaries of the Xingu River, in Mato Grosso State, Brazil. Throughout its distribution, *P. leopoldi* prefers to inhabit water up to 3 m deep in areas with substrates with rocks, pebbles, and sand, which causes the species to occur in patches along the Xingu River.

Biology

The maximum observed age for males is seven years, and for females, 14 years, reaching a maximum size of 72 cm disc width (DW). Males mature between 4 and 5 years at 34–37 cm DW and females at 6–7 years and 43–46 cm DW (Charvet et al., 2018). The estimated longevity is 28.1 years for females and 15.3 years for males (Charvet et al., 2018). The species has an annual reproductive cycle, with copulation during the flooding of the Xingu River and births starting with the dry season. The gestation time is four months, and an average uterine fecundity of almost five embryos/female ranging from 1–11 embryos per female (with an average of 4.84 pups per litter; Charvet, in litt., 2022). The brood size correlates with the mother's size. The birth size is 109–149 mm DW. The generation length was estimated to be seven years. The species exhibits sexual segregation.

Mean natural mortality rate of *P. leopoldi* was 0.27 (range 0.19–0.36) prior to the construction of the Belo Monte Dam. *Potamotrygon leopoldi* is a species with a moderate growth rate, with a difference in growth between males ($k^8 = 0.22$) and females ($k = 0.12$) (Charvet et al., 2018). The intrinsic population growth rate estimated for *P. leopoldi* was 0.065 (Santana, in litt., 2022).

Population trends

The capture of this species for the ornamental fish trade targets mainly young individuals (1–2 years old). Other impacts on populations of the species include fishing for human consumption and other human activities sometimes leading to dead discards or mutilation of individuals (Charvet-Almeida, 2006; Santana, in litt., 2022).

Field surveys carried out in 2004 and 2005 and repeated in 2021 (roughly the same period as three generation lengths), evidenced a population reduction of at least 45% in dip net catches in ornamental fishery areas; Catch Per Unit Effort (CPUE) decreased from 1.4 to 0.7 individuals per hour based on a five-hour observation period/fisher/number of individuals caught (Charvet, in litt., 2022). Other unpublished survey data found changes in CPUE from cast nets, with one survey finding a 7% population reduction over 16 years, and another showing no clear trend (Charvet, in litt., 2022).

It is inferred that *P. leopoldi* has undergone a population reduction of 30–49% over the past three generations (21 years) and it is provisionally assessed as Vulnerable (Charvet, in litt., 2022). An average of these various catch reduction estimates is a decline of 43% from baseline over three generations, which could further decline to 35% of baseline when projected into the near future (10 years if threats continue for this low productivity freshwater ray). Threats are unequally distributed across this species' range, with headwaters and environmental protected areas providing refuge in some parts.

Potamotrygon wallacei Wallace's Freshwater Stingray, Curucu Stingray

Preliminary Assessment: Vulnerable A2cd ver. 3.1 (Araújo, Torres, Sayer, in litt., not yet accepted for publication in the December 2022 Red List update).

Range

A Neotropical freshwater stingray species that is endemic to the middle Rio Negro drainage in Amazonas, Brazil, occurring from Santa Isabel (Téa River) down to Cuieiras River in the vicinity of Manaus. A new population was recently identified around Manaus, Amazonas, outside the native distribution area. The origin of this population is individuals discarded from the aquarium industry (Araújo, 2020a). This species has high habitat specificity and can be found in small creeks and the margins of flooded forest areas (igapós). The freshwater ray occurs only in specific habitats such as small black water streams bordered by flooded forest areas with low pH (pH 3–4), dissolved oxygen levels of 2 mg/l and average water temperature of around 25 °C. This stingray prefers habitat with leafy bottoms and shallow water (50 cm depth). During the hydrological cycle of the Negro River, the habitat of *P. wallacei* undergoes periodic expansions depending on the rainfall. The areas with the highest population density of the species comprise the Itu-Igarapé Daraquá-Bafuana River system, located on the bank of the Rio Negro, 100 km from the Municipality of Barcelos in Amazonas State (Araújo, 1998).

Biology

Potamotrygon wallacei has well-structured stocks and a low gene flow. It reaches a maximum size of 31 cm DW the maximum age observed is five years (Araújo, in litt., 2022). Females mature at 173 mm DW, males at 170 mm DW. The age of maturation is two years, with a gestation time of three months and an average uterine fecundity of two embryos per female (Araújo, 1998, Araújo, 2020a). It has an annual reproductive cycle, with copulation in the ebb of the Negro River and parturition in the dry season. The birth size is 90–100 mm DW. In years of a strong El Niño, there are a rising number of pregnancies per reproductive cycle, and the average fecundity of the population increases to four embryos/female. However, in the following breeding season, the average fecundity of the population is reduced to one embryo per female. The species has sexual and ontogenetic segregation in the flooded forest. Newborns, young of the year, and pregnant females at the end of gestation occur in the same area. Juveniles and adult males occur together, and pregnant females in the early and middle gestation occur in areas with a high oxygen level.

Potamotrygon wallacei is characterised as low productivity given its low fecundity (1 pup/year) and $k = 0.16$ (Araújo, 2022). This fecundity has reduced over time as the average fecundity was measured at 2.25 embryos per female in 1996 (Araújo, 2022).

Population trends

Demographic studies have indicated that if ornamental fishing has an additive effect on the population mortality, it would lead to a reduction in population growth of 3% per year. Recently, following fires in the flooded forest in

⁸ Relative growth coefficient parameter (k), see Charvet et al. (2018).

the Negro River basin, a reduction in population growth of 4% per year was estimated. At least 14 different populations were identified in Middle Rio Negro. In the region of the Itu-Bafuana- Daraquá River System, the population is decreasing by 4% a year. In the other 11 populations on either margin of the Negro River and around Barcelos Municipality the populations have remained stable. The population status in the vicinity of Manaus is unknown.

A significant loss of reproductive potential has been observed in the population since 2015–2016. From 1998–2006, the species reached 31 cm disc width (DW) with a maximum observed age of 8 years. In 2015/2016, after fires occurred in the region with the highest abundance of this species, the maximum size was observed to have reduced to 24 cm DW and maximum age to five years (Araújo, 2022).

P. wallacei is estimated to have undergone a population reduction of 33% over the past three generations (nine years) due to exploitation and habitat degradation (Araújo, in litt., 2022). Given the population dynamics data, population reduction is expected to accelerate and a decline to 13% is projected in the near future (10 years) given reductions in maximum age, fecundity, and few large females (Araújo 2020ab, 2022).

Potamotrygon albimaculata Tapajós Freshwater Stingray

Range and biology

This demersal species is endemic to the middle and upper Tapajós River basin, mainly in the middle and upper Tapajós River (Araújo et al., 2004, Carvalho, 2016). There are several rapids in the Tapajós River where the species can be found nearby, and it may descend the rapids during periods of high-water levels (Carvalho, 2016).

This species reaches a maximum size of 55 cm DW and 79 cm total length (TL) (Carvalho, 2016). Males mature at 35 cm DW, and females at 39.0 cm DW. Males age-at-maturity is three years and females four years with a maximum observed age of 10 years. Reproduction is viviparous with litter sizes of one to four pups (usually two), a biannual reproductive cycle with a four-month gestation period, and size-at-birth of 10–11 cm DW (Carvalho, 2016; Araújo, Sayer, in litt., 2022). Despite two pregnancies per year, this species has shown embryonic malformations, which may reduce fecundity (Araújo et al., 2004). Applying a demographic analysis, the generation length is 5.7 years (Araújo, in litt., 2022).

Population trends

Population information for P. albimaculata was calculated based on validated growth studies and population dynamics models (Araújo and Lessa, in. prep.).

Potamotrygon henlei Henle's Freshwater Stingray

Range and biology

This freshwater ray is endemic to the Tocantins-Araguaia River basin, Brazil, with its primary distribution along the Rio Araguaia and lower Rio Tocantins. It is found throughout the Araguaia River and lower Tocantins River, but only below the confluence with the Araguaia River (Rincon et al., 2014; Carvalho, 2016). This species lives over sandy or rocky areas where it hunts for gastropods and fishes efficiently and can reach shallow tributaries where it aggregates (Rincon, Sayer, in litt., 2022).

The maximum size of this species is 71 cm DW and 104 cm TL. Females mature at around 50 cm DW and approximately 80–85 cm TL. It is viviparous, and fecundity ranges from one to nine embryos per litter. The number of embryos is related to maternal size. Size at birth is approximately 25–30 cm TL. Gestation time and reproductive periodicity unknown. In the area of the Tucuruí Dam reservoir, when the reservoir was established, there were anecdotal reports suggesting that breeding periodicity was increased to more than once a year, probably due to the artificial water level control inside the reservoir. However, this was never confirmed, and this region represents a very limited part of this species' range (Rincon, in litt., 2022).

Population trends

The population status along the Araguaia River is unknown, but it is expected that this species is facing distribution reduction mostly due to environmental degradation (Rincon, Sayer, in litt., 2022).

Potamotrygon jabuti

Range and biology

This species occurs in the mid to upper Tapajós River and is restricted to small tributaries that run lateral to the main channel. Small creeks with clear and cold water are this species' preferred habitat.

This species matures at 39 cm DW for females and 35 cm DW for males with maturation ages of 4 and 3 years respectively. Fecundity is two embryos per female, and the reproductive cycle is annual. The maximum observed age was 10 years, and maximum size is 55 cm DW (Araújo et al., 2004). The generation length is estimated at around six years (Araújo, Sayer, in litt., 2022).

Population trends

Collected for the ornamental aquarium trade. Some of this capture (70%) is parental stock ($DW \geq 39$ cm). Demographic analysis in progress has shown a reduction of generation time and population due to anthropic impacts and fishing (Araújo, Sayer, in litt., 2022).

Potamotrygon marquesiRange and biology

This species has been collected from three different locations in the Amazon River basin: in coastal drainages of the Amapá state, Ilha de Colares, Pará state, and in the Juruá river basin in Acre state (Gama and Rosa, 2015; Silva and Loboda, 2019). It is possibly distributed across other Amazon River tributaries, portions of the Juruá Basin, and in Peru (Araújo et al., in litt., 2022) in drainages and lakes of water high in sediments (Sioli, 1984; Crampton, 2011).

For the Amazon River mouth population, the maximum registered DW is 50 cm for males and 59 cm for females. The minimum recorded size for a pregnant female was 34 cm DW. The average fecundity was three embryos and a maximum of seven embryos (Gama, 2013). In Acre state, populations are smaller and fragmented, with a maximum DW of 32 cm for females and 33 cm for males (Araújo et al., in litt., 2022). The size at first maturation is 22 cm DW for females and 21 cm DW for males. The average fecundity is two (\pm one) and birth size is around 12 cm DW. This species reproduces throughout the year (Gama, 2013), and presents sexual segregation (Araújo et al., 2022). This species is known occasionally to hybridise in the wild with specimens of *P. motoro* (Gama, 2013; Araújo et al., in litt., 2022).

Population trends

This species is abundant along the entire freshwater coast of Amapá (Gama, 2013; Gama & Rosa, 2015) and seems to be less abundant in Juruá River tributaries in Acre state, compared to those in Amapá state (Gama, 2013; Araújo et al., in litt., 2022). The population segment targeted by ornamental fishing is subadult and adult individuals, the captures are concentrated in Cachoeira do Arari Municipality in Pará State, Brazil. Habitat degradation caused by dredging, deforestation, salinisation, and cattle farming are potentially threatening this species. Upper Amazon stocks are inferred to have decreased (Araújo et al., in litt., 2022).

Potamotrygon signata Parnaíba Freshwater Stingray

This freshwater ray is endemic to Parnaíba River drainage in northeastern Brazil, in the states of Piauí and Maranhão (Rosa, 1985; Moro et al., 2016). It is found along the main river and its tributaries, usually at depths less than 3 m, including the dammed waters of the Boa Esperança reservoir. Parnaíba River basin is located for the most part in the semi-arid "Caatinga" biome in northeastern Brazil and is under strong hydrological fluctuations characterised by a long period of drought, with most tributaries to the main river being intermittent (Moro et al., 2016). Juveniles and small adults aggregate in temporary pools formed along intermittent rivers during the dry season. Larger adults are usually found in the main river channel or in marginal lakes (Moro et al., 2016).

P. signata is a medium-sized stingray that reaches 60 cm DW and at least 103 cm TL in females. Size-at-maturity is 17 cm DW for males and 19 cm DW for females. Size-at-birth is estimated to be 11 cm DW (Moro et al., 2016). This species is viviparous with an unknown gestation period and breeding frequency. Litter size ranges from 1–3 embryos per gestation (average 1.4). The generation length is estimated at around seven and a half years (Lucifora et al., 2022).

Population trends

There is no information on population trends. This species tends to aggregate during dry periods, possibly due to habitat and prey availability (Rosa et al., 2009). Local fishermen were reported to kill and mutilate freshwater rays in the Sambito River, with the deliberate poisoning of its waters with pesticides carried out in 2009 by local authorities, to eliminate the stingrays and prevent accidents (Rosa et al., in litt., 2022). This species has a restricted area of occupancy (possibly less than 500 km²) and experiences population fluctuations due to the extreme hydrological cycle in the basin. Habitat quality is inferred to have declined due to human activities (Rosa et al., in litt., 2022).

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

Ornamental trade

Potamotrygon species have been in the ornamental fish trade since the late 1970s. The estimated annual value of trade in freshwater stingrays was over USD20 million between 2001 and 2010 (Duncan et al., 2010). Based on the quota system, since 1998 over 130,000 live freshwater Potamotrygon individuals were exported from the states of Amazonas and Pará, Brazil (Duncan et al., 2010). Factoring in mortality and discards this total may have been 190,000 individuals (Duncan et al., 2010). However, the Amazon Region is an extremely vast area in which quota

enforcement can be difficult. While the quota system did provide some export control, there is ongoing illegal trade from Brazil to neighbouring South American countries (Charvet, in litt., 2022). The illegal fauna trade often uses the same trade routes as illegal drug trafficking (Charvet, in litt., 2022).

The black rays (including *P. albimaculata*, *P. henlei*, *P. leopoldi*, and *P. jabuti*) are the most sought-after ornamental freshwater stingrays in markets in Asia, Europe, and North America. *P. leopoldi* is the most popular stingray in Asia, the USA and Canada followed by *P. jabuti*. Colouration patterns of *P. albimaculata* and *P. leopoldi* hybrids are very popular (Charvet, in litt., 2022). The variety of species and variations is much smaller in the Asian market than in the North American and European markets. The European market seems to be the most complex, as the stingrays come from the wild, from local breeders, and are imported from Asian countries, and from the USA. There is also a larger domestic market than in Asia and North America. *P. leopoldi*, *P. henlei*, and *P. jabuti* are higher priced in North America markets, particularly albinos and hybrids with larger spots and/or atypical dorsal colour patterns.

Recently, the international demand for wild-sourced *Potamotrygon* species in the aquarium trade has declined due to captive breeding production. The hybridisation of *Potamotrygon* species began in 2000 in Asian captive breeding facilities.

In Brazil, the genus listing for *Potamotrygon* in CITES Appendix III came into effect in January 2017.

Potamotrygon leopoldi

The black ray *P. leopoldi* is the most valued species exported and the most frequently encountered in Brazil's official records of stingray seizures—found in some 54.7% of records between 2002 and 2018. In addition, 30% of the stingrays identified as *P. motoro* exported from Amazonas state were *P. leopoldi*. The species was intensively fished for the international ornamental aquarium trade from the 1990s to the early 2010s. Starting in the early 2000s, and intensifying between 2005 and 2011, subadult and adult *P. leopoldi* were illegally captured and exported to form the breeding stock for captive breeding establishments in Asia, the European Union, and North America. Now, Brazilian companies are suppliers of individuals for the ornamental market and suppliers of wild specimens needed to avoid inbreeding in farms and to renew part of the breeding stock. Some high value hybrids can only be obtained by breeding with wild-caught specimens, which increases the pressure on wild populations (Charvet, in litt., 2022). After the Belo Monte Dam was constructed (2015–2016), the main ornamental fishing effort moved upstream to the São Félix do Xingu region. The most sought-after coloration patterns of *P. leopoldi* are those from fishing areas located in São Félix do Xingu.

Potamotrygon wallacei

In the ornamental market, and before its formal taxonomic description (Carvalho et al., 2016), *P. wallacei* was called *P. hystrix*, or *P. hystrix*, or *Potamotrygon* cf. *hystrix* (*P. hystrix* is endemic to the Paraná-Paraguay River basin). In CITES export records, Peru and Colombia appear as exporting countries of *P. hystrix*. This error is because the species had not yet been described when Normative Instruction No. 204/2008 (see management section below) was issued. It needs to be clarified whether these records refer to *P. orbignyi* or *P. wallacei*. The registration of Indonesia as an exporting country of *P. wallacei* in the CITES records may therefore indicate an identification error. In official records from Brazil, there are no records of exports to Indonesia of *P. wallacei* from 2003–2018. It is possible that the species was imported by Indonesia from other countries such as the USA and Germany prior to registration in CITES. *P. wallacei* can sometimes also be confused with *P. marquesi* and *P. signata*. Similarly, *P. marquesi* and *P. signata* may be laundered under the name *P. wallacei*, given the export quotas for *P. wallacei* and the prohibition on exporting *P. marquesi* and *P. signata* (Charvet, in litt., 2022). *P. marquesi* is illegally exported from Brazil and used in hybrid production. Post-capture mortality and discard of some individuals cause the difference observed between the capture number and export numbers of *P. wallacei*.

Other *Potamotrygon* species

Potamotrygon albimaculata has been captured for the ornamental market in the last two decades (Araújo, Sayer, in litt., 2022), as has *P. henlei* (Charvet, in litt., 2022). *Potamotrygon jabuti* is the second most frequent species by number in seizures carried out by the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA).

Trade data

USA imports reported in the LEMIS database of *Potamotrygon* spp. in the most recent ten years, 2011–2020, totalled 6,478 live individuals and bodies; 54 of these were refused and subsequently seized or abandoned. Trade largely comprised *P. leopoldi* (4,692; 72%), and to a lesser extent *P. jabuti* (1,689; 26%). Most of this trade was in captive-born individuals (6,116; 94%), with 249 wild-sourced and 113 ranched individuals. Virtually all trade originated from non-range States, such as Thailand (5,145; 79%), with only 13 live, wild-sourced *P. leopoldi* individuals imported from Brazil for commercial purposes in 2016 and 2019. The value of 11 individuals in 2016 totalled USD4,499 (USD409 per individual). Total wild exports from Brazil to the USA (including *Potamotrygon* spp. and those identified to species level) peaked in 2012 (Figure 1).

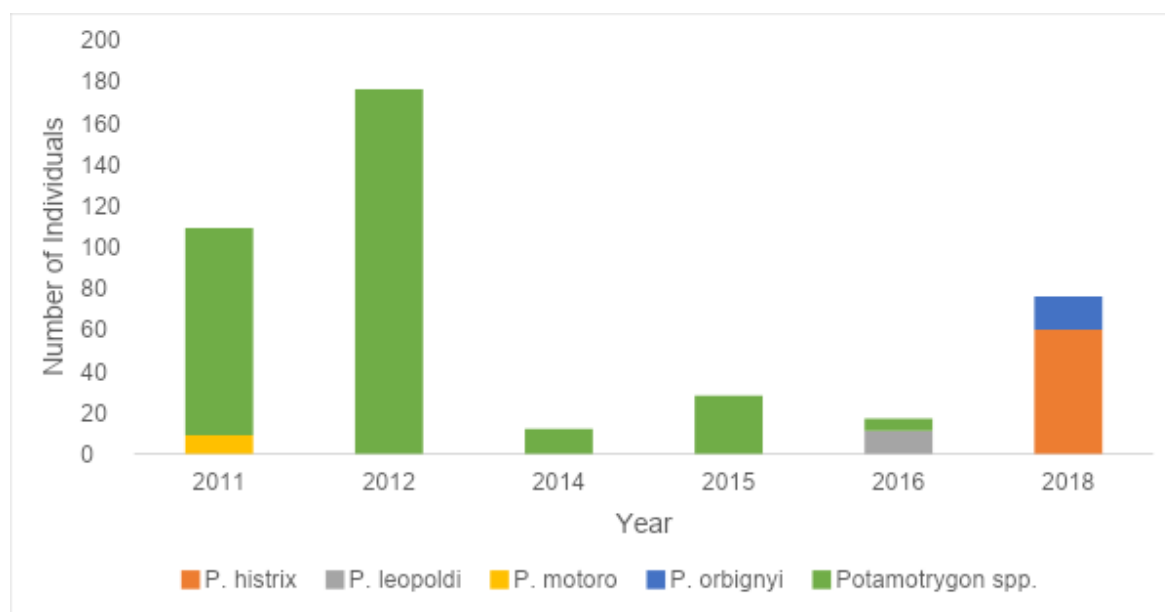


Figure 1. Summary of live wild-sourced *Potamotrygon* spp. individuals exported for commercial purposes from Brazil to the USA, 2011–2020. A 2019 import was refused and is not included here (LEMIS).

After the inclusion of the species in Appendix III of CITES in 2017, there were no *species-specific* records of exports of *P. leopoldi* from Brazil. In the CITES records *most of the export of P. leopoldi is from Asian countries. The CITES Trade Database (2017–2020) included no species-specific trade records for P. albimaculata, P. marquesi, P. signata, or P. wallacei and no wild specimens reported for P. henlei or P. jabuti. There were wild-caught records available for P. leopoldi, as well as P. histrix and P. orbignyi (both of which may possibly be P. wallacei given the absence of records under that species' name). P. wallacei is a recently described species and was previously named P. histrix (Carvalho et al., 2016). The CITES Trade Database includes records of unspecified Potamotrygon spp. (1,134 live wild imports or transiting through the USA between 2008 and 2016). The Supporting Statement to the Proposal includes a summary of export data of freshwater stingray from Brazil from the IBAMA database (2003–2016).*

In 2018, there was a single record of 23 wild-caught P. leopoldi individuals being imported to the USA from Thailand (origin unknown). In 2018, all exports of P. histrix were from Brazil except for a single import record from Peru to Denmark (13 individuals). Denmark was the largest importer, with Brazil reporting 580 individuals exported (400 fingerlings (neonates) and 180 individuals)—although Denmark only reported importing 463 individuals from Brazil. (Neonates with a 10–12 cm DW are called “fingerlings” in this database (Araújo in litt., 2022)). Denmark was followed by Japan (32 neonates) and the USA (40 neonates, 20 individuals). A handful of wild-caught exports from Brazil reported as P. orbignyi may be P. wallacei. All P. orbignyi exports were in low volumes (3–25 individuals or fingerlings) except for a single export record from Brazil to the USA of 6,010 individuals, which was recorded as an import of 16 individuals by the USA. It seems possible that this Brazilian entry in the CITES Trade Database is an error, given the large discrepancy in import and export numbers and no such significant import recorded in the LEMIS Database in 2018. This volume of import would represent a ten-fold increase from the USA's previous maximum import volume of approximately 600 individuals in 2009. While the volume traded remains uncertain, an analysis by Araújo and Prang identified these individuals as P. marquesi. Potamotrygon marquesi has some reticulate coloration patterns and can be misidentified as P. orbignyi or as P. motoro. It may also be misidentified as P. albimaculata, which is from the black ray group and is illegal to export from Brazil. Potamotrygon albimaculata may show a reticulate coloration pattern in some juveniles, which is typical of P. orbignyi (Araújo, in litt., 2022).

In summary, according to available trade databases from the USA (LEMIS; Eskew et al., 2020), CITES, and from IBAMA (Brazil, as reported in the Supporting Statement), the recent volumes of legally wild-caught Potamotrygon species exported from Brazil are low and well below the quotas set by the Brazilian government (775 individuals exported across all species in 2018 if the the likely erroneous USA record is excluded).

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

All species of the genus *Potamotrygon* exhibit polychromatism, which can make identification challenging to an untrained observer. The two major groups of brown and black rays are summarised in Table 1.

Table 1. Species in the genus *Potamotrygon* that may meet the criteria for listing in Annex 2b A of Resolution Conf. 9.24 (Rev. CoP17) based on major groupings.

| Species | Annex 2b |
|----------------------------------|--|
| <i>Potamotrygon wallacei</i> | "Brown ray group" (<i>P. wallacei</i> , <i>P. signata</i> , and <i>P. marquesi</i>) |
| <i>Potamotrygon leopoldi</i> | "Black ray group" (<i>P. leopoldi</i> , <i>P. albimaculata</i> , <i>P. henlei</i>) Some juvenile colour patterns may resemble <i>P. motoro</i> |
| <i>Potamotrygon albimaculata</i> | "Black ray group" (<i>P. leopoldi</i> , <i>P. albimaculata</i> , <i>P. henlei</i>). In particular, <i>P. albimaculata</i> is black with white spots and can be confused with <i>P. leopoldi</i> , which also is black with white spots, although the spots on <i>P. albimaculata</i> are smaller |
| <i>Potamotrygon henlei</i> | "Black ray group" (<i>P. leopoldi</i> , <i>P. albimaculata</i> , <i>P. henlei</i>) |
| <i>Potamotrygon jabuti</i> | Some colour patterns resemble those of the "Black ray group" (<i>P. leopoldi</i> , <i>P. albimaculata</i> , <i>P. henlei</i>) |
| <i>Potamotrygon marquesi</i> | "Brown ray group" (<i>P. wallacei</i> , <i>P. signata</i> , and <i>P. marquesi</i>) |
| <i>Potamotrygon signata</i> | "Brown ray group" (<i>P. wallacei</i> , <i>P. signata</i> , and <i>P. marquesi</i>) |

Information provided by Araújo in litt., 2022.

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Additional Information

Threats

Additional threats include habitat loss and degradation (deforestation, wildfires in Amazon Forest, construction and operation of hydroelectric power plants, agricultural expansion) and local consumption or persecution.

Habitat loss and degradation

Potamotrygon leopoldi has restricted geographical distribution and faces the loss of habitat integrity in the Xingu River basin, caused by the growth of the agricultural frontier, an increase in ranching, mining, and construction of the Belo Monte Dam and hydroelectric power plant. Floods from development projects have interfered with the hydrodynamics of the Xingu River and changed the biotic and abiotic factors of the area. This has caused the disappearance of an essential habitat of *P. leopoldi* in the Xingu River and reduced female minimum maturation size. An increase in mining in the Xingu River has led to contamination by heavy metals (mainly mercury) and increased turbidity in the mid and upper Xingu River, leading to embryo malformations for *P. leopoldi*. Climate change has altered the length of the wet and dry seasons, caused fluctuations in rainfall, and increased the mean water temperature along the Xingu River basin by at least 2 °C over the past 20 years. Climate change is also a concern for *P. wallacei*.

The essential flooded forest habitat of *P. wallacei* was reduced by 30% due to fires that occurred in the Rio Negro basin from 2015–2016. This led to an observed reduction of the generational length from 3.9 to 2.9 years, a reduction of the maximum observed disc width from 31.0 cm to 24.0 cm and an absence of females older than five years old.

One of the main areas of *P. jabuti* occurrence is the Jamanxin River, a traditional gold mining region (Araújo et al., 2004; Carvalho, 2016). About two-thirds of the preferred habitat of this species is in areas of illegal gold mining activity and it is profoundly impacted by the consequences of the mining pollution and silting in the river. *Potamotrygon albimaculata* is also exposed to mining contaminants.

Domestic consumption

The targeting of *P. leopoldi*, mainly adults, for human consumption in parts of the Xingu drainage has increased in recent years. Other stingrays are discarded or have their tails mutilated by fishers. *P. wallacei* is subject to tail mutilation or sometimes killed by local riverine people to prevent sting injuries.

Conservation, management and legislation

National

IBAMA prohibited the export of freshwater stingrays for ornamental purposes in 1990. Subsequent regulations to determine a quota system for specific *Potamotrygon* species for Amazonas state were established in 1998 (regulation No 022/98). The initial quotas were 2,000 units and were based on the reproductive potential of the species, fishing effort, and distribution area, reinforced by existing knowledge about the dynamics of other species

of *Potamotrygonidae*, such as *P. wallacei* (Araújo, in litt., 2022). More species were added to the list in 2003 (regulation No.036/2003) when a bi-state regulation was established for Amazonas and Pará states. After 2005, the adjustment of quotas occurred based on population dynamics studies by Charvet-Almeida (2006), who estimated natural and fishing mortality rates based on an age structured model (Araújo, in litt., 2022). Current legislation (No. 204/2008; MA/IBAMA, 2008) determines a quota for five species (specifically: *P. henlei*, *P. leopoldi*, *P. orbignyi*, *P. schroederi*, and *P. wallacei*), which can be exported legally from Brazil (see Table 1). It has been reported there is no effective control over export of species not listed in Brazilian legislation (Duncan et al., 2010). Furthermore, the quota system is reported to have poor compliance and monitoring and the species are at risk from handling and transportation that can increase mortality. *Illegal trade is ongoing, and trafficking occurs with Brazilian endemic species being exported from neighbouring countries* (Charvet, in litt., 2022). There is no regulation for freshwater stingray species under fishing pressure for human consumption.

Potamotrygon leopoldi is included in the National Plan for Threatened Amazonian fishes. However, no conservation measures have been implemented. At least 25% of the *P. leopoldi* distribution area is in the Xingu River Extractive Reserve, a protected area in Altamira Municipality, but there are no conservation measures for the species in the reserve management plan. In the *P. wallacei* distribution area, there are two fully protected conservation units in the lower Rio Negro basin, but there is no registered *P. wallacei* occurrence. *Despite having part of their distribution ranges in reserves or protected areas, monitoring and enforcement is challenging* (Charvet, in litt., 2022).

Table 2. Quotas for five *Potamotrygon* species in Brazil (legislation No. 204/2008; MMA/IBAMA, 2008).

| Scientific Name | Max Disc Width (cm) | Quotas | | |
|---|---------------------|----------|-------|--------|
| | | Amazonas | Pará | Total |
| <i>Potamotrygon motoro</i> | 30 | 4,000 | 1,200 | 5,200 |
| <i>Potamotrygon cf. hystrix</i> (<i>Potamotrygon wallacei</i>) | 14 | 6,000 | - | 6,000 |
| <i>Potamotrygon schroederi</i> | 30 | 1,000 | - | 1,000 |
| <i>Potamotrygon orbignyi</i> | 30 | 1,200 | 1,200 | 2,400 |
| <i>Potamotrygon cf. henlei</i> | 30 | - | 1,000 | 1,000 |
| <i>Potamotrygon leopoldi</i> | 30 | - | 5,000 | 5,000 |
| Total | | 12,200 | 8,400 | 20,600 |

International

All Brazilian populations of *Potamotrygon* spp. are listed in CITES Appendix III, which came into effect in 2017.

Captive breeding

The international demand for wild-caught specimens for the aquarium trade has declined due to captive breeding facilities supplying hybrids to the aquarium market. The hybridisation by Asian breeders began in the early 2000s. By 2020, they had had two decades of cross-breeding to create animals of varying patterns and colours, some of which have value far above the animals from the wild. The degree of hybridisation of *P. leopoldi* in the Asian market is more significant than in the North America and European markets. *Potamotrygon leopoldi* is more targeted for cross-breeding than any other freshwater stingray species.

Compared to other stingrays, reproduction in captivity is difficult for *P. wallacei*. However, captive breeding of *P. wallacei* on a small-scale has occurred in the United Kingdom, Germany, and in the Netherlands. In addition, the production of hybrids with *Potamotrygon motoro* has been reported in the USA and the UK. There is a shrinking market for *P. wallacei* in Asia because there is difficulty in reproducing this species in captivity on a large scale. A total of five companies in Brazil requested an exportation quota of 4,498 specimens for the export of *P. wallacei* in 2022. This represents 75% of the current quota allowed for export (6,000 units) but exceeds the new suggested quota of 2,500 individuals. *However, there is no indication that these five companies will fulfil this quota.*

Implementation challenges (including similar species)

Multiple drivers appear to be causing population reductions, and those related to habitat loss and domestic consumption will continue even if international trade is regulated. This could undermine the effectiveness of any

listing. Hybrid individuals will need to be treated as specimens of species included in Appendix II, as they fall under interpretations set out under Res. Conf. 10.17 (Rev CoP14), which may make enforcement challenging.

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Inclusion of Guitarfish (Family Rhinobatidae) in Appendix II

Proponents: Israel, Kenya, Panama, Senegal

Summary: Guitarfish are shark-like rays in the Order Rhinopristiformes and family Rhinobatidae. Three other families within the Order Rhinopristiformes are listed in Appendix I (Pristidae at CoP14) and Appendix II (Glaucostegidae and Rhinidae at CoP18). There are 37 species within the family Rhinobatidae in three genera (*Acroteriobatus*, *Pseudobatos*, and *Rhinobatos*). These demersal species occur mostly in shallow (<50 m) coastal continental shelves to depths of up to 366 m.

Six of the species now proposed for listing (*Acroteriobatus variegatus*, *Pseudobatos horkelii*, *Rhinobatos albomaculatus*, *Rhinobatos irvinei*, *Rhinobatos rhinobatos* and *Rhinobatos schlegelii*), have recently been assessed as Critically Endangered on the IUCN Red List. These species have undergone population declines suspected or inferred to be at least 80% over the past three generations, except *A. variegatus* for which declines are projected. These six species are proposed for listing in Appendix II in accordance with Article II, paragraph 2(a) of the Convention (from here onwards referred to as lead species). All remaining species in the family are proposed in accordance with Article II paragraph 2(b) (as lookalike species).

Maximum total length ranges from 65 cm (*A. variegatus*) to 138 cm (*P. horkelii*). The shortest inferred generation length is 5 years (*A. variegatus*), and the longest is 18.5 years for *P. horkelii*. *A. variegatus*, *R. albomaculatus*, and *R. irvinei* typically produce relatively small litters ranging from 1–4 pups, while *P. horkelii*, *R. rhinobatos* and *R. schlegelii* produce larger litters on average, ranging from 1–14 pups. Maximum annual intrinsic rates of population increase have been estimated for *P. horkelii* and *R. rhinobatos*, and these rates suggest moderate and moderate-to-high productivity relative to other guitarfishes. However, relative to aquatic fishes more generally, productivity is medium-to-low for the family Rhinobatidae.

Three of the lead species, have restricted distributions within the western Indian Ocean (*A. variegatus*), southwestern Atlantic Ocean (*P. horkelii*) and northwestern Pacific Ocean (*R. schlegelii*). *R. albomaculatus*, *R. irvinei*, and *R. rhinobatos* have larger overlapping distributions from Angola to Mauritania, with *R. irvinei* extending northwards to Morocco, and *R. rhinobatos* extending into the Mediterranean Sea.

The primary threat to the family Rhinobatidae is targeted or incidental catch for meat and fins in unmanaged and unregulated fisheries. Due to their coastal, inshore nature, these species are often caught in a variety of fishing gear especially artisanal and semi-industrial gillnets, trawls, line, trammel nets and seine nets, including as incidental catch in demersal trawls and gillnets.

Global population sizes are unknown for all species in the family. There is little species-specific information, and recent taxonomic revisions mean that available data for Rhinobatidae are sometimes inferred from other Rhinidae and Glaucostegidae species. Available catch-per-unit-effort (CPUE) data for *P. horkelii* in Brazil, as well as decreased catches in Uruguay, suggest population reductions of >99% and >80% over the last three generations, respectively. When available data for *R. schlegelii* are scaled to three generation lengths, they represent reductions of 63%, 88%, 40%, and 90% in Taiwan POC, Japan, China, and the Republic of Korea, respectively.

For other species, information on population trends is patchy. There is evidence of population reductions for *R. rhinobatos* in Senegal (90% reduction in landings over three generation lengths), Mauritania (85% population reduction over three generation lengths) and Ghana with fishers estimating 80–90% decreases. *Rhinobatos rhinobatos* appears to be rare in Cameroon and absent in Angola. In the northern part of its range, *R. rhinobatos* is considered locally extinct in western and central Mediterranean waters. It is still caught and is considered relatively common in the eastern Mediterranean, especially in Lebanon.

In Ghana, *R. irvinei* is relatively common, although Guitarfish fishers interviewed reported a 40–60% decrease in catches. In these target fisheries, *R. irvinei* was the most landed species (70% of the relative catch). *Rhinobatos irvinei* was found to be present in low numbers during landing surveys in Cameroon and Angola.

While *R. albomaculatus* was absent during trawl surveys from Guinea. In the southern extent of this species' ranges, it appears to be relatively common in Cameroon (7% of all shark and ray records over a two-year survey).

No specific population trend data are available for *A. variegatus*; general reductions in Rhinidae, Glaucostegidae, and Rhinobatidae have been used to infer future population declines.

Information on international trade in guitarfish fins is sparse. The fins of Rhinobatidae, Rhinidae, and Glaucostegidae are considered together as a single category known as “white” shark fins or “Qun chi”, which has one of the highest values of any fin category. However, references to trade in any members of the Rhinobatidae are rare. Researchers have reported the presence of *A. variegatus* in a 2020 survey of low-value fins in Hong Kong SAR, noting that this was the first time this species had been recorded in trade despite extensive earlier sampling (over 9,000 samples between 2014 and 2016). There are also reports that in Ghana fins of landed *R. s rhinobatos* are detached and sold separately to traders from Nigeria, Togo, Mali, The Gambia, and Senegal, presumed to be supplying Asian markets. There is also evidence from fish market interviews of small guitarfish fins, likely of *R. albomaculatus* and *R. irvinei*, being sold. However, quantitative data are lacking and these species have not been detected in fin hub surveys. The meat and fins of *R. schlegelli* are low-value and are likely consumed domestically. There is no evidence of *P. horkelii* in trade, and meat, which is considered valuable, is consumed domestically.

National instruments in place for Rhinobatidae vary by range State. Some instruments are species-specific, and others may indirectly benefit Rhinobatidae species (such as finning bans, fishing gear restrictions, and area and closed seasons). The Mediterranean population of *R. rhinobatos* is included in several environmental agreements including the Convention on Migratory Species (CMS), where it is listed in Appendix I, and the Barcelona Convention.

Analysis: Guitarfishes are, or have been, subject to targeted and incidental fisheries throughout their range. Of the six species proposed as lead species five are already suspected or inferred to have undergone marked declines, with *A. variegatus* considered to have been impacted by intense fishing and projected to decline markedly by 2032. These species would appear to meet the biological criteria for inclusion in Appendix I in Res. Conf. 9.24 (Rev CoP17). Any trade could be considered to require regulation meeting the criteria for inclusion in Appendix II in Annex 2aA of the Resolution. There is very little quantitative evidence of products (fins) of any of these species in fin hub surveys, although there are reports from fishers of *R. rhinobatos*, *R. albomaculatus* and *R. irvinei* apparently entering trade. *Acroteriobatus variegatus* was identified in surveys in fin markets in Hong Kong SAR for the first time in 2020. While some of the targeted harvest of these species may be driven by international trade, domestic consumption is likely to be the dominant driver.

Within the family Rhinobatidae, it can be challenging to identify parts and derivatives to species level. Juveniles of Rhinidae and Glaucostegidae are often difficult to distinguish from Rhinobatidae both whole and when traded as parts and derivatives. Parts and derivatives of juvenile Pristidae are also difficult to distinguish from Rhinobatidae. As such, all members of the family Rhinobatidae meet the criteria for listing in Annex 2bA of Res. Conf. 9.24 (Rev CoP17) based on the difficulty of distinguishing whole carcasses as well as parts and derivatives from those of species already listed in the Appendices (Rhinidae, Glaucostegidae, and Pristidae). This would ensure that already listed species could not be traded illicitly under the names of non-listed Rhinobatidae species that resemble them.

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

Rhinobatidae

Lead species: *Acroteriobatus variegatus*, *Pseudobatos horkelii*, *Rhinobatos albomaculatus*, *Rhinobatos irvinei*, *Rhinobatos rhinobatos*, *Rhinobatos schlegelii*

All other species in the family are proposed for listing as lookalikes.

Range and IUCN Global Category

Stripenose Guitarfish *Acroteriobatus variegatus*

Global – Critically Endangered (assessed 2017, ver 3.1) (Kyne et al., 2017)

Occurs in the Arabian Seas region and in southern India and Sri Lanka. A small species reported from the continental shelf to 366 m (Weigmann, 2016), but typically prefers shallow waters of 10–40 m depth (Last et al., 2016), including coral habitats. *Acroteriobatus variegatus* is found in the following FAO Fishing Areas: Western Indian Ocean. Range countries and territories are: India (Tamil Nadu); Sri Lanka.

Brazilian Guitarfish *Pseudobatos horkelii*

Global – Critically Endangered (assessed 2019, ver 3.1) (Pollom et al., 2020)

This species occurs in the southwestern Atlantic from Rio de Janeiro, Brazil, to northern Argentina. It is a demersal shark-like ray that inhabits soft substrates of the continental shelf, from inshore to 150 m depth. Its seasonal migration and breeding cycle in southern Brazil are described below, from data published by Lessa (1982), Lessa et al. (1986) and Vooren et al. (2005). In southern Brazil, the adults migrate to coastal waters with depths of less than 20 m from November to March. Parturition and mating take place in March. Soon after, the males and females return to deeper waters and disperse to depths of 40–150 m over the continental shelf. Newborn pups and juveniles remain in shallow waters throughout the year. *Pseudobatos horkelii* is found in the following FAO Fishing Areas: Southwest Atlantic. Range countries and territories are: Argentina; Brazil; Uruguay.

Whitespotted Guitarfish *Rhinobatos albomaculatus*

Global – Critically Endangered (assessed 2020, ver 3.1) (Jabado et al., 2021a)

Occurs in the eastern central Atlantic and southeastern Atlantic from Mauritania to Angola and is demersal in shallow coastal waters on the inner continental shelf to a depth of ~132 m. *R. albomaculatus* is found in the following FAO Fishing Areas: Eastern Central Atlantic, Southeast Atlantic. Range countries and territories are: Angola; Benin; Cameroon; Congo; Côte d'Ivoire; Democratic Republic of the Congo; Equatorial Guinea; Gabon; Gambia; Ghana; Guinea; Guinea-Bissau; Liberia; Mauritania; Nigeria; Senegal; Sierra Leone; Togo.

Spineback Guitarfish *Rhinobatos irvinei*

Global – Critically Endangered (assessed 2020, ver 3.1) (Jabado et al., 2021b)

This species is demersal and inhabits shallow coastal waters on the inner continental shelf to a depth of ~49 m. It occurs in the eastern central Atlantic and southeastern Atlantic from Morocco to Angola. *R. irvinei* is found in the following FAO Fishing Areas: Eastern Central Atlantic, Southeast Atlantic. Range countries and territories are: Angola; Benin; Cameroon; Congo; Côte d'Ivoire; Democratic Republic of the Congo; Equatorial Guinea; Gabon; Gambia; Ghana; Guinea; Guinea-Bissau; Liberia; Mauritania; Morocco; Nigeria; Senegal; Sierra Leone; the non-self-governing territory of Western Sahara; Togo.

Common Guitarfish *Rhinobatos rhinobatos*

Global – Critically Endangered (assessed 2020, ver 3.1) (Jabado et al., 2021c)

This species is demersal across sandy, muddy, and shell-rich habitats, and occurs inshore on the continental shelf to a depth of 180 m (Ebert and Stehmann, 2013). It occurs in the Mediterranean Sea and in the eastern Atlantic Ocean from the southern Bay of Biscay to Angola. *Rhinobatos rhinobatos* is found in the following FAO Fishing Areas: Eastern Central Atlantic, Southeast Atlantic, Northeast Atlantic, Mediterranean and Black Sea. Range countries and territories are: Albania; Algeria; Angola; Benin; Bosnia and Herzegovina; Cabo Verde; Cameroon; Congo; Croatia; Cyprus; Côte d'Ivoire; Democratic Republic of the Congo; Egypt; Equatorial Guinea; France (Corsica); Gabon; Gambia; Ghana; Gibraltar; Greece; Guinea; Guinea-Bissau; Israel; Italy (Italy (mainland), Sardegna, Sicilia); Lebanon; Liberia; Libya; Malta; Mauritania; Monaco; Montenegro; Morocco; Namibia; Nigeria; Portugal; Senegal; Sierra Leone; Slovenia; Spain; Syrian Arab Republic; the non-self-governing territory of Western Sahara; Togo; Tunisia; Turkey (European part).

Brown Guitarfish *Rhinobatos schlegelii*

Global – Critically Endangered (assessed 2019, ver 3.1) (Rigby et al., 2021)

Occurs in the northwestern Pacific Ocean from Japan to Taiwan POC, including the Republic of Korea and mainland China and is demersal on the continental shelf at depths of 1–230 m. *Rhinobatos schlegelii* is found in the following FAO Fishing Areas: Northwest Pacific. Range countries and territories are: China; Japan; Republic of Korea; Taiwan POC.

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

Biology

A simple but robust estimator of natural mortality (M) for sharks and rays is the reciprocal of average lifespan ($M=1/w$) where average lifespan “ w ” is the generation length of the species (Pardo and Dulvy, 2016; Pardo et al., 2016). Hence, these CITES mortality thresholds can be back converted to calculate the corresponding generation length productivity thresholds ($GL = M^{-1}$) and these are as follows: high productivity (natural mortality >0.5 or a generation length of <2 years), medium productivity (natural mortality $0.2–0.5$ or generation lengths $2–5$ years), and low productivity (natural mortality <0.2 or generation length >5 years).

Acroteriobatus variegatus

Reproduction is presumably lecithotrophic viviparous, but relatively little is known of the biology of the species. Pregnant females occur from 58 cm total length (TL). Maximum size is 65 cm TL for males and 75 cm TL for females. Litter size mostly 1–4, but occasionally up to six. Generation length is inferred to be five years from the Lesser Guitarfish (*Acroteriobatus annulatus*) from southern Africa (Compagno et al., 1989).

Pseudobatos horkelii

Reaches a maximum size of 138 cm TL; females mature at 86–91 cm TL and males at 70 cm TL (Lessa et al., 1986, Martins et al., 2018). Reproduction is viviparous and females give birth to 4–12 pups (Last et al., 2016). This species also exhibits embryonic diapause, where fertilised eggs held by the female enter a period of arrested embryonic development for several months (Waltrick et al., 2012). While the full mechanisms are not understood, elasmobranch species that employ embryonic diapause have above average litter sizes and so higher productivity (Simpfendorfer, 1992; Waltrick et al., 2012). This species has an estimated age-at-maturity of nine years and a maximum age of 28 years, and thus a generation length of 18.5 years (Lessa et al., 1986, Vooren et al., 2005). It has a moderate maximum annual intrinsic rate of population increase compared to other guitarfishes, with median estimates of 0.13–0.26 (D’Alberto et al., 2019).

Rhinobatos albomaculatus

Reaches a maximum size of 80 cm total length (TL) with males mature at ~46 cm TL and females mature at ~52 cm TL. Reproduction is possibly lecithotrophic viviparous with litter sizes of 2–3 pups and a size-at-birth of ~15 cm TL (Last et al., 2016). Little else is known of its biology. There is no information on this species’ age-at-maturity and maximum age, hence, generation length was inferred based on data for *R. rhinobatos*. *Rhinobatos rhinobatos* has an age-at-maturity of four years and a maximum age of 24 years and, thus, a generation length of 14 years (Başusta et al., 2008). *Rhinobatos albomaculatus* has a smaller maximum size, and thus based on scaled-size, the generation length is inferred to be 8 years for *R. albomaculatus*.

Rhinobatos irvinei

Reaches a maximum size of 100 cm TL with males mature at ~42 cm TL and the size-at-maturity of females unknown. Reproduction is possibly lecithotrophic viviparous with litter sizes of 1–3 pups (Last et al., 2016). Little else is known of its biology. There is no information on this species’ age-at-maturity and maximum age, hence generation length was inferred based on data for *R. rhinobatos*. *Rhinobatos rhinobatos* has an age-at-maturity of four years and a maximum age of 24 years and, thus, a generation length of 14 years (Başusta et al., 2008). *Rhinobatos irvinei* has a smaller maximum size (100 cm TL) than *R. rhinobatos* (162 cm TL), and thus based on scaled size, the generation length is inferred to be 10 years for *R. irvinei*.

Rhinobatos rhinobatos

Reaches a maximum size of 162 cm TL. Males mature at 56–79 cm TL and females mature at 64–85 cm TL (Capapé et al., 1996, Last et al., 2016). Reproduction is lecithotrophic viviparous with litters of 1–13 pups born after a gestation period of 10–12 months at a size-at-birth of 25–30 cm TL (Enajjar et al., 2008, Last et al., 2016). This species also exhibits embryonic diapause, where fertilised eggs held by the female enter a period of arrested embryonic development for several months (Waltrick et al., 2012). While the full mechanisms are not understood, elasmobranch species that employ embryonic diapause have above average litter sizes and so higher productivity (Simpfendorfer, 1992; Waltrick et al., 2012). The age at maturity is two to three years and maximum age is at least 24 years; generation length is therefore 14 years (Ismen et al., 2007, Başusta et al., 2008). It has a moderate-to-high maximum annual intrinsic rate of population increase compared to other guitarfishes, with median estimates of 0.10–0.53 (D’Alberto et al., 2019).

Rhinobatos schlegelii

Reaches a maximum size of ~100 cm TL. Males mature at ~55 cm TL and size-at-maturity of females is unknown (Last et al., 2016). Reproduction is aplacental viviparous with litter sizes of 1–14 and a likely gestation period of 12

months (Yamada et al., 2007, Last et al., 2016). Age parameters are unknown but based on estimated generation lengths of other rhinoprismoid rays, generation length is estimated as 10 years (Kyne et al., 2020). This is an estimate only and research is required on the age parameters of this species.

Population size

Global population sizes are unknown.

Population trends

Acroteriobatus variegatus

This species is regularly caught in southern India. There are no species-specific data available on this species, although elasmobranchs are heavily exploited in Tamil Nadu and Kerala. Significant reductions in wedgefish and guitarfish (including rhinobatids) landings have been documented in Tamil Nadu through monitoring at Chennai (Mohanraj et al., 2009). Even though this is northeast of the range of *A. variegatus*, trawlers in Tamil Nadu fish widely throughout southern India (Karnad et al., 2014) and data can be considered representative of the broader area. Wedgefish and guitarfish landings decreased by 86% over the five years of monitoring (2002–2006), this is the equivalent to a >97% local reduction for *A. variegatus* over the last three generation periods (15 years).

Data available from Maharashtra, although outside the area of occurrence of this species, further demonstrate the reductions in inshore batoid landings. There, the annual average catch of rays landed by trawlers at New Ferry Wharf, Mumbai, between 1990–2004 was 502 t. During this period trawler hours doubled, and consequently, the catch rate decreased by 60% from 0.65 kg/h in 1990 to 0.24 kg/h in 2004 (Raje and Zacharia, 2009). This would equate to an overall reduction of approximately 60% over a period of three generations for the species. Ongoing intense fishing pressure, as well as reduced quality of coral reefs raise serious concerns for this species, and a future population reduction is suspected over the next three generations (2017–2032). The species is therefore assessed as Critically Endangered A2cd+3cd (Kyne et al., 2017).

Using a simple average of these two population reductions (99% and 63%) scaled over the past three generation lengths (15 years), these data suggest a global depletion to 19% of baseline. However, the steeper reduction is likely to be more fully representative of the species' entire range. If global depletion to 19% was projected into the near future (10 years), it suggests a depletion to 9% of baseline for this medium productivity species.

Pseudobatos horkelii

Commercial and artisanal fisheries pressure is intense on the southern Brazilian shelf off Rio de Janeiro and Sao Paulo, and it is likely that a steep population reduction of this species has occurred there. In Rio Grande do Sul, Brazil, total landings have decreased from 1,804 t in 1984 to 157 t in 2001, which is equivalent to a reduction of approximately 99% scaled over three generations (55.5 years). The average trawl CPUE of *P. horkelii* in southern Brazil over the years 1993–1999 was 17% of that observed during 1975–1986, indicating a decrease in abundance of >80% since 1986 in southern Brazil (Miranda and Vooren, 2003, Vooren et al., 2005), equivalent to a reduction of >99% scaled over three generations. While formerly abundant, this guitarfish was scarce in coastal waters by 2004. Fishing pressure has not ceased in Brazil. Despite protection, this species is still landed and traded, and a further reduction in population size is suspected. In Uruguay, the catches from research trawls in the 1980s and early 1990s were on average about 1,400 kg/h, and between 2013 and 2017 were just over 480 kg/h, equivalent to a 94% reduction over three generations. Overall, the species has undergone a population reduction of >80% over the past three generations (55.5 years). It is assessed as Critically Endangered A2bd (Pollom et al., 2020).

Available population reduction information suggests a global depletion to <1–6% over the past three generation lengths (55.5 years) for this low productivity species.

Rhinobatos albomaculatus

There are no species-specific time-series data available for guitarfish species that can be used to estimate population reduction. This is due to a lack of species-specific reporting as well as taxonomic and identification issues (Jabado, 2018). Given the lack of reporting from artisanal fisheries and the large number of nations fishing in African waters, actual landings are likely to be much higher than reported. Although *R. albomaculatus* was never very abundant across the Eastern Central Atlantic and Southeast Atlantic, it has become increasingly rare, particularly in landings (Jabado et al., 2021a). There have been limited records of this species in the past decade from across the region. In Mauritania, between 2010 and 2019, only two specimens were recorded in regular fisheries monitoring surveys undertaken by the Institut Mauritanien de Recherches Océanographiques et de Pêches in the Parc National du Banc d'Arguin (Jabado et al., 2021a). In Gambia, this species was not recorded during landing site surveys conducted annually between 2010 and 2018 despite other species of guitarfish being present (Moore et al., 2019). Similarly, there have been no recent records of this species in landing site surveys in Guinea-Bissau (Bijagos Archipelago), Côte d'Ivoire, Nigeria, and Angola (Jabado et al., 2021a). Between 2004 and 2011, this species was only recorded in Guinea during landing site surveys across the Sub-Regional Fisheries Commission region (Diop and Dossa, 2011) but more recently it has not been found despite regular monitoring (Jabado et al., 2021a).

In Ghana, trawl surveys operated at depths between 20 and 100 m recorded four specimens in four tows (2.1% of sets) (Ishihara and Kimono, 2006). Landing and market sites surveys in Ghana in 2020 and 2021 reported that *R. irvinei* was the most landed species (71% of the relative catch), followed by *R. albomaculatus* (16%), and *R. rhinobatos* (6%) (Seidu et al., 2022a). *Rhinobatos albomaculatus* was not recorded in western Ghana (Seidu et al., 2022b). Cruise reports from the "Dr. Fridtjof Nansen" surveys indicate that this species was frequently caught in 2004 (Congo, Gabon, and Angola), 2006 (Nigeria, Cameroon, São Tomé and Príncipe, Gabon, and Congo), 2007 (Angola) and 2008 (Côte d'Ivoire, Ghana, Benin, Togo, Cameroon, São Tomé and Príncipe, Gabon, and Congo) particularly in the waters off Gabon where, when captured, it represented between 1.13 and 9.96% of the total catch per tow with up to 102 individuals caught in one tow (Krakstad et al., 2004, Krakstad et al., 2006, Krakstad et al., 2008). Subsequent surveys undertaken in Gabon and Congo in 2010 failed to record this species (Mehl et al., 2010). In the trawl fisheries observer data from Gabon, this species along with other *Rhinopristiformes* represented 1% of rays captured. Although captured throughout the year in artisanal fisheries operating in Mayumba using demersal-set gillnets, records were rare (Jabado et al., 2021a). In the Congo, 89 individuals were recorded during landing site surveys from January–December 2019 representing less than 1% of all shark and ray landings (Jabado et al., 2021a). In Cameroon, 146 individuals were recorded over two years of landing site surveys representing 6.6% of all shark and ray records (Jabado et al., 2021a). *Rhinobatos albomaculatus* was reported in only half of guitarfish fishing communities surveyed in Ghana, where 59% of interviewed fishers reported their catches of the smaller guitarfish species, including *R. albomaculatus*, have undergone population reductions of 40–60%.

Considering intense and increasing fishing pressure throughout the range of this species it is suspected that the species has undergone a population reduction of >80% over the last three generation lengths (24 years), and it is assessed as Critically Endangered A2d (Jabado et al., 2021a). Information about this species appears to be limited due to its naturally low abundances in the northern part of its range (Mauritania to Côte d'Ivoire), and there is evidence of significant reductions in the southern part of its range (Gabon and Congo). While there are very few data available, it seems possible that this low productivity species (productivity estimated based on its congener *R. rhinobatos*) has declined to 15–20% from a historical baseline.

Rhinobatos irvinei

There are no species-specific time-series data available for this guitarfish species that can be used to estimate population reduction. This is due to a lack of species-specific reporting as well as taxonomic and identification issues (Jabado, 2018). There have been limited records of this species in the past decade. Although *R. irvinei* was never very abundant across the Eastern Central Atlantic and Southeast Atlantic, it has become increasingly rare, particularly in landings (Jabado et al., 2021b). In Mauritania, this species has not been recorded in regular fisheries monitoring surveys undertaken by the Institut Mauritanien de Recherches Océanographiques et de Pêches in the Parc National du Banc d'Arguin since 2009 and has likely disappeared from that area considering other species of guitarfish with the same catchability are still being landed (Jabado et al., 2021b). Between 2004 and 2011, this species was recorded in low numbers in Senegal, Guinea, and the Gambia during landing site surveys (Diop and Dossa, 2011) but more recently it has not been found despite regular monitoring (Jabado et al., 2021b). In the Gambia, this species was not recorded during landing site surveys conducted annually between 2010 and 2018 despite other species of guitarfish being present (Moore et al., 2019). In Senegal, 2–3 individuals were recorded during surveys at Ouakam in Dakar in 2011–2012, but records have become increasingly rare (Jabado et al., 2021b). Similarly, there have been no recent records of this species in landing site surveys in Guinea-Bissau (Bijagos Archipelago) (last records from 2008 to 2009), Côte d'Ivoire, Nigeria (Jabado et al., 2021b). It was also recorded in trawl surveys undertaken in Guinea from 1985–2012 (Camara et al., 2016). Cruise reports from trawl surveys in Togo indicate that this species was caught in 12% of tows with up to five individuals in each tow in 1984 (Lhomme, 1984).

In Ghana, *R. irvinei* was recorded in demersal fish surveys undertaken by the Japan International Cooperation Agency between 2000 and 2003. These trawl surveys operated at depths between 20 and 100 m and recorded 10 specimens in six tows (3.1% of sets) (Ishihara and Kimono, 2006). Landing and market sites surveys in Ghana in 2020 and 2021 reported that *R. irvinei* was the most landed species (71% of the relative catch), followed by *R. albomaculatus* (16%), and *R. rhinobatos* (6%) (Seidu et al., 2022a). Similarly, in western Ghana, 30 specimens of *R. irvinei*, representing 4% of the relative catch, and 11 specimens of *R. rhinobatos* representing 1.5% of the relative catch, were recorded in 2020 (Seidu et al., 2022b). In the trawl fisheries observer data from Gabon, this species along with other *Rhinopristiformes* represented 1% of rays captured while it was regularly recorded in artisanal fisheries landings operating in Mayumba using demersal-set gillnets (Jabado et al., 2021b). In the Congo, 65 individuals, representing less than 1% of landings were recorded during landing site surveys from January–December 2019 (Jabado et al., 2021b). In Angola, seven individuals were recorded during opportunistic landing site surveys in 2018 (Jabado et al., 2021b). In Cameroon, 26 individuals have been recorded over two years of landing site surveys representing 1.2% of all shark and ray records (Jabado et al., 2021b).

It is suspected that the species has undergone a population reduction of >80% over the past three generation lengths (30 years) due to high levels of exploitation. It is assessed as Critically Endangered A2d (Jabado et al.,

2021b). While there are very few data available, it seems possible that this low productivity species (productivity estimated based on its congener *R. rhinobatos*) has declined to 15–20% from a historical baseline.

Rhinobatos rhinobatos

There are limited species-specific time-series data available for this guitarfish species that can be used to estimate population reduction. This is due to a lack of species-specific reporting as well as taxonomic and identification issues (Jabado, 2018). This species has a relatively large range, but it is also under intense fishing pressure and suffers from severe habitat degradation. The species was prevalent in the 1970s and 1980s along the north African coast and eastern basin of the Mediterranean. By 1990, this species was extinct from the western and central regions of the Mediterranean (the coastal waters of Spain, France, and Italy), based on a combination of fishers' knowledge and data from the Mediterranean International Trawl Survey (MEDITS). MEDITS experimental trawl surveys (from the Alboran to the Aegean Sea) between 1994 and 1999 and trawl surveys in the Adriatic Sea between 1948 and 2005 failed to record any individuals (Relini and Piccinetti, 1991, Baino et al., 2001, Ferretti et al., 2013). This species is still caught in Tunisia and Egypt. It is not uncommon in Turkey, Lebanon, and Israel. This species is reported as one of the most common rays landed in Lebanon although fishing pressure has led to a loss of shark and ray diversity in these waters (Lteif, 2015; Lteif et al., 2016). It is fully protected in Israel and is not landed there.

In Mauritanian waters, species-specific population trend data show an annual rate of decrease of 4.6%, consistent with an estimated 85% reduction in population over three generation lengths. In a study investigating discards in Mauritanian shrimp fisheries between 2004 and 2006, this species was present in 14.9% of total catches and contributed 2.8% in weight (Goudswaard and Meissa, 2006). More recently, only 134 individuals have been recorded between 2006 and 2018 in regular fisheries monitoring surveys (Jabado et al., 2021c). Landings in north Africa indicate reductions in abundance with catches containing a large proportion of immature individuals. Between 2004 and 2011, this species was still a commonly recorded species in Mauritania, Senegal, the Gambia, Guinea-Bissau, Guinea, and Sierra Leone during landing site surveys (Diop and Dossa, 2011). Although it is still recorded in landings across these countries, data on guitarfishes (not species-specific and including *Glaucostegidae*) from Senegal indicate that landings peaked in 1997 at 4,218 t but decreased to an estimated 821 t by 2005 (Jabado et al., 2021c). This would represent a population reduction of 90% for *R. rhinobatos* over three generation lengths (42 years). In the Gambia, this species was not recorded in abundance during landing site surveys conducted annually between 2010 and 2018 although single individuals or small numbers were frequently recorded at landing and processing sites in all years (Moore et al., 2019). There have been no recent records of this species in landing site surveys in Guinea-Bissau (although restricted to the Bijagos Archipelago) or Côte d'Ivoire (Jabado et al., 2021c). In Nigeria, it is one of the more common guitarfishes landed although catch information is not available as they are aggregated with other guitarfish species (Jabado et al., 2021c). Landing and market sites surveys in Ghana in 2020 and 2021 reported that *R. irvinei* was the most landed species (71% of the relative catch), followed by *R. albomaculatus* (16%), and *R. rhinobatos* (6%) (Seidu et al., 2022a). In western Ghana, 11 specimens representing 1.5% of the relative catch of this species have been documented (Seidu et al., 2022b). In the Congo, there were no records of this species during landing site surveys from January–December 2019 although other guitarfish with similar catchability were recorded (Jabado et al., 2021c). In Cameroon, 15 individuals have been recorded over two years of landing site surveys representing 0.7% of all shark and ray records (Jabado et al., 2021c). In Angola, it was not recorded during opportunistic landing site surveys in 2018 (Jabado et al., 2021c).

This species' reduction in range, the ongoing high levels of exploitation in some areas, and available species-specific trend data, suggest a severe population reduction. Overall, fishing pressure is high and increasing across the West African range of the species and there has been a long history of fisheries overexploitation in the Mediterranean part of its range, which is ongoing. It is therefore inferred that the species has undergone a population reduction of >80% over the last three generations. It is assessed as *Critically Endangered A2bd* (Jabado et al., 2021c).

This species appears to be heavily depleted or rare in many areas of its range (parts of the Mediterranean, Mauritania, possibly Senegal) and relatively abundant in others (Lebanon, Nigeria). A simple average of available data suggests a global decline to 12% of baseline over the past three generations (42 years) for this low productivity species. If this decline was projected into the near future (10 years), it suggests a depletion to 7% of baseline.

Rhinobatos schlegelii

This species is subject to intense fishing pressure across its entire range. The species is now rare in Japan. It has virtually disappeared from the Republic of Korea over the past 20–25 years, and as such has been reduced by 75–96% over the past three generations (30 years). There used to be targeted guitarfish fisheries in the Republic of Korea but rhinobatids are now rarely seen (Rigby et al., 2021). In Taiwan POC, the abundance of *R. schlegelii* has decreased by 60–80% over the past 15–20 years at Penghu Island; previous landings of 50 or more individuals have now decreased to 10–20 individuals (Rigby et al., 2021). Landings in Taiwan POC decreased by 80% over 49 years from 1953–2001. The annual landings rose from 560 t in 1953 to a peak of 1,800 t in 1973,

then fell steadily to 114 t in 2001 (Rigby et al., 2021). This represents a reduction of 75–96% over the past three generations (30 years).

When these reductions are scaled to three generations (30 years) of *R. schlegelii*, they represent reductions of 63%, 88%, 40%, and 90% in Taiwan POC, Japan, China, and the Republic of Korea, respectively. The species is rare in Japan, has virtually disappeared from the Republic of Korea over the past 20–25 years, and has experienced reductions of 75–96% in part of Taiwan POC where mainly gravid females are landed. Overall, it is inferred that the species has undergone a population reduction of >80% over the past three generations (30 years).

Rhinobatos schlegelii is assessed as Critically Endangered A2bd (Rigby et al., 2021).

Taking a simple average of the known reductions for this species, these data suggest a global depletion to 30% (ranging from 10–60%) over the past three generations (30 years). If this decline was projected into the near future (10 years), it suggests a depletion to 22% of baseline for this low productivity species (ranging from 5–51%).

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

Fins

Fins from rhinobatids often enter international trade. For example, Haque and Spaet (2021) demonstrated that even the small fins derived from rhinobatids were exported from Bangladesh to China through Myanmar. While this study did not relate specifically to any of the focal species in the Proposal, it is feasible that similar trade patterns could be applicable.

A Ghanaian landing and market site survey conducted in 2020 and 2021 examined species composition among guitarfishes (Glaucostegidae and Rhinobatidae) (Seidu et al., 2022a). This study reported that *R. irvinei* was the most landed guitarfish species (71% of the relative catch), followed by *R. albomaculatus* (16%) and *R. rhinobatos* (6%). Fishers considered *R. irvinei* and *R. albomaculatus* to be “small” guitarfish, whereas *R. rhinobatos* was considered a “large” guitarfish (along with a species of Glaucostegidae). Small guitarfish were sold with fins attached, while the fins of large guitarfish were sold separate to the carcass (Seidu et al., 2022a). Fins are bought by local buyers or foreign buyers from Nigeria or Togo (Leeney and Quayson, 2022) as well as from the Gambia, Mali, and Senegal (Seidu et al., 2022a). Fins of these three species are dried and appear mostly to be destined to Asian markets through complex regional trade routes (e.g., from Cameroon to Nigeria and then exported to Asia (Jabado et al., 2021a, b, c)).

Pseudobatos horkelii is utilised bycatch across its range, and in some areas is targeted (e.g., Silveira et al., 2018). The fins of *R. schlegelii* are of lower value due to their small size and are consumed domestically in China, rather than traded internationally (Rigby et al., 2021). In Taiwan POC, the fins of larger *R. schlegelii* individuals are removed, although this may be for domestic consumption. Fins of this species and other small *Rhinobatos* spp. have not been recorded in trade surveys in Hong Kong SAR or Singapore (Rigby et al., 2021). However, a recent survey of retail vendors selling small fins in Hong Kong SAR in 2018 and 2019 identified fins of *Acroteriobatus variegatus* (Cardeñosa et al., 2020).

Meat

There is a growing understanding that meat from rhinobatids enters international trade, often within regions. For example, Haque and Spaet (2021) and Haque et al. (2021) demonstrated that meat from rhinobatids caught in Bangladesh was traded to Myanmar. While these studies did not specifically relate to any of the focal species in the Proposal, it is feasible that similar trade patterns could be applicable.

While little-species specific information is available, the meat of guitarfishes is consumed fresh across many coastal communities in the West African region as an important source of protein (Walker et al., 2005). It is also dried or dried and smoked and exported across West Africa to supply countries including Ghana, Guinea, Nigeria, Mali, and Burkina Faso. *R. albomaculatus*, *R. irvinei*, and *R. rhinobatos* are heavily utilised across their range for meat and fins. In Ghana, the meat of these three species is sold to local merchants and consumers (Seidu et al., 2022a).

The meat of *A. variegatus* is often sold fresh for human consumption at local markets and enters the international trade in dried form (Kyne et al., 2017). In India, demand for ray meat is rising and therefore prices are also increasing. The meat of *P. horkelii* is consumed or sold locally and can fetch a high price (Pollom et al., 2020).

Other products

The skin of *R. schlegelii* may be dried and traded internationally as a luxury leather product (Haque et al., 2018), although this species was not specifically identified in the study. The eggs of shark-like rays are sometimes dried and consumed locally while the heads may also be dried and used as either fish meal or fertiliser (Haque et al., 2018, Rigby et al., 2021). Based on its congener, the Ringed Guitarfish *R. hynnicephalus*, *R. schlegelii* individuals that are too small for human consumption are used for fish meal in Taiwan POC and likely also in mainland China.

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 “white” fins of shark-like rays (including *Rhinobatos* spp.) are considered the best quality fins for human consumption and are among the highest valued in the international shark fin trade (Moore, 2017). The shark fin category “Qun chi” is one of the highest value categories of fins, which comprises shark-like ray fins from *Rhinidae*, *Glaucostegidae*, and *Rhinobatidae* (Hau et al., 2018). Without specialised knowledge or advanced techniques, such as DNA barcoding, it is very challenging for customs officials to distinguish from which species the dorsal and caudal fins of species in the family *Rhinobatidae* have been taken, both in comparison with species in the family itself and with the wider group of related species of shark-like rays, particularly juvenile *Rhinidae* (wedgetfishes) and *Glaucostegidae* (giant guitarfishes). A random sample of shark-like batoid fins collected during market surveys in Hong Kong SAR and Guangzhou, China genetically confirmed the presence of *Rhinidae* and *Glaucostegidae*, but no *Rhinobatae* were detected (Hau et al., 2018). However, a more recent survey in Hong Kong SAR specifically of small, low-value fins in the shark fin retail markets, genetically confirmed the presence of *Rhinobatae* (Cardeñosa et al., 2020).

Rhinobatidae are one of the five families of shark-like rays. Like the *Rhinobatidae*, the *Rhinidae* and *Glaucostegidae* are also demersal shark-like rays, and their parts and derivatives, especially dorsal and caudal fins, are very similar to those of the *Rhinobatidae*, particularly in the case of juvenile specimens (which are also in demand in international trade). Whole specimens of *Pristidae* are readily identifiable as they are unique in possessing teeth protruding along the extended snout or rostrum. However, their dorsal and caudal fins are also very similar to those of the *Rhinobatidae*, again, particularly for juveniles. The entire group of shark-like rays includes species that are known to have the highest value of all shark and ray fins in trade, raising the risk of increased targeting of the family *Rhinobatidae* now that the other families found in trade are listed in CITES Appendix II.

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Species from *Glaucostegidae* and *Rhinobatidae* are sometimes caught together, and fishers reportedly consider there to be groups of “small guitarfish” (*R. albomaculatus* and *R. irvinei*) and “large guitarfish”, which includes a combination of a *Rhinobatidae* species (*R. rhinobatos*) and *Glaucostegidae* species (*Glaucostegus cemiculus*) (Seidu et al., 2022a). Therefore, (a) fishers may increase fishing pressure on smaller *Rhinobatidae* as they move away from targeting Appendix II listed *Glaucostegidae* and *Rhinidae*; and (b) fishers do not distinguish between some species of *Rhinobatidae* and *Glaucostegidae*.

Additional Information

Threats

The major threats for all species in this Proposal are unsustainable capture rates and habitat deterioration. Target fisheries for guitarfish currently exist in several countries, particularly in the Indo-west Pacific and in West Africa. Fins of many species enter the international fin trade destined for the Asian market (Cardeñosa et al., 2020; Seidu et al., 2022a). They are readily caught in a variety of fishing gear especially artisanal gillnets, trawls, line, trammel nets and seine nets, including as bycatch by demersal trawls and gillnets. Their occurrence along inshore areas of the continental shelf makes them an easy target.

Conservation, management and legislation

National

Some range States have adopted species-specific national instruments. Other range States may lack species-specific instruments but have other instruments in place that members of the family *Rhinobatidae* may indirectly benefit from (such as finning bans, fishing gear restrictions, and area and closed seasons).

International

Rhinobatos rhinobatos is listed in Appendix II of the Convention on Migratory Species of Wild Animals (CMS), with the Mediterranean Sea population listed in CMS Appendix I. CMS Parties are required strictly to protect Appendix I listed species, with prohibitions on “taking, hunting, fishing, capturing, harassing, deliberate killing, or attempting to engage in any such conduct.” Most range States for *R. rhinobatos* have legal measures in place that do strictly protect this species (Lawson and Fordham, 2018). *Rhinobatos rhinobatos* is also listed in Annex II of the Memorandum of Understanding on the Conservation of Migratory Sharks (Sharks-MOU), which is a non-binding daughter agreement under the CMS umbrella. In 2020, CMS also adopted two Concerted Actions aimed at facilitating international conservation of *R. rhinobatos*.

Rhinobatos rhinobatos is listed in Annex II of the Protocol Concerning Specially Protected Areas and Biological Diversity in the Mediterranean (SPA/BD Protocol) of the Barcelona Convention. Parties that have ratified this protocol are required to provide legal protection for the species. Members of the General Fisheries Commission for the Mediterranean (GFCM) are not permitted to retain *R. rhinobatos*.

Potential risk(s) of a listing

Fishers who target guitarfish may suffer economic losses if they are unable to harvest these fish anymore. This is especially concerning in West African countries where target fisheries for guitarfish are largely driven by the international export of fins while lower-value meat is sold and consumed domestically (Seidu et al., 2022a). Guitarfish meat has important food security benefits to local communities and is an important source of protein (Walker et al., 2005).

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Inclusion of the Zebra Pleco *Hypancistrus zebra* in Appendix I

Proponent: Brazil

Summary: *Hypancistrus zebra* is a highly distinctive small freshwater fish, endemic to shallow waters (<1–10 m) along a restricted stretch of the Xingu River, a tributary of the Amazon, in the state of Pará, Brazil. Its known area of occupancy has been estimated at just under 400 km². It was discovered in 1987: captive breeding started in the early 1990s and commercial captive breeding from the early 2000s. Observations in the 1990s indicated its population was decreasing, apparently due to overcollection for the aquarium trade and by 2004 the species was nationally considered to be vulnerable. Capture was prohibited in 2004 after which the population was believed to have recovered. The construction of the Belo Monte hydroelectric dam in 2016 affected the entire range of the species, which was assessed for the IUCN Red List in 2018 as Critically Endangered on the basis of a projected population decline of more than 80% over 10 years. Since then, observations suggest that the impact of the dam has been less severe than anticipated.

Hypancistrus zebra is a desirable fish with hobbyists and is widely available in the international pet trade with much of the supply coming from captive breeding, largely supplied by companies in Indonesia. Since 2000 large quantities of *H. zebra* have been produced with prices lower than those of illegally sourced wild individuals. However, illegal trade in *H. zebra* is evident. Over the last decade, over 4,100 individuals were reported as seized by the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA). Interviews with exporters indicate that hundreds to thousands of individuals per month, during peak season, have been smuggled out of Brazil into neighbouring countries. In 2017, the species was included in CITES Appendix III by Brazil. The majority of trade reported since then has been in captive-bred individuals from Indonesia (reporting nearly 30,000 exported specimens). In captivity individuals live for 10–15 years so that brood stock for captive breeding could have been legally acquired before Brazil's 2004 export ban.

Analysis: *Hypancistrus zebra* has a relatively restricted distribution in Brazil with a range of less than 400 km². Construction of the Belo Monte Dam in 2016 has divided the population into two subpopulations and resulted in a decrease in habitat quality, although it appears that this has had a less severe effect on the population than anticipated. The species is in demand and, despite a ban on exports since 2004, seizure data indicate that illegal exports take place, although there is little information on the scale of such exports or of the impact of collection on wild populations. The restricted range with declining habitat quality and continuing presence in trade indicate the species appears to meet the criteria for inclusion in Appendix I in Res. Conf. 9.24 (Rev CoP17). This would reflect Brazil's national regulations covering the species. Any additional benefits of an Appendix I listing would depend on an increase in enforcement efforts and there is a risk that negative regulatory impacts on the currently dominant supply from captive breeding could actually increase pressure on the wild population. An Appendix II listing with a zero-export quota for trade in wild specimens for commercial purposes would have the same regulatory effect. This would ensure that breeding operations would still be able to trade captive-bred individuals without their needing to be registered in CITES under Res. Conf 12.10 (Rev CoP15).

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

Hypancistrus zebra

The CITES standard reference for Actinopteri is the *Taxonomic Checklist of Fish species listed in the CITES Appendices and the Annexes of EC Regulation 338/97 (Elasmobranchii, Actinopteri, Coelacanthi, and Dipneusti, except the genus Hippocampus)*. Information extracted from Eschmeyer, W.N., and Fricke, R. (eds): *Catalogue of*

Fishes, an online reference, version update from 3 February 2015. See Annex 6 of CoP17 Doc. 81.1 at <https://cites.org/sites/default/files/eng/cop/17/WorkingDocs/E-CoP17-81-01-A6.pdf>.

While this species is listed in the Catalogue of Fishes database, it was listed in Appendix III in 2017 and therefore not included in the standard reference version from 2015.

Range

Brazil

IUCN Global Category

Critically Endangered, A3c (assessed 2018 ver. 3.1)

Biological criteria for inclusion in Appendix I

A) Small wild population

Hypancistrus zebra is endemic to Brazil. No population estimate exists for this species. In 2011 it was considered to be “not rare” (Roman, 2011), but was infrequent and low in abundance (Chamon, Cramer, Zuanon, Sousa, and Oliveira, quoted in ICMBio, 2022). The fecundity of *H. zebra* is relatively low, with 8–30 eggs per clutch and multiple broods per year.

B) Restricted area of distribution

Hypancistrus zebra is endemic to Brazil, restricted to the middle and lower part of the Xingu River basin in the region known as “Volta Grande” in the Brazilian state of Pará. It has an extent of occurrence currently calculated at 6,930 km², and an area of occupancy of 392 km² (ICMBio, 2022).

See next section for impacts of Belo Monte dam.

C) Decline in number of wild individuals

In 2004, the species was assessed as vulnerable due to the impacts generated mainly by its commercial harvest, and during the same year its harvest and trade was prohibited. In 2014, the species was recognised as critically endangered at the national level (MMA, 2014).

It was estimated that over a period of 10 years (2016–2026; more than three generations with an estimated generation time of 2.5 years), the species’ population would decline by more than 80% and therefore be at a high risk of extinction due to the building of the Belo Monte Dam. *On this basis the Red List assessment was Critically Endangered.*

However, according to Information provided to a FAO Expert Panel (2022), although the population of H. zebra has become fragmented by the construction of the hydroelectric dam, large numbers of H. zebra remain in approximately 66% of the original habitat. A large portion of this downstream population is found in protected areas, and a population of H. zebra remains in habitat upstream of the dam. Species of the Locariidae family have become some of the most popular exported ornamental fish from Brazil, contributing to this multi-billion dollar industry (Biondo and Burki, 2020).

While field observations revealed a decrease in the species’ wild population since 2015, it is argued that the decreasing trend is due to the local negative impact of Belo Monte Dam and also to overfishing to meet the increased consumer demand for illegally sourced fish (Sousa, 2021).

The species is not rare (Roman, 2011) but is currently infrequent and not very abundant. Occasional observations carried out between 1990 and 1997 indicated a strong population decline, apparently due to overfishing for the aquarium trade. In 2004, the species was considered as vulnerable due to this estimated population decrease.

With the prohibition of the capture of *H. zebra* in 2004 there has been an apparent recovery of the population and specimens can often be found in their natural environment. This is despite ongoing illegal fishing of specimens that are smuggled into Colombia from where they are regularly exported.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Legal trade

According to the CITES Trade Database, between 2017 and 2022, Indonesia was the only exporter, and reported almost 30,000 live specimens (unspecified and captive-born individuals), predominantly imported by Singapore (30%) and the USA (26%). During the same period, importers only reported trade of 10,092 live individuals and 100 fingerlings, from a range of sources. This combined trade was largely reported as imported by Germany (31%) and the UK (26%) and mainly exported by Indonesia (95%).

According to trade data from the US (LEMIS), the USA reported 4,897 individuals imported between 2017 and 2020, with 375 reported as having been harvested from the wild or ranched (Table 1). There may have been a reporting error as it is likely that these 375 individuals were captive-bred as they originated in Indonesia, not Brazil.

Table 1. Total cleared imports of live *Hypancistrus zebra* into the USA for commercial trade between the year 2017 and 2020, (LEMIS).

| Year | Source | Reported number | Country of origin and percentage (%) | Year total |
|-------|--------------|-----------------|--------------------------------------|------------|
| 2017 | captive-bred | 582 | Indonesia (>99) | 582 |
| | | | Taiwan POC (<1) | |
| 2018 | captive-bred | 590 | Indonesia (97) | 925 |
| | captive-born | 160 | Taiwan POC (3) | |
| | wild-sourced | 175 | Indonesia (100) | |
| 2019 | captive-bred | 705 | Indonesia (100) | 1,100 |
| | captive-born | 295 | | |
| | wild-sourced | 100 | | |
| 2020 | captive-bred | 1,955 | Indonesia (100) | 2,290 |
| | captive-born | 235 | Indonesia (57) | |
| | wild-sourced | 100 | Sri Lanka (43) | |
| Total | | | | 4,897 |

Illegal trade

In 2017, 132 live individuals were refused import into the USA (113 captive-bred individuals) from Indonesia were re-exported and 10 were abandoned; nine wild individuals reported from Colombia were seized). Additionally in 2019, three ranched individuals reported from Indonesia were seized (LEMIS, 2022).

The Supporting Statement states that despite the ban on capturing this species in Brazil, and the efforts of the authorities at the national level, illegal fishing continues and around 10,000 individuals/month are trafficked via Colombia (Sousa et al., 2021), however an FAO Expert Panel Report (2022) clarified that this figure actually refers to "projected" availability from illegal traders in Colombia as they were asked "How many fish could they supply?". It has been reported that *H. zebra* is consistently among the most common fish trafficked out of Brazil (Charity and Ferreira, 2020), contributing to the illegal global wildlife trade (Figure 1). The demand for trafficked *H. zebra* specifically is primarily driven by aquarists who want to keep wild-sourced specimens (Sousa, 2021).

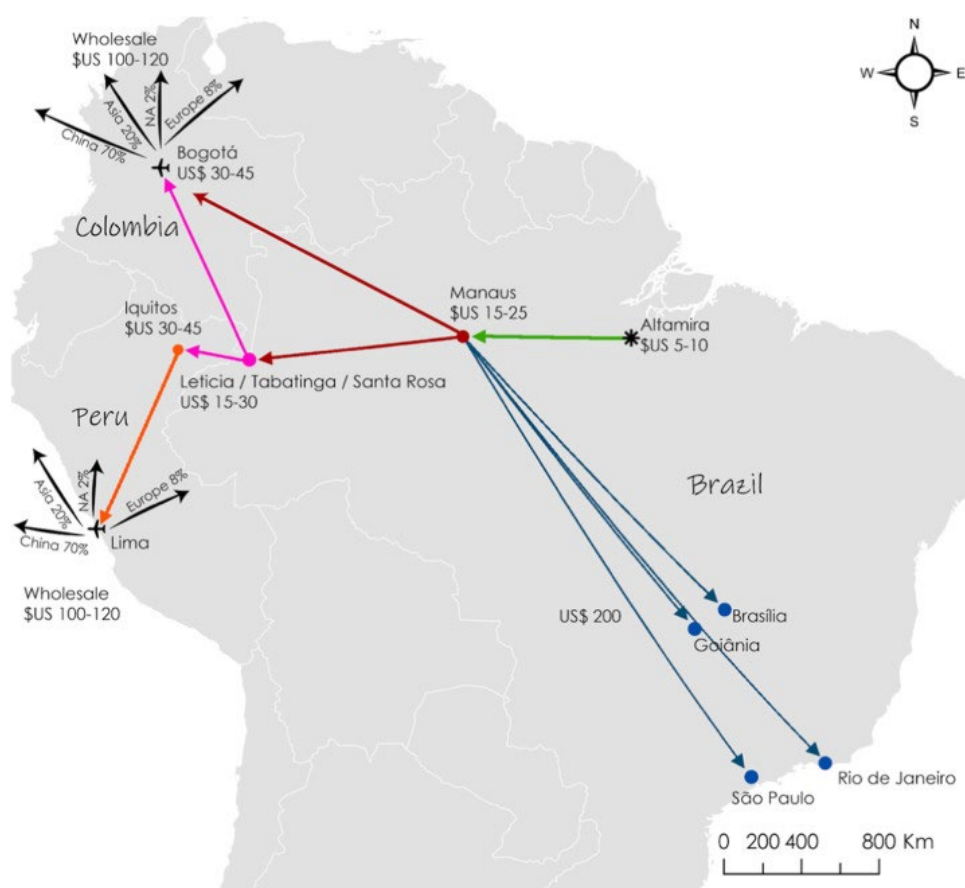


Figure 1. Common trafficking routes for illegally caught and exported *Hypancistrus zebra*. Some fish are sent to international destinations directly from Bogotá although from Iquitos they are first sent to Lima where they are loaded onto international flights. Internal shipments of *Hypancistrus zebra* to the national market are also shown but are a small fraction of the numbers trafficked internationally (Sousa, 2021).

The inclusion of *H. zebra* in CITES Appendix III led to an increase in demand and increased the price of trafficked fish from a wholesale price of USD45 (USD48, inflation corrected) to an average USD90–120 (USD95–127, inflation corrected) per individual (interview, E3–5; Sousa, 2021).

Seizures have been reported in Brazil:

In 2017, over 300 *H. zebra*, along with many other Amazonian fish, seized at Manaus airport, Brazil

In 2018, 199 individuals were seized from a river boat in Iquitos, Peru.

In 2019, 505 individuals were seized from a plane bound for Manaus.

Between 2006 and 2019, 4,115 individuals were reported seized by the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA) (Table 2).

Table 2. Number of specimens of *H. zebra* seized by IBAMA, 2006–2019.

| Year of seizure | Number of <i>H. zebra</i> seized |
|-----------------|----------------------------------|
| 2006 | 17 |
| 2007 | 2 |
| 2008 | 80 |
| 2009 | 67 |
| 2011 | 105 |
| 2012 | 1,478 |
| 2014 | 819 |
| 2015 | 740 |
| 2017 | 302 |
| 2019 | 505 |
| Total | 4,115 |

Additional information

Threats

This species was impacted by habitat loss and degradation due to the construction of the Belo Monte Dam in 2016. This affected the distribution and flow of water, impacting the species' ability to reproduce. In slow-flowing sections of the Xingu River the species is more vulnerable to capture for the illegal ornamental fish trade.

Conservation, management and legislation

Legal trade in the species occurs on a large-scale worldwide (although not in Brazil). Individuals legally traded are supposedly from captive breeding stock.

In Brazil, no export permits have been issued for individuals bred in captivity as the CITES Management Authority needs to ensure the legal acquisition of spawning stock and verify the operations with Brazilian standards. As there are no permits to export specimens, any export of the species for commercial purposes is illegal. Also, in Brazil, commercial farming (captive breeding) of endangered species is not allowed.

*Prior to 2004, *Hypancistrus zebra* was considered a species in the genus *Peckoltia* of which there were 21 recognised species that were permitted for export and were in international trade (LEMIS). The genus *Peckoltia* has gone through various reviews and its taxonomic status is not fully resolved (de Oliveira et al, 2012).*

*Following its inclusion in the Brazilian Red List of threatened species in 2004, exports of *H. zebra* were stopped (Figure 2). In 2004 a biodiversity assessment by the Brazilian environmental authorities categorised *H. zebra* as vulnerable. In 2008 the State of Pará listed *H. zebra* in the regional list of endangered species as vulnerable. In 2014 the Brazilian environmental authorities re-categorised *H. zebra* as critically endangered. In 2017 *H. zebra* was listed in CITES Appendix III, and Annex C of the European Union Wildlife Trade Regulations.*

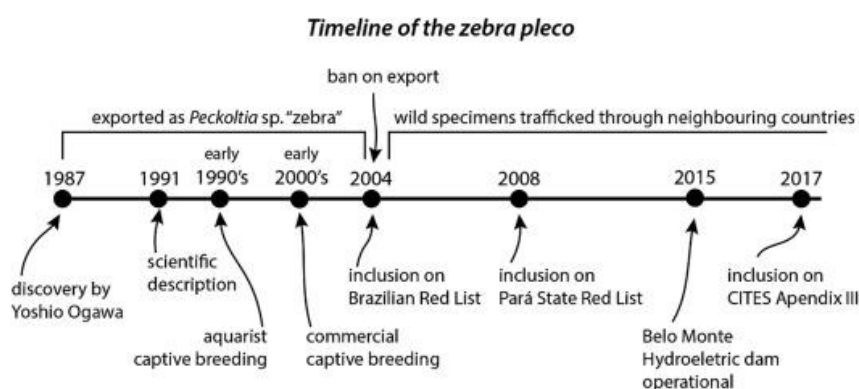


Figure 2. A timeline of the main events in the history of *Hypancistrus zebra* as an ornamental asset, including restrictions on capture and export and subsequent trafficking (reproduced with permission, Sousa et al., 2021).

Captive breeding

The species has been bred in captivity since the late 1990s in Europe and the USA. Since 2000, it has been extensively captive-bred in Indonesia, Ukraine, and the Czech Republic.

The characteristic of sustained sperm quality throughout the year greatly assists artificial reproduction and cryopreservation techniques and means a germplasm bank can be developed to support the conservation of this endangered Amazonian fish (Caldas and Godoy, 2019).

*There are currently only a few large-scale commercial *H. zebra* breeders worldwide. The largest one in Indonesia produces on average 10,000 young per year from approximately 5,300 breeding adults (interview, C1) (Sousa et al., 2021).*

Sousa et al. (2021) conducted international anonymous online surveys targeted at aquarium hobbyists and commercial breeders. Main findings include:

- Out of 317 valid interviews, the majority of respondents (47%; 149) reported purchasing or receiving their first *H. zebra* after 2004 when the Brazilian export ban had been implemented.
- Most respondents (57%; 181) also reported purchasing or receiving their most recent *H. zebra* after the species was listed in CITES Appendix III in 2017.
- Some 10% of the *H. zebra* individuals mentioned by the respondents were said to be wild-caught specimens. 35 respondents from Europe and North America, predominantly hobbyists intending to breed the species, indicated their entire colony of *H. zebra* (up to 30 individuals) was wild-caught. Conversely, 58% of respondents (184) indicated all individuals of this species were captive-bred.

- Many respondents (44%; 139) would only purchase captive-bred H. zebra or would prefer captive-bred ones if available (27%; 86).
- Only 30 respondents (9.5%) would prefer wild-caught specimens.

Sousa et al. (2021) found that 70% of the hobbyists keeping H. zebra intended to breed them. The 2,687 specimens reported as kept outside of Brazil is likely to be an underestimate of the total number. Sousa et al. (2021) received responses from approximately 53% of the total number registered H. zebra keepers on planetcatfish.com. Accounting for low responses received from Asia, and the volume of trafficked H. zebra (90%) regularly shipped there, a conservative estimate of 60,000–75,000 H. zebra are likely being kept in hobby aquariums worldwide, but this too likely represents an underestimate.

Sousa et al. (2021) reported that respondents' awareness of the geographic origin and legal constraints on harvesting and exporting H. zebra was high. Virtually all respondents (97%) stated they were aware that H. zebra is endemic to the Xingu River and most (88–89%) stated they were aware that H. zebra had been illegal to catch and export from Brazil since 2004. However, only 74% of respondents indicated they were aware that wild-caught specimens purchased from Peru, Colombia or other countries since 2004 had been smuggled out of Brazil and sold illegally. Most respondents (61%) indicated they were aware that many H. zebra die in transport when smuggled out of Brazil.

Potential risk(s) of a listing

Currently, H. zebra is commonly legally traded in a large number of countries outside Brazil. Hobbyists around the world can legally purchase the species from stocks gathered prior to the trade ban and breed and sell to local fish stores. This commonly takes place in Europe and the USA. Asian companies specialising in breeding ornamental fish (particularly in Indonesia) have already produced a consistent number of H. zebra since 2000 that cost less than wild specimens. These companies are responsible for the majority of legal H. zebra available today (Sousa, in litt., 2022).

Increased restrictions on harvesting of wild individuals of certain species can sometimes increase demand and price in the international market. Keepers of H. zebra and other ornamental fish have expressed their concern that if the species is listed in Appendix I, they will be "forbidden to keep breeding and selling the fish" which would likely increase the price and demand for wild-caught specimens. CEOs of Indonesian companies reported that after the Proposal to up-list H. zebra to Appendix I was published on the CITES website, they received the highest number of orders in decades and their stocks were depleted. A concern is that smugglers of the species have already received many orders and will increase their trafficking (Sousa, in litt., 2022).

Potential benefit(s) of listing for trade regulation

Under Res. Conf. 13.9 Parties who are involved in captive breeding and international trade will be encouraged to contribute to the in-situ conservation of wild populations in Brazil.

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Inclusion of Sea Cucumbers *Thelenota* spp. in Appendix II

Proponents: European Union, Seychelles, United States of America

Summary: Sea cucumbers or holothurians are a class (Holothuroidea) of marine echinoderm with a worldwide distribution, comprising around 1,700 species. In dried form, known as trepang or bêche-de-mer, they are traded and used for food and in traditional medicine, particularly in China.

Thelonota is a genus of three species (*T. ananas*, *T. anax*, and *T. rubralineata*) of large bodied, widely distributed Indo-Pacific sea cucumbers. *Thelenota ananas*, which may weigh up to 6 kg, is found on sandy sea floors at depths of 0–30 m off the coast of east Africa, the Indo-Pacific region and Australia. *Thelenota anax*, the largest known holothurian, often exceeding 7 kg in weight, is found on sandy portions of the ocean floor at depths of 5–25 m off the coast of eastern Africa, the Indo-Pacific region and Australia. *Thelenota rubralineata* is found on outer coral reef slopes at depths below 20 m in the west Pacific.

Holothurians are sedentary animals that are particularly vulnerable to overexploitation because they are large, typically occur in shallow waters, and do not require sophisticated fishing techniques. Specimens in shallow water can be collected through free-diving, but SCUBA or hookah equipment are needed to access those in deeper water. Most sea cucumbers are broadcast spawners, releasing their gametes into the water column. The success of reproduction depends directly on the density of adults to ensure high enough concentrations of sperms and eggs so that fertilisation takes place. Very little is known about generation length and recruitment of most species; it is thought that some *Thelenota* spp. can live for several decades in a natural undisturbed environment.

Consumption of sea cucumbers for food is common throughout many Asian countries. Sea cucumber fisheries have developed in many parts of the world, and most have declined as a result of overfishing. Aquaculture of sea cucumbers has increased in China since the 1990s, with production of over 160,000 t reported in 2021 and similar production expected in 2022. Publication of a paper in 2020 claiming that sulphated polysaccharides from sea cucumbers could inhibit the activity of the COVID-19 virus has reportedly led to explosive growth in sea cucumber consumption in parts of China.

No global population estimates are available for any of the species. There are indications of localised decreased populations throughout *T. ananas*'s range. Local declines have been reported for *T. ananas* in New Caledonia between the 1980s and 2013, in Tonga between 1984 and 2004, in the Red Sea between 2000–2016 (when none were observed in surveys), and for *T. anax* in Tonga between 1984 and 1996. In Fiji surveys of both species in 2012 and 2013 found markedly depleted population densities (one individual per ha or less, compared with expected densities of 10–20 per ha). A survey in Fiji between 2014–2015 recorded no *Thelonota* spp. *T. rubralineata* is rarely recorded and very few population density estimates are available.

Thelenota ananas was assessed as Endangered on the IUCN Red List in 2010; at that time populations were estimated to have declined by 80–90% across at least half of its range. Both *T. anax* and *T. rubralineata* were assessed as Data Deficient on the IUCN Red List (2010), with *T. anax* reported as uncommon.

Sea cucumbers are generally traded without taxonomic identification so that it is difficult to assess volumes of individual species in trade. There are no reliable estimates of the volume of *Thelenota* species in international trade although a low volume is inferred from the frequency of sale in wholesale and retail stores in China. One 2016 market study found *T. ananas* to be relatively widely available in shops (present in 22 out of 59 surveyed). This species is reported to have high nutritional value and is one of the highest value species in international trade (up to USD219 per kg (dried)). In the same survey *T. anax* was found at low frequency (3 out of 59). However, *T. anax* is apparently becoming increasingly popular as stocks of other species have been depleted. It was

sold in Chinese markets for USD31 per kg in 2016, a 70% price increase in five years. In Fiji, exporters indicated that *T. anax* was the most exported species by volume in 2014. In 2004, *Thelenota rubralineata* was identified along with 28 other species of sea cucumber as having commercial value in the Solomon Islands; the species is reported to be harvested there but there is no information on harvest levels.

Illicit trade in sea cucumbers is known to occur, however, there is little specific documentation of illegal trade in *Thelenota* species.

Numerous countries have instituted area closures in response to overexploitation of sea cucumbers. Egypt has employed no take zones (NTZ) and India employed a total ban on sea cucumber fishing, but these are not widely adopted strategies and there have been issues with implementation. Harvest of *Thelenota ananas* is prohibited in Mozambique. Limited-access fisheries in Australia 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 in force in Australia, Papua New Guinea, Fiji, and Tonga. However, large parts of the ranges of *Thelonota* spp. are not protected or regulated.

Analysis: *Thelenota* sea cucumbers are prone to overexploitation due to their limited mobility as adults, late sexual maturity, density-dependent reproduction, habitat preferences and low rates of recruitment. Moreover, they can be easily exploited because adults are large, often active during daytime, easy to detect and collect, and do not require sophisticated fishing or processing techniques.

Historic and recent localised declines have been observed in *T. ananas* which are consistent with the indicative guidelines for the inclusion in Appendix II of commercially exploited aquatic species suggested in the footnote to Annex 5 of Resolution 9.24 (Rev. CoP17). In some cases (parts of the Red Sea and Fiji) there are indications of local extirpations. It is likely that most accessible populations in its range are exploited given its high value and may have experienced similar declines, indicating that regulation of trade in the species is necessary to avoid it becoming eligible for inclusion in Appendix I in the near future (criterion 2a A).

Thelenota anax and *Thelenota rubralineata* are considered uncommon and have not appeared at all in some surveys of areas where they have been recorded in the past. Records of *T. rubralineata* are too sparse to allow conclusions to be drawn on likely changes of the population as a whole. Observations from some parts of the range indicate that *T. anax* may have experienced widespread population declines. As harvest for international trade is likely to be driving the majority of fishing for these species (across much of their ranges), and demand for this species may be increasing, it is likely that regulation of trade is required to ensure that harvest from the wild is not reducing their wild populations to a level at which their survival might be threatened by continued harvest or other influences (criterion 2a B). In summary, *T. ananas* and *T. anax* may meet the criteria for inclusion in Appendix II, but there is insufficient evidence to determine whether or not *T. rubralineata* meets the criteria. The three species can be distinguished from each other in wet and dried form.

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 bêche-de-mer in its dried, traded form) contains 1,743 species, while the genus *Thelenota* contains three species:

- *Thelenota ananas* Jaeger, 1833 – Pineapple Sea Cucumber, Prickly Redfish
- *Thelenota anax* Clark, 1921 – Giant Sea Cucumber, Amberfish
- *Thelenota rubralineata* Massin & Lane, 1991 – Red-lined Sea Cucumber

Range

Thelenota ananas is widely distributed throughout the Indo-Pacific: Hawaii. Australia, Bangladesh, Brunei Darussalam, Cambodia, China, Cocos (Keeling) Islands, Comoros, Cook Islands, Djibouti, Egypt, Eritrea, Fiji,

French Polynesia, Guam, India, Indonesia, Islamic Republic of Iran, Israel, Japan, Jordan, Kenya, Kiribati, Madagascar, Malaysia, Maldives, Marshall Islands, Mauritius, Mayotte, Mozambique, Myanmar, New Caledonia, Niue, Oman, Pakistan, Palau, Papua New Guinea, Philippines, Réunion, Samoa, Saudi Arabia, Seychelles, Singapore, Solomon Islands, Somalia, South Africa, Sri Lanka, Sudan, Taiwan POC, United Republic of Tanzania, Thailand, Tonga, Tuvalu, United States of America (Northern Marianas Islands), Vanuatu, Viet Nam and Yemen.



Distribution of *T. ananas*, IUCN (2010). The IUCN Red List of Threatened Species. Version 2020-2

Thelenota anax occurs throughout the Indo-Pacific: Australia, Bangladesh, Brunei Darussalam, Cambodia, China, Christmas Island, Comoros, Cook Islands, Djibouti, Federated States of Micronesia, Fiji, French Polynesia, Guam, India, Indonesia, Japan, Kenya, Kiribati, Madagascar, Malaysia, Maldives, Marshall Islands, Mauritius, Mayotte, Mozambique, Myanmar, Nauru, New Caledonia, Niue, Oman, Palau, Papua New Guinea, Philippines, Réunion, Samoa, Seychelles, Singapore, Solomon Islands, Somalia, South Africa, Sri Lanka, Taiwan POC, United Republic of Tanzania, Thailand, Tokelau, Tonga, Tuvalu, United States of America (American Samoa and Northern Marianas Islands), Vanuatu, Viet Nam, Wallis and Futuna and Yemen.



Distribution of *T. anax*, IUCN, 2010. The IUCN Red List of Threatened Species. Version 2020-2

Thelenota rubralineata is found only in the west Pacific; it has not been identified in the Indian Ocean (Kinch., 2005; Lane, 2008), unlike the other two *Thelenota* species. It is found throughout Australia, China, Cook Islands, Fiji*, Guam, Indonesia, Malaysia, Federated States of Micronesia, New Caledonia*, Palau, Papua New Guinea, Philippines, Solomon Islands, Taiwan POC, Timor-Leste, United States of America (Northern Mariana Islands) and Vanuatu*.

*these locations do not have any recent (post 2010) records of *T. rubralineata* sightings.



Distribution of *T. rubralineata*, IUCN, 2010. The IUCN Red List of Threatened Species. Version 2020-2

IUCN Global Category

Thelenota ananas – Endangered A2bd (assessed 2010, ver. 3.1)

Thelenota anax – Data Deficient (assessed 2010, ver. 3.1)

Thelenota rubralineata – Data Deficient (assessed 2010, ver. 3.1)

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) Annex 2a)

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

The volume of sea cucumber harvests began to increase in the late 1980s in South-East Asia and the South Pacific in response to increasing international demand. There are no *Thelenota* harvesting statistics, however overall, global catch and production (including aquaculture) of sea cucumber fisheries has increased 13- and 16-fold over the past two to three decades. Sea cucumbers are frequently traded without species-level identification and underreporting is common. *Sea cucumber fisheries have developed in many parts of the world, and most have declined as a result of overfishing (Conand, 2004; Uthicke, 2004).* Globally, reported exports are less than half of reported imports. In 2013, 10% of global sea cucumber fisheries were reported as depleted, 38% over-exploited, and 14% fully exploited. Prices have risen: two studies concluded that market prices across the species studied increased six- to twelve-fold over a decade.

China's top three e-commerce websites (Taobao.com, 1688.com, and JD.com) contain many advertisements for dried Thelenota species, but there is very little trade volume and no reviews from customers (Ling in litt., 2022).

There are no reliable estimates of the volume of Thelenota species in international trade although a low volume is inferred from the frequency of sale in wholesale and retail stores in China. In a market study in 2016, T. anax was found on only three occasions in 59 stores selling sea cucumbers in southern China (Purcell et al., 2018). Thelenota ananas was more commonly found on sale, at 22 out of the 59 stores visited. Additionally, USA import data between 2010 and 2020 from the U.S. Fish and Wildlife Service LEMIS database show that a total of 1,177 kg of T. ananas and 1,341 kg of T. anax were reported in cleared direct imports to the USA from other countries for commercial purposes. The majority of T. ananas was reported as coming from Mauritania (which may be a reporting error as the species is not found in that region) while the majority of T. anax came from Fiji. Furthermore, during this time period, 5,700 individual T. ananas were reported in cleared direct imports to the USA from Indonesia.

Thelenota ananas

According to IUCN's 2010 Red List assessment, *T. ananas* is Endangered having declined by 80–90% in at least 50% of its range (Philippines, Papua New Guinea, India, Indonesia, Madagascar) and is overexploited in the majority of its range. A 2004 CITES workshop on sea cucumbers expressed a “high level of concern” for the species, finding it “generally overfished”. Surveys throughout much of the species’ range have shown local declines:

- In French Polynesia, the species was reported to be present “but in low numbers” following the lifting of a sea cucumber fishing moratorium.
- In Tonga, deep-water occurrence declined from 48 in 1984 (one hour search period at 21 sites) to just four in 2004 (100 m transects, even after a fishing moratorium).
- In New Caledonia, *T. ananas* has declined more than 60% over the past 30 years, with 10–30 individuals/ha found in the 1980s, compared to just six individuals/ha found in the most preferred habitat in the most recent survey.
- In the Torres Strait, where there is a quota for this species, a stock survey conducted in 2019–2020 found an average density over four zones of 1.73 individuals per hectare (ind./ha) which was noted to be lower than earlier surveys (ranging from 1.81–2.41 ind./ha) but was generally consistent with densities of 1–2 ind./ha reported elsewhere.
- In Indonesia, it is heavily exploited, but there are few available statistics.
- In India, catch per unit effort and size of specimens dramatically declined.
- In the Red Sea, densities decreased dramatically from 48.1 ind./100 m² in 2000 to only 5.6 ind./100 m² in 2006, and the species was not recorded in 2016.
- In Eritrea, densities are estimated at 3.5 ind./ha (2013).
- In Madagascar and Seychelles, the species was reported to be becoming increasingly rare.

When dried/processed, *T. ananas* is around 20–25 cm long, elongated and brown-black in colour with spikes, and around 5% of its original live body weight (Ngaluafu et al., 2013). *Thelenota ananas* harvests included 40 t in Queensland (2019–2020) and 15.7 t in the Torres Strait in 2020; it was reported that 9.3% of sea cucumber exports from the Queensland fishery comprised *T. ananas*.

The sea cucumber fishery in the Maldives is recent as it started in the mid-1980, with the export of a trial shipment of around 30 kg of Thelenota ananas to Singapore (Conand et al., 2022).

Following a four-year period of intensive fishery in Mauritius (beginning late 2005), there was a net drop in Simpsons Diversity Index from 0.75 (in 1998) to 0.25 (in 2016). High to medium value species of sea cucumber, including *T. ananas*, had almost completely disappeared during the exploitation period and had not recovered by 2016 (Conand et al., 2022).

Thelenota ananas can be found for sale online as dead or dried specimens. One advertisement was selling 1 lb (0.45 kg) of wild-caught *T. ananas* from Kupang, East Nusa Tenggara, Indonesia for USD98.00 worldwide (picclick.com, 2022). Another advertisement was selling 1 kg of wild-caught *T. ananas* from Australia for ~USD230 worldwide, with more than 10 kg available (eBay, 2022).

Conand (2004) identified 42 species under population stress as a result of international trade to satisfy the *bêche-de-mer* market. Bruckner (2006) refined this list and identified species for international conservation and protection based on the following seven criteria: (i) commercial value; (ii) vulnerability to harvest and environmental fluctuations; (iii) geographic distribution; (iv) historical and present status of the different populations; (v) importance in world trade; (vi) concern raised by several countries; and (vii) knowledge of particular biological features (i.e., slow growth) or genetic information (i.e., isolated populations). During a CITES workshop, each species was categorised according to different levels of conservation concern (Bruckner, 2006). Five species were identified as being of high concern, one of which was *T. ananas*.

Thelenota anax

Naturally relatively uncommon, *T. anax* has been increasingly targeted in fisheries as other species decline. While few published surveys are available, data reveal likely declines in some locations.

- In French Polynesia, *T. anax* was found, but only 0.2–0.5 individuals per diving minute, across 23 islands and atolls.
- In Tonga, surveys in 1996 showed densities of 3.57 ± 1.55 ind./ha. Following a moratorium on fishing and export of sea cucumbers, a 2019 survey found 9.2 ± 2.71 ind./ha, though scientists recommend the moratorium continue to allow for further recovery.
- In Samoa, the species was not found in any surveys.
- In Fiji, annual harvests of *T. anax* increased ten-fold between 2003 and 2012, and it was found in densities of less than one individual per ha in broad-scale underwater surveys across nine regions in 2012 and 2013. This is markedly lower than the “reference density” of 20 ind./ha for healthy populations in the Pacific Islands (Pakoa et al., 2013)
- *Thelenota anax* was found “infrequently” in New Caledonia, with an average density of 14 ind./ha.
- In Papua New Guinea, densities decreased from one to 0.7 ind./ha between 1992 and 2006.
- *Thelenota anax* was found at only one site during a survey of 74 sites in Guam and was not considered a locally abundant species. In Malaysia, reports suggest a decrease in population and average size of the species.
- Surveys in the northwest and east coasts of Sri Lanka in 2008 and 2009 found an average of 26 ind./ha. *Thelenota anax* appeared in surveys conducted by Dalpathadu (2021) in eastern Sri Lanka. The study revealed that the stock of *T. anax* along with other species in the coastal waters might be moving towards extinction if the fishery prevails without proper management.

IUCN has concluded that *T. anax* is “potentially very vulnerable to overexploitation” and has recommended that “the exploitation of this species should be avoided”.

Thelenota ananas is a high value species and *T. anax* is a medium value species, although apparently becoming increasingly popular as stocks of other species have been depleted. Surveys of sea cucumber populations were conducted in a year-long study between 2014 and 2015 across eight sites in Fiji, and among the 23 species that were recorded, *T. ananas* and *T. anax* were included. However, no individuals of either species were recorded across the eight regions (Mangubhai et al., 2017). A regional density for the Pacific of 10 ind./ha had been estimated for *T. ananas* and 20 ind./ha for *T. anax* (Pakoa et al., 2013). The low richness of species at all sites suggests that local depletions of some species, particularly high-value species, have occurred.

Thelenota rubralineata

The species is rare and is only infrequently identified in surveys throughout its range. However, “due to its rarity and low population densities,” *T. rubralineata* is “extremely vulnerable to overexploitation”. The IUCN has similarly concluded that “given the rarity of this species and the fact that it is slow-growing and long-lived, it is likely very vulnerable to overfishing”. The Red List assessment considered that when the species is encountered, it is generally collected. This species occurs in many countries, and in the Solomon Islands, at least, it is exploited—to an as yet unqualified extent—as *bêche-de-mer* resource (Kinch, 2005) and considered “rare” (Ramohia, 2006).

- *Thelenota rubralineata* is rare in Guam, having been seen only once.

- The highest densities recorded were at a single reserve in the Solomon Islands with 45 ind./ha (2008).
- In Papua New Guinea (PNG), the species was found at less than 0.1 ind./ha. Other reports from PNG indicate that *T. rubralineata* is rarely observed during major sea cucumber stock assessments. For example, just four specimens of *T. rubralineata* were recorded during large-scale surveys (1,126 dives over an area of 256,000 km²) carried out throughout the province of Milne Bay, and only one individual was observed at Yap during a stock assessment conducted at depths of over 60 m.
- In Indonesia, wild population survey densities were recorded as one ind./220m², but generally the species is found in densities of less than one ind./ha.

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

Sea cucumbers are sedentary animals that are particularly vulnerable to overexploitation because they are large in size, easy to collect due to their shallow area of occurrence, and do not require sophisticated fishing techniques. Specimens in shallow water can be collected through free-diving, but SCUBA or hookah equipment are needed to access those in deeper water. Strong fishing pressure causes a decrease in species biomass density and populations are unable to replenish once they have fallen below critical mass. Most sea cucumbers are broadcast spawners, releasing their gametes into the water column. The success of reproduction depends directly on the density of adults to ensure high enough concentrations of sperms and eggs that they come into contact. 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. (Toral-Granda, 2006).

Overall, sea cucumbers are internationally traded without proper taxonomic identification (i.e. no genus or species), which makes it difficult to account for the true impact of international trade on wild Thelenota species (Toral-Granda, in litt., 2022).

According to FAO statistics, a total of 317,263 t of sea cucumber commodities have been imported by 68 different countries/territories between 1990 and 2016 (this includes commodities that 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 and 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 recorded in exports by 71 different countries/territories between 1990 and 2016. The top ten exporters 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 especially as these FAO figures do not distinguish between farmed and wild-caught specimens.

Table 2. Top ten exporters of sea cucumber (not species specific) commodities between 1990 and 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 |

Implementation issues/illegal trade

Illegal trade in sea cucumbers is known to occur generally, however there is little documentation on illegal trade in *Thelenota* spp. specifically.

Egypt

Harvesting of sea cucumbers in the Red Sea has been illegal since 2003, however, there is still evidence of illegal harvesting. In 2022, news reports from Cairo reported that the Environment Protection Police had apprehended suspects collecting and processing large numbers of sea cucumbers (Egypt Independent, 2022; Cairo Scene, 2022). In 2017, Hong Kong customs seized over 1,900 kg of dried sea cucumbers smuggled from Egypt with an estimated market value of USD850,000 (Hong Kong Customs, 2017).

India

A study by ICAR-Central Marine Fisheries Research Institute (ICAR-CMFRI) in India found that after the declaration of the ban on fishing sea cucumbers in 2001, 31% of respondents discontinued the activity, however, others are continuing the sea cucumber fishing/trade. Stock from India finds illegal market access through Sri Lanka, which remains a trade hub for South-East Asian countries and China (The News Minute, 2020). In 2020, India seized 22 dried sea cucumbers including *T. ananas* with an estimated value between USD25,000–27,000. In 2021, Dalian Customs detected a smuggling case worth USD14,810,000, of which the main product was dried sea cucumber (species not specified), imported from Japan (Gongbei Customs, 2021).

Solomon Islands

At Ontong Java Atoll, when the Area Council was strong, the sea cucumber fishery proved to be a sustainable and reliable income source. The collapse of the Area Council's authority in 1996 resulted in a lack of compliance with the former closed-season restrictions, which in turn led to sea cucumbers being harvested in greater quantities, leading to an eventual collapse of wild stocks (Bayliss-Smith et al., 2010; Christensen, 2011). More recently, the Solomon Islands Ministry of Fisheries and Marine Resources and Ontong Java community leaders, are in the process of a fishery management improvement programme at the atoll that will lead to the development of a community-based sea cucumber fishery management plan (Shedrawi et al., 2022).

Australia

Illegal, unreported and unregulated (IUU) sea cucumber harvesting is known to occur in the Australian Fishing Zone, but exact levels of IUU harvesting are unknown. Reported IUU incidents in Australian waters over the past decade include the seizure of 860 kg of sea cucumbers from 19 Indonesian fishing vessels in 2022; the interception of a Vietnamese fishing vessel with a "substantial amount" of sea cucumber on board; and the discovery of 6 t of sea cucumber on board two Vietnamese vessels in 2016. It was not specified whether these seizures comprised fresh or dried product.

In 2022, Australia experienced the highest rates of illegal fishing by Indonesian fisherman since 2005. The fishers were targeting sea cucumbers among other marine species. This is believed to be because of the pandemic and a lack of tourism income resulting in individuals turning to illegal fishing, particularly for high value species such as sea cucumbers (Barker et al., 2022).

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

B) 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.

A recent study showed that *T. anax* is particularly efficient at processing reef sediments, reworking 34 g dry weight of sediment per hour and contributing to bioturbation of half a kg of sediments per day. The sediment turnover by this species “surpasses those of other holothuroid deposit feeders,” making individuals of this species the most significant in this taxonomic class for sediment bioturbation on coral reefs and lagoons.

Additional information**Threats**

Overexploitation to satisfy demand for bêche-de-mer is the main threat to sea cucumber populations. Additionally, the mechanisation of fishing techniques, like the use of hookah or SCUBA diving, are rendering otherwise unexploitable populations suitable for fishing (i.e., inhabiting deeper zones) (Toral-Granda, 2006). Thelenota species, along with other Holothurians, are known to be used for bêche-de-mer in the Western Pacific (Table 3).

Bêche-de-mer is the product after gutting, cooking, salting and drying sea cucumbers. Demand is primarily in Asia. Sea cucumbers are one of the five primary luxury foods consumed at traditional festive dinners, along with bird nests, abalone, swim bladder, and shark fin. *According to statistics released by Chinese customs, a total of 1,488 tonnes of sea cucumbers were imported in 2015, the peak was in 2011 with 2,250 tonnes (Li ChenLin and Hu Wei, 2017). Additionally, according to the statistics of relevant departments, the annual import volume of dry sea cucumbers in normal years is 3,000 tonnes, worth nearly 10 billion yuan (USD1,479,000,000 at 2022 rates) (Sea Cucumber Industry Branch of China Fisheries Association, 2022).*

For sea cucumbers, extinction risk is primarily driven by high market value (*Thelenota* spp. are medium-high value species), as well as accessibility of harvest (often dependent on shallowness of their habitat) and how well-known the species is in the marketplace. Strong fishing pressure causes a decrease in species biomass density, and populations are unable to replenish once they have fallen below critical mass. As gonochorist broadcast spawners, sea cucumbers are particularly vulnerable to the Allee effect, which is characterised by failure of reproductive output associated with insufficient density of ripe individuals. Likewise, despite their commercial importance, little is known about their biology, ecology and population dynamics. This lack of scientific information constitutes an indirect threat, since it is essential for management plans and harvesting regimes.

Since a paper came out in 2020 claiming that sulfated polysaccharides from sea cucumbers could inhibit the activity of the COVID-19 virus, (Son et al., 2020), sea cucumber consumption in Shanghai, Wuhan, Xi'an, Chengdu, Changsha and other places has apparently shown explosive growth (Sea Cucumber Industry Branch of China Fisheries Association, 2022).

Thelenota ananas is one of highest value sea cucumber species in international trade, sold for as much as USD219 per kg. IUCN assessed the species as Endangered on the Red List in 2010. It is targeted throughout its range, and fishing pressure has dramatically increased in the past 25–50 years and is expected to continue, even as stocks are depleted. According to IUCN, this species is considered depleted in at least 50% of many parts of its range and is considered overexploited in the majority of its range.

Thelenota anax is the largest of the commercial sea cucumber species. It is of lower value than *T. ananas* but prices are increasing: in Chinese markets, *T. anax* sold for USD31 per kg on average in 2016, a 70% price increase from five years prior. In Sri Lanka and Fiji, *T. anax* is considered a “medium value” species. It was once considered non-commercial but has become an increasingly important species in the past 20 years as stocks of other species have been depleted. *Thelenota anax* is considered naturally uncommon. Rare species may be reproductively precarious and thus vulnerable to overexploitation. It is now being collected by skin diving or using diving gear, making the populations potentially very vulnerable to overexploitation. While more biological data needs to be collected on the species, the IUCN has concluded “the exploitation of this species should be avoided”.

Thelenota rubralineata is not one of the most commercially important species, likely due to its rarity, but the species is expected to become more popular after the depletion of other species of higher commercial importance and value. The species is commercially harvested in Papua New Guinea, the Solomon Islands and the Philippines. While little price information is available, it is sold within the Philippines for prices close to those

fetched for *T. ananas*, suggesting *T. rubralineata* is also a medium-to-high value species. *Thelenota rubralineata* is “extremely vulnerable to overexploitation” due to its rarity and low population densities.

Table 3. *Holothurians used for the production of bêche-de-mer in the Western Central Pacific region. The table includes sea cucumber species known to be utilised (shaded) and other possible species that are misidentified once processed (Toral-Granda, 2006)*

| Holothuriidae | | Stichopodidae |
|-----------------------------|--|---------------------------------|
| <i>Actinopyga albonigra</i> | <i>Holothuria arenicola</i> | <i>Stichopus chloronotus</i> |
| <i>A. caerulea</i> | <i>H. atra</i> | <i>S. godeffroyi</i> |
| <i>A. echinites</i> | <i>H. cinerascens</i> | <i>S. herrmanni</i> |
| <i>A. lecanora</i> | <i>H. coluber</i> | <i>S. horrens*</i> |
| <i>A. mauritiana</i> | <i>H. difficilis</i> | <i>S. monotuberculatus*</i> |
| <i>A. miliaris</i> | <i>H. edulis</i> | <i>S. naso*</i> |
| <i>A. palauensis</i> | <i>H. flavomaculata</i> | <i>S. pseudohorrens</i> |
| <i>A. spinea</i> | <i>H. fuscocinerea</i> | <i>S. vastus</i> |
| <i>Bohadschia anaes</i> | <i>H. fuscogilva</i> | <i>S. ocellatus</i> |
| <i>B. argus</i> | <i>H. fuscopunctata</i> | <i>Australostichopus mollis</i> |
| <i>B. bivitatta</i> | <i>H. grises</i> | <i>Thelenota ananas</i> |
| <i>B. geoffreyi</i> | <i>H. guamensis</i> | <i>T. anax</i> |
| <i>B. maculisparis</i> | <i>H. hilla</i> | <i>T. rubralineata</i> |
| <i>B. marmorata</i> | <i>H. impatiens</i> | |
| <i>B. similis</i> | <i>H. leucospilota</i> | |
| <i>B. subrubra</i> | <i>H. maculata</i> | |
| <i>B. tenuissima</i> | <i>H. pardalis</i> | |
| <i>B. vitiensis</i> | <i>H. pervicax</i> | |
| | <i>H. scabra</i> | |
| | <i>H. scabra</i> var. <i>versicolor*</i> | |
| | <i>H. verrucosa</i> | |
| | <i>H. whitmaei</i> | |
| | <i>Holothuria</i> sp. (Hongpai – Solomon Islands) | |
| | <i>Holothuria</i> sp. (Tulele – Solomon Islands) | |
| | <i>P. graeffei</i> | |

*taxonomic review in progress (in 2008)

Conservation, management and legislation

There are numerous domestic restrictions throughout the species' ranges that apply generally to sea cucumbers; however, the literature discusses few legal instruments that are specific to the *Thelenota* species. Numerous countries have instituted fishing moratoria and area closures in response to overexploitation:

- In India, sea cucumber fishing has been prohibited since 2001, when all sea cucumbers were listed in Annex I of India's Wildlife Protection Act.
- In French Polynesia, a 2012 moratorium banned sea cucumber fishing due to overexploitation; since the ban was lifted, management measures now restrict fishing in certain locations and require monitoring of exports.
- Tonga instituted a moratorium in 1996, which was lifted and then reinstituted due to continued overfishing.
- In Guam, no commercial export is allowed and local harvests are restricted to a daily limit; subsistence harvests are also permitted in American Samoa but commercial fishing and trade is prohibited.
- A moratorium on fishing of sea cucumbers has been in place in the Northern Mariana Islands since 1998.
- In the Philippines, there is a size limit for harvesting sea cucumbers.
- In the Federated States of Micronesia, a sea cucumber management plan sets weight restrictions and export volume restrictions.
- Madagascar, Fiji, Kenya, Seychelles, Solomon Islands, and Vanuatu set gear restrictions, such as prohibiting the use of SCUBA for harvesting sea cucumbers.
- Papua New Guinea has set a species-specific Total Allowable Catch (TAC), requires licensing, and prohibits the use of SCUBA.

- Seychelles also requires licensing and compliance with management measures.
- In Egypt, a ban on the harvest of sea cucumbers was implemented in 2003; areas where fishing was prohibited had a greater diversity and density of several commercial species of sea cucumbers.

In Australia, commercial harvest and export of sea cucumbers is regulated through the Environment Protection and Biodiversity Conservation (EPBC) Act. There are five commercial sea cucumber fisheries in the country, all of which have been assessed by the Australian Government as ecologically sustainable. In Queensland, Australia, the fishery has been managed by quotas since 1991 with a total allowable commercial catch (TACC). The TACC is allocated amongst individual transferable quota (ITQ) units for Black Teatfish *Holothuria nobilis*, White Teatfish *H. fuscogilva* and other sea cucumber species. The TACC is adjusted according to the decision rules in the harvest strategy each year. Size limits for the species are 50 cm in Queensland, 35 cm in the Torres Strait, and 30 cm in Northern Territory and Western Australia. Other harvest controls used by Australia alongside size limits, include temporal, spatial and gear restrictions. In 2021, the Australian Government identified concerns and uncertainties with regards to the Queensland fishery, and outlined ten new conditions, including independent surveys and stock assessments for *T. ananas*, alongside other sea cucumber species (*Stichopus hermanni*, *S. vastus*, and *Actinopyga palauensis*) as part of a 2021–2026 harvest strategy.

Thelenota ananas

Palau instituted an export moratorium for *T. ananas* in 1994. In the Maldives, the Bay of Bengal programme recommended a 4–5-year ban on take of *T. ananas*, and night fishing being discouraged. In Papua New Guinea, there is a minimum size limit of 25 cm live and 10 cm dry. In New Caledonia, there is also a legal minimum length (45 cm live and 20 cm dry), gear restrictions (collection using compressed air diving is prohibited) and there are no-take reserves. In Queensland, Australia, the fishery is regulated by limited access, combined ITQ, vessel and tender restrictions, the number of divers “to take” restrictions and rotational harvest arrangement. The TAC is 40 tonnes. *The minimum legal length in Torres Strait, Northern Territory and Western Australia is 30 cm live, whereas it is 50 cm live on the Great Barrier Reef. On the Great Barrier Reef, there is a TAC, limited entry and permits, and this species is subject to a rotational harvest strategy (Purcell et al., 2012). The harvesting of T. ananas in Mozambique is prohibited under Mozambican Legislation (REP-MAR- General Regulation on Maritime Fisheries, Decree 89/2020) (Conand et al., 2022).*

Thelenota anax

The TAC for *T. anax* in Queensland Australia is 50 t and in Papua New Guinea, fishery regulations include a minimum landing size of 20 cm live and 10 cm dry. *On the Great Barrier Reef (Australia), there is a minimum legal length of 50 cm live, limited entry of fishers, who are licensed, a combined TAC (with other species), no-take marine reserves, and this species is subject to a rotational harvest strategy (Purcell et al., 2012).* However, there are no other known species-specific conservation measures for these species at this time. Both species may be present in some marine protected areas within their range.

Thelenota rubralineata

It seems there are few regulations for the exploitation of this species within its distributional range (Purcell et al., 2012).

Captive breeding

Some countries have developed new methods to produce bêche-de-mer. These measures have gained importance since methods of reproduction and rearing of larvae and juveniles have been developed for some commercial species. *At present, a highly developed sea cucumber breeding industry has been established in China, concentrated in Shandong, Liaoning, Fujian, and Hebei provinces. According to statistics from the Sea Cucumber Industry Branch of the China Fisheries Association, national sea cucumber production was about 160,000 tonnes in 2021, and will remain at the same volume in 2022 (Sea Cucumber Industry Branch of China Fisheries Association, 2022).*

Recent advances in aquaculture provide the potential to reduce greatly the time required to re-establish depleted stocks through the release of cultured juveniles or to restore spawning biomass to a more productive level (restocking) and may also increase productivity of operational fisheries by overcoming recruitment limitation by maintaining populations closer to the carrying capacity of the habitat (stock enhancement) (Dance et al., 2003; Bell and Nash, 2004).

However, the access to technology for producing juveniles alone is not enough to proceed with restocking and stock enhancement programmes. For populations where stock assessments reveal that the spawning biomass is at chronically low levels, it can be assumed that by the release of hatchery-reared juveniles, the number of spawners could be restored (Bell and Nash, 2004). However, for such restocking to be effective (i.e., to achieve the desired outcome of restoring fishery productivity), the juveniles that are released into the wild and the remnant wild sea cucumbers need to be protected from fishing. Moreover, the protected area should be closed for long enough time to allow the progeny of the released sea cucumbers to replenish the population to the desired spawning

biomass. If this is the case, managers need to verify whether a moratorium on fishing for the remnant wild population alone would be sufficient to achieve replenishment (Toral-Granda, 2006).

Implementation challenges (including similar species)

In the Solomon Islands, *T. rubralineata* is called 'lemonfish' and because of its body shape, there is likely some confusion in species identification for export; it has a similar appearance to *Stichopus* spp. and they are probably marketed under the same trade name (FAO, 2022).

Potential risk(s) of a listing

There is a general feeling that because the sea cucumber fishery supports the livelihood of many rural communities in Fiji, managing the fishery should be viewed in isolation from any other development issues in order to gain a better understanding of and appreciation and concern for what is required for effective management. Therefore, the lack of or slow progress in developing other livelihood options in rural areas should not be used as a reason for taking no management action in regard to sea cucumbers (Pakoa et al., 2013).

However, one 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 species collected to maintain their income. These results suggest that CITES Appendix II trade regulation will not have a significant impact on livelihoods.

Bruckner (2006) identified problems with including sea cucumbers in the CITES Appendices, which are: (i) burdens on both range and importing countries to comply with permitting requirements, undertake research, eventually develop new legislation, train stakeholders in new trade provisions and specimen identification, promulgate regulatory measures to comply with CITES provisions, etc.; (ii) short-term socio-economic impacts by the reduction of fishery income, tax revenue and disruption of local fishing communities; and (iii) potential diminished co-operation in market surveys and IUU trade investigations.

Potential benefit(s) of listing for trade regulation

A CITES listing could benefit sea cucumber species by (i) curtailing illegal international trade and associated harvest; (ii) enhancing sustainable exploitation of wild stocks that are harvested for international trade through the making of non-detriment findings (for Appendix II listings); (iii) contributing to a reduction of overharvests; (iv) increasing awareness amongst stakeholders and decision-makers for the need to manage and conserve sea cucumbers; (v) providing better opportunities for technical assistance, targeted research and capacity building; (vi) helping to address FAO's concerns about overexploitation; (vii) assisting conservation and management for the long-term socio-economic benefit of fishers (Appendix II and III listings); (viii) promoting regulatory measures to comply with CITES provisions; (ix) installing standardised and comprehensive trade reporting amongst countries, and centralised data gathering and analysis on trade; and, (x) encouraging the development of Regional Fisheries Management Organizations (RFMOs) for sea cucumbers (Bruckner 2006). A CITES listing can also offer Parties international tools to avoid sea cucumber species becoming seriously endangered due to international trade, and promote the development of management strategies both at the national and regional levels (Toral-Granda, 2006).

Other comments

There is a need to address major gaps in knowledge of species' population sizes and trends. Such work will underpin future sustainable management and conservation plans to help preserve diversity and avoid extinctions. In a recent assessment (November 2021), the CITES Scientific Authority of Australia placed conditions on the fishery with respect to *H. fuscogilva* as well several IUCN-listed species (e.g. *Thelenota ananas*, *Stichopus herrmanni*) and heavily targeted species (*Actinopyga spinea*) (DAWE 2021). These conditions must be addressed over the next three years, with fishery reassessment scheduled for 2024 when the export of Black Teatfish can again be considered (Byrne et al., 2022).

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).

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Amend annotations #1, #4, #14 and the annotation to species of Orchidaceae listed in Appendix I

Proponent: Canada

Summary: The proposed amendments to annotations #1, #4, #14 and the annotation for taxa of Orchidaceae listed in Appendix I, are a result of extensive deliberation by the Standing Committee's Working Group on Annotations. They were endorsed by the Standing Committee at its Seventy-fourth meeting. The main change proposed is to remove the phrase “, in solid or liquid media” from annotations providing exemptions for trade in plant seedlings and tissue cultures obtained *in vitro* on the basis that this reflects the evolution in techniques for propagating and transporting such specimens since the adoption of the current text while maintaining the original intent of the exemption.

All other changes proposed are grammatical but, because they relate to a substantive annotation, a proposal to the CoP is required under Res. Conf. 11.21 (Rev. CoP18) on *Use of annotations in Appendices I and II*.

Analysis: The amendments propose removal of the phrase “, in solid or liquid media” from annotations providing exemptions for trade in plant seedlings and tissue cultures obtained *in vitro*; these were supported by consensus by the Standing Committee at its 74th meeting in 2022 (SC74). They are in line with current propagation techniques and should have no impact on the conservation of species in the wild. Submission of this proposal to the CoP is in line with Res. Conf. 11.21 (Rev. CoP18) on *Use of annotations in Appendices I and II*.

Summary of Information and Discussion

On the basis of discussions of the Standing Committee's Working Group on Annotations, the proposed amendments were endorsed by the Standing Committee at SC74. Amendments are proposed to #1, #4, #14 and the parenthetical annotation to Orchidaceae listed in Appendix I, and they are all along the same lines and aim to reflect the evolution of techniques for propagating and transporting seedling or tissue cultures obtained *in vitro*, while they appear to maintain the original intent of the exemption of specimens of the species produced this way and listed under one of these annotations.

The phrase “, in solid or liquid media” as a qualifier for exempted seedlings and tissue cultures in CITES annotations for plant listings reflected the *in vitro* process at the time the current text was proposed. The presence or absence of such media has no impact on the appropriateness of a trade exemption or conservation of a species. Current experience suggests that when tissue cultures and seedlings are transported internationally the accompanying solid or liquid media may be depleted or not easily observed. As a result, tissue cultures and seedlings obtained *in vitro* and transported in sterile containers may appear not to contain solid or liquid media and may be deemed not to meet the requirements of the exclusion in the annotations. It is proposed to remove “, in solid or liquid media” from all these annotations, in order to facilitate understanding and implementation.

The amendment proposed will harmonise all occurrences of the phrase “, in solid or liquid media” in the CITES Appendices and in hash-series annotations, to read as follows:

Parenthetical annotation to Orchidaceae listed in Appendix I

(For all of the following Appendix-I species, seedling or tissue cultures obtained *in vitro*, ~~in solid or liquid media~~, and transported in sterile containers are not subject to the provisions of the Convention only if the specimens meet the definition of ‘artificially propagated’ agreed by the Conference of the Parties)

Annotation #1: All parts and derivatives, except:

...b) seedling or tissue cultures obtained *in vitro*, ~~in solid or liquid media~~, transported in sterile containers; ...

Annotation #4: All parts and derivatives, except:

... b) seedling or tissue cultures obtained *in vitro*, ~~in solid or liquid media~~, transported in sterile containers; ...

Annotation #14 All parts and derivatives except:

... b) seedling or tissue cultures obtained *in vitro*, ~~in solid or liquid media~~, transported in sterile containers; ...

The amendment proposed also includes grammatical corrections in Annotation #14 with the replacement of a comma with a semicolon (highlighted for clarity) in paragraph f to read

f) finished products packaged and ready for retail trade; this exemption does not apply to wood chips, beads, prayer beads and carvings.

And grammatical correction to the French of Annotation #14 Toutes les parties et tous les produits, sauf:

... f) les produits fini conditionnés et prêts pour la vente au détail; cette dérogation ne s'applique pas aux copeaux ~~en~~ de bois, aux perles, aux grains de chapelets et aux gravures....

These amendments ensure that annotations are in line with current propagation techniques.

Other comments

The working group also noted in SC74 Doc. 81 (2022) that regarding exceptions for specimens obtained *in vitro* transported in sterile containers, the seedlings or tissue cultures taken out of sterile containers no longer qualify for the exception included in the annotation, regardless of whether the seedlings or tissue cultures are used for artificial propagation, or for another purpose, including the production of extracts. According to Res. Conf. 11.11 (Rev. CoP18), paragraph 11, the country in which the specimens are removed from sterile containers is then to be treated as the country of origin.

The working group had extensive input from the Standing Committee and Plants Committee members, Parties and Non-Party observers (see SC74 Doc. 81) and this proposal is made in accordance with the consensus recommendation made by the Standing Committee at its 74th meeting.

References

SC74 Doc. 81 (2022). *Annotations: Report of the Working Group*.

<https://cites.org/sites/default/files/eng/com/sc/74/E-SC74-81.pdf>

Inclusion of Trumpet Trees *Handroanthus* spp., *Roseodendron* spp., and *Tabebuia* spp. in Appendix II with annotation #17

Proponents: Colombia, European Union, Panama

Summary: *Handroanthus*, *Tabebuia* and *Roseodendron* are genera of Bignoniaceae distributed from southern USA to Argentina and Chile, including the Caribbean. There are currently 113 recognised tree species across the three genera (35 in *Handroanthus*, 76 in *Tabebuia* and two 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. While some are widely distributed, over half of the species are endemic to one range State or restricted to islands.

Most species within these genera produce a very hard, heavy, and durable wood that is used locally in the construction of houses, bridges, flooring, decking, and handicrafts. Internationally, where it is marketed under the single common name of Ipê, it is one of the preferred timbers for decking. Distinguishing between species and genera is reportedly difficult even at the microscopic level, and there are no identification guides covering all species. The bark is also traded internationally for medicinal and aromatic purposes. Little is known about this trade or whether harvesting bark is detrimental to the species in the wild.

Ipê is one of the most valuable timbers 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 as ornamentals.

The proponents seek to include the genera *Handroanthus*, *Tabebuia* and *Roseodendron* in Appendix II with annotation #17 (Logs, sawn wood, veneer sheets, plywood and transformed wood). *Handroanthus serratifolius* and *H. impetiginosus* are proposed under Annex 2a Criterion B, with the remaining species proposed under Annex 2b Criterion A based on timber being traded under the same trade name (Ipê) and under genus names, as well as due to identification, nomenclature, and taxonomic uncertainties. The three genera were previously proposed for Appendix II listing at CoP18 by Brazil (CoP18 Prop. 49), but the proposal was withdrawn before consideration.

Although no estimates for the total global trade in Ipê exist, Brazil is identified as a major exporter. Brazil reported a total of 255,723 m³ of Ipê in trade in 2010–2016. Brazil reportedly exports Ipê to 60 countries, the principal importers being the USA and Europe. Trade from Brazil accounted for 93% of Ipê sawn wood and around 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 (as well as low volumes of seizures reported in Colombia, Mexico, and Venezuela), and there are concerns over inappropriate management measures including overestimation of sustainable offtakes, but it is unclear what proportion of illegally harvested timber enters international trade. In the forests of northeastern Brazil, *H. impetiginosus* and *H. serratifolius* have shown severe population declines, with no evidence of long-term population recovery.

- ***Handroanthus impetiginosus*:** *Handroanthus impetiginosus* was assessed as Near Threatened on the IUCN Red List in 2020, noting that its populations have declined considerably as a result of unsustainable exploitation for the international timber trade, with declines projected to continue. The species is currently categorised as near threatened in Brazil in 2019 (but was not included in the most recent assessment), threatened in Mexico and endangered in Peru. Populations of *H. impetiginosus* in parts of Brazil have reportedly suffered significant declines through overexploitation. Brazil reported exports of 1,644 m³ of

H. impetiginosus in 2010–2016. Exports of *H. impetiginosus* are also reported by Venezuela (20,491 m³ from 2007–2017).

- ***Handroanthus serratifolius*:** *Handroanthus serratifolius* was categorised as globally Endangered on the IUCN Red List in 2020 on the basis that it is threatened by international trade and is predicted to experience a significant population decline in the future.

Of Ipê exports reported by Brazil from 2010–2016, 70% (~180,000 m³) 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.

In the period 2010–2016, Brazilian exports of *H. serratifolius* peaked in 2012, with 36,000 m³ reported. Brazil reported exports of 220,000 m³ in 2017. 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 m³/ha. Colombia reported harvests of 1,727 m³ in 2019–2021.

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 capitatus*, *Handroanthus chrysanthus*, and *Handroanthus incanus*:** These three species are all assessed as Vulnerable. Volumes in trade included reports of *H. capitatus*: ~3000 m³ by Brazil 2010–2016 and ~13,000 m³ by Suriname 2017–2019; *H. chrysanthus*: 50 m³ by Brazil and ~24,000 m³ by Colombia 2019–2021; and *H. incanus*: ~2000 m³ by Brazil. There were additionally reports of seizures in Mexico and Colombia of illegally obtained *H. chrysanthus*.

Ipê is of increasing economic importance; it is mainly exported as decking, sawn wood and flooring for use in furniture and construction. The main importers are the EU and the USA. Over 525 million kg (or ~470,000 m³) of Ipê timber products were exported from Brazil, Paraguay, Peru, and the Plurinational State of Bolivia (henceforth Bolivia) between 2017–2021. The majority of Ipê is exported from Brazil, which accounted for virtually all trade (96% based on volume). At least 13 species of *Handroanthus* were reportedly exported from Brazil during 2010–2016, however some trade is reported at the genus level, in many cases under synonymous names in the genus *Tabebuia*. The low natural density and low growth rate of *H. serratifolius*, as well as *H. impetiginosus*, typical of most of the other species within the three genera, combined with high demand for international trade, habitat loss and degradation, has had a negative impact on populations.

Although the known main international trade is in two species (*H. serratifolius* and *H. impetiginosus*), the trade name Ipê widely refers to any species of the three genera, as timber trade data are generally not recorded at the species level. Other species reported in international trade include *H. capitatus* (6,000 m³ sawn wood exported from Suriname from 2011–2015), *H. heptaphyllus* (5,000 m³ sawn wood exported from Guyana from 2011–2015), *Roseodendron donnell-smithii* (183 m³ sawn wood and 510 roundwood pieces exported from Mexico from 2010–2012), and *Tabebuia rosea* (exports from Venezuela totaling ~27,000 m³ from 2007–2017). 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.

According to IUCN Red List assessments, Ipê (species of *Handroanthus*, *Tabebuia*, and *Roseodendron*) is increasingly being exploited unsustainably. Distinguishing distinct species of the three genera based on timber is macroscopically and microscopically not possible. Evidence

suggests that current levels of exploitation of *H. serratifolius*, *H. impetiginosus* and potentially numerous other Ipê-producing species for which trade data cannot be clearly assigned to a specific taxon, may lead to serious population decreases.

Analysis: *Handroanthus*, *Tabebuia*, and *Roseodendron* are genera of New World trees comprising over 100 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 in 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 several countries from Mexico to Argentina. *H. capitatus*, *H. chrysanthus*, and *H. incanus* are additionally reported in trade at lower levels according to available records.

While global data on trade are not available, Brazil appears to be the main exporter of Ipê, the majority of which is *H. serratifolius*, with 19 other species also reported in trade. *Tabebuia* spp. reported at the genus level was the second most reported in trade by Brazil, and *T. rosea* was reportedly exploited in high levels by Colombia and Venezuela. There are also reports of illegal Ipê harvest and trade taking place in Brazil, as well as seizures of timber reported by Brazil, Colombia, Mexico, and Venezuela. 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. *Handroanthus capitatus*, *H. chrysanthus*, *H. impetiginosus*, *H. incanus* and *H. serratifolius* have all been assessed as threatened (*H. serratifolius* as Endangered) with significant projected future population declines.

On this basis, *Handroanthus capitatus*, *H. chrysanthus*, *H. impetiginosus*, *H. incanus* and *H. serratifolius* appear to meet criterion B for inclusion in Appendix II in Annex 2a of Res. Conf. 9.24 (Rev. CoP17), and this may also be the case for many other species in the three genera for which distinct trade records are unavailable. The remaining species in all three genera meet the criteria for inclusion in Annex 2b as lookalikes, based on the reported identification difficulties, taxonomic and nomenclatural uncertainties, as well as being in trade under the same trade name.

Dipteryx alata and *D. odorata* are said to be commonly confused with *Handroanthus* spp., *Tabebuia* spp., and *Roseodendron* spp. which are proposed for listing in Proposal 48; these would also meet the lookalike criteria in Annex 2bA of Res. Conf. 9.24 (Rev. CoP17) were the current Proposal to be accepted.

Annotation

The majority of trade appears to be as sawn wood (HS code 4407) and wood flooring and decking (under HS code 4409) and clearly within the CITES definition of transformed wood. Bark has also been reported in trade, but no information exists on trade volumes and impact on the species. Therefore annotation #17 to include “Logs, sawn wood, veneer sheets, plywood and transformed wood” would seem to cover the main items first in trade from range States. If international bark trade is found to be detrimental to the species in the wild in the future, it may be appropriate to list the species with a new annotation, #17 with the inclusion of bark.

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 the World Checklist of Vascular Plants (WCV, 2022), the proposed Standard Reference, 35 species of *Handroanthus*, 76 of *Tabebuia*, and 2 of *Roseodendron* are currently accepted. 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.

Range

Handroanthus, *Tabebuia*, and *Roseodendron* are distributed in the Americas, from southern USA to Argentina and Chile, including the Caribbean. Of the 113 species, 67 are endemic to one range State with 29 endemic to Cuba, 20

to Brazil, 9 to the Dominican Republic, 3 to Haiti, 3 to Puerto Rico (USA), 1 to Colombia, and 1 to Guyana (WCVP, 2022). The Proposal lists the distribution and range States for each species in Annex 1.

For the species proposed under Annex 2a B:

- *Handroanthus serratifolius*: Bolivia, Brazil, Colombia, Ecuador, French Guiana, Guyana, Peru, Suriname, Trinidad and Tobago, and Venezuela. It is reported to be introduced in Cuba, Puerto Rico (USA), and the Venezuelan Antilles. It has an extent of occurrence of 12.4 million km².
- *Handroanthus impetiginosus*: Argentina, Bolivia, Brazil, Colombia, Costa Rica, El Salvador, French Guiana, Guatemala, Honduras, Mexico, Nicaragua, Panamá, Paraguay, Peru, Suriname, and Venezuela and has an extent of occurrence of over 24 million km².

Other species that are in trade and may meet criterion B of Annex 2a:

- *Handroanthus capitatus*: Bolivia, Brazil, Colombia, French Guiana, Guyana, Peru, Suriname, Trinidad and Tobago, and Venezuela and has an extent of occurrence of over 5.1 million km² (Hills, 2021c).
- *Handroanthus chrysanthus*: Belize, Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Guyana, Honduras, Mexico, Nicaragua, Panama, Peru, Trinidad and Tobago, and Venezuela and has an extent of occurrence of 11.1 million km², and an area of occupancy of 2,332 km² (Hills, 2021d).
- *Handroanthus incana*: Bolivia, Brazil, Colombia, Ecuador, and Peru (Hills, 2021e).

IUCN Global Category

Of the 113 accepted species, 54 species have been assessed by the IUCN Red List, with a further seven accepted for publication in the December 2022 update of the IUCN Red List (version 2022-2), and draft assessments for seven species not previously assessed (five as Endangered, one as Vulnerable, and one as Near Threatened; IUCN in litt., 2022). Of the 113 species, 24 are globally assessed as critically endangered, endangered, or threatened according to national assessments (for endemic species) or the IUCN Red List. The assessed species and those accepted for v.2022-2 of the IUCN Red List are listed in Table 1.

Table 1. Assessed species. Species claimed by the proponents to merit listing under Annex 2a are in bold. National assessments are included for endemic species that have not been assessed on the IUCN Red List. Asterisk (*) indicates IUCN Assessments accepted for publication in the December 2022 update of the IUCN Red List (version 2022-2). IUCN Red List population trends: –: stable, ▼: declining and ?: unknown. National threatened status sources: Brazil – Government of Brazil, 2022; Cuba – González Torres et al., 2016.

| IUCN Global Category and national threat category | Species and assessment information |
|---|---|
| Critically Endangered | <i>Handroanthus grandiflorus</i> , B1ab(i,ii,iii)+2ab(i,ii,iii); D (Assessed 2018, version 3.1) ▼ <i>Tabebuia buchii</i> , (Possibly Extinct) B2ab(ii,iii) (Assessed 2020, version 3.1) ▼ <i>Tabebuia multinervis</i> , B2ab(ii,iii) (Assessed 2020, version 3.1) ▼ |
| Nationally critically endangered | <i>Tabebuia sauallei</i> (Cuba, 2016) |
| Endangered | <i>Handroanthus serratifolius</i> , A3cd+4cd (Assessed 2020, version 3.1) ▼ <i>Tabebuia conferta</i> , B1ab(iii) (Assessed 2020, version 3.1) ▼ <i>Tabebuia elongata</i> , B1+2c (Assessed 1998, version 2.3) |
| Nationally endangered | <i>Handroanthus arianeae</i> (Brazil, 2022) <i>Handroanthus botelhensis</i> (Brazil, 2022) <i>Handroanthus cristatus</i> (Brazil, 2022) <i>Handroanthus riodocensis</i> (Brazil, 2022) <i>Handroanthus spongiosus</i> (Brazil, 2022) <i>Tabebuia bibracteolata</i> (Cuba, 2022) <i>Tabebuia clementis</i> (Cuba, 2016) <i>Tabebuia bullata</i> (Dominican Republic) <i>Tabebuia maxonii</i> (Dominican Republic) <i>Tabebuia ophiolitica</i> (Dominican Republic) <i>Tabebuia paniculata</i> (Dominican Republic) <i>Tabebuia ricardii</i> (Dominican Republic) <i>Tabebuia vinosa</i> (Dominican Republic) <i>Tabebuia zanonii</i> (Dominican Republic) |
| Nationally threatened | <i>Tabebuia caleticana</i> (Cuba, 2016) <i>Tabebuia inaequalis</i> (Cuba, 2016) <i>Tabebuia pinetorum</i> (Cuba, 2016) |

| IUCN Global Category and national threat category | Species and assessment information |
|---|---|
| Near Threatened | <i>Tabebuia platyantha</i> , Lower Risk/Near Threatened (Assessed 1998, version 2.3) <i>Handroanthus barbatus</i> , A3cd+4cd (Assessed 2020, version 3.1) ▼ <i>Handroanthus impetiginosus</i> , A3cd+4cd (Assessed 2020, version 3.1) ▼ <i>Handroanthus lapacho</i> , B2ab(iii) (Assessed 2020, version 3.1) ▼ <i>Handroanthus subtilis</i> , B2ab(iii) (Assessed 2020, version 3.1) ▼ <i>Handroanthus uleanus</i> , B2ab(iii) (Assessed 2020, version 3.1) ? <i>Roseodendron chryseum</i> , B2b(iii) (Assessed 2020, version 3.1) ? <i>Tabebuia insignis</i> , A3cd+4cd (Assessed 2020, version 3.1) ▼ <i>Tabebuia orinocensis</i> , B1ab(iii)+2ab(iii) (Assessed 2020, version 3.1) ? <i>Tabebuia pilosa</i> , B2ab(iii) (Assessed 2020, version 3.1) ? <i>Tabebuia roseoalba</i> , A2cd+3cd (Assessed 2020, version 3.1) ▼ |
| Nationally near threatened | <i>Tabebuia brooksiana</i> (Cuba, 2016) <i>Tabebuia elegans</i> (Cuba, 2016) |
| Vulnerable | <i>Handroanthus capitatus</i> , A3cd+4cd (Assessed 2020, version 3.1) ▼ <i>Handroanthus chrysanthus</i> , A3cd+4cd (Assessed 2020, version 3.1) ▼ <i>Handroanthus coralibe</i> , B2ab(iii) (Assessed 2020, version 3.1) ▼ <i>Handroanthus incanus</i> , A3cd+4cd (Assessed 2020, version 3.1) ▼ <i>Tabebuia anafensis</i> , B1+2c (Assessed 1998, version 2.3) <i>Tabebuia arimaoensis</i> , D2 (Assessed 1998, version 2.3) (nationally threatened, Cuba endemic, 2016) <i>Tabebuia bibracteolata</i> , D2 (Assessed 1998, version 2.3) <i>Tabebuia densifolia</i> , B2ab(ii,iii,v) (Assessed 2022, version 3.1)* ▼ <i>Tabebuia dubia</i> , B1+2c (Assessed 1998, version 2.3) (nationally less concern, Cuba endemic, 2016) <i>Tabebuia furfuracea</i> , B1+2c (Assessed 1998, version 2.3) <i>Tabebuia hypoleuca</i> , B1+2c (Assessed 1998, version 2.3) (nationally less concern, Cuba endemic, 2016) <i>Tabebuia jackiana</i> , D2 (Assessed 1998, version 2.3) <i>Tabebuia oligolepis</i> , B1+2c (Assessed 1998, version 2.3) <i>Tabebuia palustris</i> , A2c (Assessed 2008, version 3.1) ▼ <i>Tabebuia polyantha</i> , B2ab(i,ii,iii,iv,v) (Assessed 2022, version 3.1)* ▼ <i>Tabebuia polymorpha</i> , B1+2c (Assessed 1998, version 2.3) (nationally threatened, Cuba endemic, 2016) <i>Tabebuia shaferi</i> , B1+2c (Assessed 1998, version 2.3) (nationally less concern, Cuba endemic, 2016) <i>Tabebuia striata</i> , C2a (Assessed 1998, version 2.3) |
| Nationally vulnerable | <i>Tabebuia cassinoides</i> (Brazil, 2022) |
| Data Deficient | <i>Handroanthus diamantinensis</i> , (Assessed 2018, version 3.1) ? <i>Handroanthus parviflorus</i> , (Assessed 2018, version 3.1) ? |
| Nationally data deficient | <i>Tabebuia glaucescens</i> (Cuba, 2016) |

| IUCN Global Category and national threat category | Species and assessment information |
|---|---|
| Least Concern | <i>Handroanthus albus</i> , (Assessed 2018, version 3.1) – <i>Handroanthus bureavii</i> , (Assessed 2018, version 3.1) ? <i>Handroanthus guayacan</i> , (Assessed 2020, version 3.1) ? <i>Handroanthus heptaphyllus</i> , (Assessed 2018, version 3.1) – <i>Handroanthus obscurus</i> , (Assessed 2020, version 3.1) ? <i>Handroanthus pedicellatus</i> , (Assessed 2018, version 3.1) ? <i>Handroanthus umbellatus</i> , (Assessed 2018, version 3.1) ? <i>Handroanthus vellosi</i> , (Assessed 2018, version 3.1) ? <i>Roseodendron donnell-smithii</i> , (Assessed 2020, version 3.1) ? <i>Tabebuia acrophylla</i> , (Assessed 2022, version 3.1)* – <i>Tabebuia bahamensis</i> , (Assessed 2019, version 3.1) – <i>Tabebuia berteroi</i> , (Assessed 2022, version 3.1)* – <i>Tabebuia calcicola</i> , (Assessed 2022, version 3.1)* – <i>Tabebuia elliptica</i> , (Assessed 2018, version 3.1) ? <i>Tabebuia fluvialis</i> , (Assessed 2020, version 3.1) ? <i>Tabebuia heterophylla</i> , (Assessed 2019, version 3.1) ? <i>Tabebuia lepidota</i> , (Assessed 2021, version 3.1) ▼ <i>Tabebuia microphylla</i> , (Assessed 2022, version 3.1)* – <i>Tabebuia nodosa</i> , (Assessed 2020, version 3.1) ▼ <i>Tabebuia obovata</i> , (Assessed 2022, version 3.1)* – <i>Tabebuia pallida</i> , (Assessed 2020, version 3.1) ? <i>Tabebuia reticulata</i> , (Assessed 2018, version 3.1) ? <i>Tabebuia rosea</i> , (Assessed 2018, version 3.1) – <i>Tabebuia impetiginosa</i> , (Assessed 1998, version 2.3) |
| Nationally less concern | <i>Handroanthus catarinensis</i> (Brazil) <i>Tabebuia lepidophylla</i> (Cuba, 2016) <i>Tabebuia leptoneura</i> (Cuba, 2016) <i>Tabebuia linearis</i> (Cuba, 2016) <i>Tabebuia obtusifolia</i> (Brazil) <i>Tabebuia simplicifolia</i> (Cuba, 2016) <i>Tabebuia trachycarpa</i> (Cuba, 2016) |

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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

There is limited information available on the population sizes of Ipê species. However, *Handroanthus* species are reported to occur at low natural densities, and show a high level of endemism, with 59% of the species endemic to one range State, and a further 13 restricted to islands in the Caribbean (WCVP, 2022). Of the 61 species assessed on the IUCN Red List, approximately a third have declining population trends (Table 1).

The Supporting Statement notes that in some commercial timber concessions in the Brazilian Amazon, pre-harvest inventories do not identify the different Ipê-species at the species level. This is problematic as it prevents species-specific population estimates and could lead to over-proportional harvesting of rare but highly exploited species.

Population status

Handroanthus impetiginosus

The species is assessed as Near Threatened in the IUCN Red List with a decreasing population trend as a result of unsustainable exploitation for the international timber trade, and a population decline of at least 25% over the next 100 years is expected, and the assessment noted that monitoring of trade was essential (Hills, 2021a). The species was described as widespread and sometimes abundant and has an extent of occurrence of over 24 million km² and a minimum area of occupancy of 3,338 km² (Hills, 2021a). The species is nationally threatened in Mexico and endangered in Peru. The species is nationally assessed as near threatened in Brazil (Flora do Brasil, 2020).

An inventory conducted in the 1970s by the Brazilian Ministry of Mines and Energy recorded all trees >30 cm dbh (Diameter Breast Height) in 2,364 plots of 1 ha throughout the Legal Amazon, finding an average density of 0.11 trees/ha for *H. impetiginosus*. In a 2004 study, a fragment of deciduous seasonal forest in the northeast of the Brazilian state of Goiás, an absolute density of 18.27/ha of *H. impetiginosus* with dbh ≥5 cm. Forest inventory data from Bolivia showed mean densities of 2.5 trees/ha for *H. impetiginosus*.

Handroanthus serratifolius

The species is categorised as globally Endangered on the IUCN Red List with a decreasing population trend on the basis that it is threatened by international trade and is predicted to experience a population decline of at least 50% over the next 100 years. Whilst the population size of *H. serratifolius* is considered to be large given its wide distribution, it is declining. The species is nationally threatened in Peru and Venezuela. In Venezuela, the species has reportedly been depleted in its natural populations as a consequence of the demand for wood for the production of handicrafts in the states of Lara and Falcón.

For *H. serratifolius*, industrial development of the Amazon is considered a major threat, with Brazil having lost 20% of its forest cover between 2002–2019. In Brazil, exploitation was reported to have resulted in significant declines, with no evidence of long-term recovery.

Forest inventory data from 2000 in Bolivia indicated that the mean density of *H. serratifolius* was 0.45 trees/ha with few young trees recorded relative to large, very old mature trees (Hills, 2021b). An inventory conducted in the 1970s by the Brazilian Ministry of Mines and Energy recorded all trees >30 cm dbh in 2,364 plots of 1 ha throughout the Legal Amazon, finding an average density of 0.32 trees/ha for *H. serratifolius*. A study published in 2008 noted that forest inventories carried out in the state of Pará, one of the main timber-producing states in the Brazilian Amazon, recorded *H. serratifolius* densities between 0.2–0.4 trees/ha with dbh ≥50 cm.

Handroanthus capitatus, *H. chrysanthus*, and *H. incanus*

These species are categorised as globally Vulnerable in the IUCN Red List with a decreasing population trend projected to experience a population decline of at least 30% over the next 100 years (Hills, 2021c; Hills, 2021d; Hills, 2021e). According to the assessments, these species are considered threatened by international trade, and threats to the species are predicted to increase.

Other species

Of the remaining species in the three genera, 16 have been recorded in the international timber trade. These include *H. albus*, *H. barbatus*, *H. billbergii*, *H. chrysotrichus*, *H. guayacan*, *H. heptaphyllus*, *H. incanus*, *H. ochraceus*, *H. speciosus*, *H. umbellatus*, *T. angustata*, *T. aurea*, *T. cassinoides*, *T. heterophylla*, *T. rosea*, and *T. roseoalba*. Of these, three have been assessed by the IUCN as Least Concern with a stable population trend, three as Least Concern with an unknown population trend, two as Near Threatened with decreasing population trends, one as Vulnerable with a decreasing population trend, and seven have not been assessed on the IUCN Red List. *H. chrysotrichus* is nationally assessed as critically endangered in Bolivia, and *T. cassinoides* is nationally assessed as vulnerable in Brazil, where it is also endemic.

In a recent study on the risk of extinction of 80 socio-economically viable Neotropical tree species, *H. pulcherrimus* was assessed as one of seven species that deserve special attention because it is highly threatened throughout its distribution in South America. *H. grandifloras* (a Brazilian endemic with only one sub-population) is assessed as critically endangered, alongside two other *Handroanthus* species (Table 1).

Table 2. National threatened status, and endemism in range States according to Plants of the World Online (POWO), 2022. Bold indicates the species proposed under Annex 2a criterion B.

| Range State | Nationally threatened species |
|-------------|---|
| Argentina | Threatened: <i>H. lapacho</i> |
| Bolivia | Critically endangered: <i>H. chrysotrichus</i> , <i>H. lapacho</i> |
| Brazil | Critically endangered: <i>H. grandifloras</i> (Endemic) Endangered: <i>H. arianae</i> (Endemic), <i>H. botelhensis</i> (Endemic), <i>H. cristatus</i> (Endemic), <i>H. riodocensis</i> (Endemic), <i>H. spongiosus</i> (Endemic), <i>T. nodosa</i> Near threatened: <i>H. impetiginosus</i> Vulnerable: <i>T. cassinoides</i> (Endemic) Less concern: <i>H. albus</i> , <i>H. catarinensis</i> (Endemic), <i>H. heptaphyllus</i> , <i>T. obtusifolia</i> (Endemic) |
| Colombia | Threatened: <i>T. palustris</i> , <i>T. striata</i> |
| Costa Rica | Threatened: <i>H. guayacan</i> , <i>T. palustris</i> |
| Cuba | Critically endangered: <i>T. leptopoda</i> , <i>T. pergracilis</i> , <i>T. pulverulenta</i> (Endemic), <i>T. sauvallei</i> (Endemic) Endangered: <i>T. bibracteolata</i> (Endemic), <i>T. clementis</i> (Endemic) Threatened: <i>T. arimaoensis</i> (Endemic), <i>T. bahamensis</i> , <i>T. berteroi</i> , <i>T. billbergii</i> , <i>T. caleticana</i> (Endemic), <i>T. densifolia</i> , <i>T. elongata</i> , <i>T. inaequalpès</i> (Endemic), <i>T. obovate</i> , <i>T. pinetorum</i> (Endemic), <i>T. polymorpha</i> (Endemic) Vulnerable: <i>T. jackiana</i> (Endemic) Near threatened: <i>T. brooksiana</i> (Endemic), <i>T. elegans</i> (Endemic) Less concern: <i>T. angustata</i> , <i>T. dubia</i> (Endemic), <i>T. heterophylla</i> , <i>T. hypoleuca</i> (Endemic), <i>T. lepidophylla</i> (Endemic), <i>T. lepidota</i> , <i>T. leptoneura</i> (Endemic), |

| Range State | Nationally threatened species |
|--------------------|---|
| | <i>T. linearis</i> (Endemic), <i>T. microphylla</i> , <i>T. moensis</i> , <i>T. myrtifolia</i> , <i>T. shaferi</i> (Endemic), <i>T. simplicifolia</i> (Endemic), <i>T. trachycarpa</i> (Endemic) Data deficient: <i>T. calcicole</i> , <i>T. crispiflora</i> , <i>T. glaucescens</i> (Endemic) |
| Dominican Republic | Endangered: <i>T. bullata</i> (Endemic), <i>T. crispiflora</i> , <i>T. dominguensis</i> , <i>T. maxonii</i> (Endemic), <i>T. obovata</i> , <i>T. ophiolitica</i> (Endemic), <i>T. paniculata</i> (Endemic), <i>T. ricardii</i> (Endemic), <i>T. vinosa</i> (Endemic), <i>T. zanonii</i> (Endemic) |
| Haiti | Endangered: <i>T. conferta</i> (Endemic) |
| Jamaica | Almost threatened: <i>T. platyantha</i> (Endemic) |
| Mexico | Threatened: <i>H. chrysanthus</i> , <i>H. impetiginosus</i> |
| Panama | Threatened: <i>T. palustris</i> , <i>T. striata</i> |
| Peru | Endangered: <i>H. impetiginosus</i> Threatened: <i>H. serratifolius</i> |
| Venezuela | Threatened: <i>H. serratifolius</i> |

International trade

In the international market the very hard, heavy, and durable wood is highly sought after and is used for flooring, decks, exterior woods, veneer, and other turned objects, crafts, and posts. In several countries in the Americas *Handroanthus* spp. are considered multipurpose trees that provide both, high value timber and non-timber forest products (NTFP). The wood that most *Handroanthus*, *Tabebuia*, and *Roseodendron* species produce is used locally in the construction of houses and bridges, pavements, decks, exterior woods and handicrafts. The bark is used for medicinal purposes and is also traded internationally.

In the period 2017–2021, roughly 77% of Ipê exports were classified as flooring or decking, with 19% exported as sawn wood. In the same period, around 4% of the Ipê tracked was exported under other product categories or HS codes including joinery, particleboard, veneer, and plywood. Paraguay was the only country where more Ipê was exported as sawn wood than flooring. In 2018, it was estimated that the value of Ipê processed into flooring or decking can reach USD2,500/m³ in international markets. Over 525 million kg (or 469,613 m³) of Ipê timber products were exported from Bolivia, Brazil, Paraguay, and Peru between 2017–2021 (Table 3). The majority of Ipê was exported from Brazil, which accounted for 96% of the trade (based on volume).

Table 3. Volumes (m³) of Ipê timber in international trade by reported origin country of four range States, 2017–2021 (Norman & Zunino, 2022).

| Origin | Volume (m ³) | | | | | |
|----------|--------------------------|--------|---------|---------|--------|---------|
| | 2017 | 2018 | 2019 | 2020 | 2021 | Total |
| Brazil | 77,846 | 94,258 | 99,323 | 101,310 | 76,643 | 449,381 |
| Bolivia | 3,052 | 2,473 | 1,885 | – | – | 7,410 |
| Paraguay | 1,955 | 1,663 | 1,231 | 922 | 1,306 | 7,077 |
| Peru | 974 | 1,183 | 1,157 | 923 | 1,508 | 5,744 |
| Total | 83,827 | 99,578 | 103,596 | 103,156 | 70,213 | 469,613 |

No total estimates for the global trade in Ipê exist, but wood is exported to at least 60 countries. The main species in trade appear to be *H. serratifolius* and *H. impetiginosus*. Nevertheless, at least 13 other species were reported as exported from range States. *The main range States for which data are available are outlined below.*

Brazil

According to the Supporting Statement (SS), all Brazilian Ipê wood comes from natural forests, as there are no plantations in the country. The SS notes that there was a 500% increase from 1998–2004 in Ipê timber exports from the Brazilian Amazon. In the period 2010–2016, Brazil reported exports of 255,723 m³, and over 70% of these exports were reported to be *H. serratifolius* (Table 4). *H. serratifolius* was primarily exported as decking (75%), followed by sawn wood (17%), and flooring (4%). Exports between 2010–2016 peaked in 2012 with over 35,000 m³ exported.

Table 4. Timber volume of Ipê reported as exports by Brazil from 2010–2016 (CoP18 Prop. 49, 2019). Bold indicates the species proposed under Annex 2a criterion B. National threat based on national red list from 2022 (Government of Brazil, 2022), “(end.)” denotes endemic.

| Reported species | Equivalent species according to proposed CITES Standard Reference | IUCN Red List | National status (2022) | Volume (m ³) |
|------------------------|---|---------------|------------------------|--------------------------|
| <i>T. serratifolia</i> | <i>H. serratifolius</i> | EN (2020) | | 180,110 |
| <i>Tabebuia</i> spp. | | | | 61,227 |
| <i>T. capitata</i> | <i>H. capitatus</i> | VU (2020) | | 2,887 |
| <i>T. incana</i> | <i>H. incanus</i> | VU (2020) | | 2,243 |

| | | | | |
|-------------------------------|---|------------------|-----------|--------------|
| <i>T. impetiginosa</i> | <i>H. impetiginosus</i> | NT (2020) | | 1,644 |
| <i>T. ochracea</i> | <i>H. ochraceus</i> | | | 1,439 |
| <i>T. vellosi</i> | <i>H. speciosus</i> | | | 1,436 |
| <i>T. alba</i> | <i>H. albus</i> | LC (2018) | Lc | 1,373 |
| <i>T. heptaphylla</i> | <i>H. heptaphyllus</i> | | Lc | 1,246 |
| <i>T. chrysotricha</i> | <i>H. chrysotrichus</i> | | | 898 |
| <i>T. Ipê</i> | <i>H. heptaphyllus</i> ; <i>H. impetiginosus</i> | NT (2020) | | 317 |
| <i>T. barbata</i> | <i>H. barbatus</i> | NT (2020) | | 316 |
| <i>T. cassinoides</i> | | | Vu (end.) | 223 |
| <i>T. umbellata</i> | <i>H. umbellatus</i> | LC (2018) | | 115 |
| <i>T. aurea</i> | | | | 67 |
| <i>T. caraiba</i> | <i>T. aurea</i> | | | 61 |
| <i>T. chrysantha</i> | <i>H. chrysanthus</i> | VU (2020) | | 50 |
| <i>T. angustata</i> | | | | 24 |
| <i>T. roseo-alba</i> | <i>T. roseoalba</i> | NT (2020) | | 24 |
| <i>T. avellanedae</i> | <i>H. impetiginosus</i> | NT (2020) | | 21 |
| Total | | | | 255,723 |

A study from 2022 noted that Brazil exported at least 449,381 m³ between 2017 and 2021 in shipments that were listed as only containing Ipê products, suggesting that Brazilian Ipê exports grew at least 76% (by volume) between the periods 2010–2016 and 2017–2021.

During 2013–2015, the main importing countries for exports from Brazil were the USA with 47,372 m³, France with 23,868 m³, and Belgium with 11,763 m³ of Ipê sawn wood. The EU (primarily France, Belgium, Spain, and Portugal) and the UK reportedly purchased 45% of all Ipê species (by volume) exported by Brazil between 2017–2021. French imports of Brazilian Ipê increased by 84% and Belgian imports by ~70% by volume for the period 2017–2021, compared with previously reported figures for 2010–2016.

According to the International Tropical Timber Organization (ITTO), Brazil exported a total of 83,992 metric tonnes of Ipê sawn wood between 2018–2021. The USA purchased roughly 36% of the Ipê exports based on volume over the period, with Canada purchasing 4% during the same span. While the USA imported less than the EU, the USA remains the primary single global buyer of Ipê. It was estimated that USA consumption of Ipê from Brazil has increased by 126% for the period 2017–2021, compared with previously reported figures for 2010–2016. According to the ITTO the USA imported 260,203 m³ of Ipê sawn wood and 148,983 m² Ipê wood flooring between 2018–2020.

Colombia

Colombia reported a total of 126,223 m³ of harvests of Ipê between 2019–2021 (Table 5). The main species in trade were *H. billbergii* and *H. chrysanthus*. Colombia noted that 2021 was the year in which the highest volumes were exploited. *It is unclear how much this was, and whether these volumes were also exported internationally.* Three export licences were approved in 2019–2020 for a total of 114m³, of which 82 m³ were for *Tabebuia* spp., 20 m³ were for *T. rosea* and 12 m³ were for *H. serratifolius*.

Table 5. Exploited volumes of Ipê by Colombia in the period 2019–2021. Bold indicates the species proposed under Annex 2a criterion B.

| Reported species | Equivalent species according to proposed CITES Standard Reference | IUCN Red List | Volume (m ³) |
|--------------------------------|---|------------------|--------------------------|
| <i>H. billbergii</i> | | | 72,637 |
| <i>H. chrysanthus</i> | | VU (2020) | 23,899 |
| <i>H. serratifolius</i> | | EN (2020) | 1,727 |
| <i>Handroanthus</i> spp. | | | 348 |
| <i>H. guayacan</i> | | LC (2020) | 14 |
| <i>T. rosea</i> | | LC (2018) | 25,700 |
| <i>Tabebuia</i> spp. | | | 1,731 |
| <i>T. ochracea</i> | <i>H. ochraceus</i> | | 156 |
| <i>T. chrysantha</i> | <i>H. chrysanthus</i> | VU (2020) | 9 |
| <i>T. heterophylla</i> | | LC (2019) | 2 |
| Total | | | 126,223 |

Ecuador

H. chrysanthus was reported as among the 10 most used and commercialised forest species and from 2012–2013, 7,775 m³ was logged.

French Guyana

The annual inventory of *H. serratifolius* and *H. impetiginosus* in French Guyana averaged 1,099 m³ between 2007–2021, and all timber was PEFC (Programme for the Endorsement of Forest Certification) certified. *Over the same period, the average annual exploitation was 881 m³, equivalent to 175 logs annually (CoP18 Prop. 49, 2019).*

Guyana

The 2019–2020 biennial review noted that Guyana exported ~2,000 m³ of *Tabebuia* spp. in 2017 (ITTO, 2020).

Mexico

Mexico reported exports of 183 m³ of sawn wood of *R. donnell-smithii* to Guatemala in the period 2010–2011. In 2012, Mexico exported 510 pieces of logs of *R. donnell-smithii* to Honduras.

Peru

Peru reported exports of *H. serratifolius* totalling 1,131 m³ from January 2016–March 2018. The biggest importers from Peru were China and the Dominican Republic. *Table 3 notes that Peru was reported as the origin for 5,744 m³ of Ipê between 2017–2021 (Norman and Zunino, 2022).*

Suriname

ITTO biennial reports for the period 2011–2015 include exports from Suriname of 5,000 m³ logs and 1,000 m³ sawn wood of *H. serratifolius*, destination unspecified. *The 2019–2020 biennial review noted that Suriname exported ~3,000 m³ of H. serratifolius and ~13,000 m³ of H. capitatus (ITTO, 2020).*

Venezuela

Venezuela reported exports of 29,637 m³ *T. rosea* and 20,491 m³ *H. impetiginosus* during 2007–2017 (Table 6). During this period, exports fluctuated and peaked in 2007 (Figure 1).

Table 6. Timber volume of Ipê reported as exports by Venezuela from 2007–2017 (CoP18 Prop. 49, 2019). Bold indicates the species proposed under Annex 2a criterion B.

| Reported species | Equivalent species according to proposed CITES Standard Reference | IUCN Red List | Volume (m ³) |
|-------------------------------|---|------------------|--------------------------|
| <i>H. impetiginosa</i> | <i>H. impetiginosus</i> | EN (2020) | 20,491 |
| <i>T. fluviatilis</i> | | LC (2020) | 232 |
| <i>T. rosea</i> | | LC (2018) | 29,637 |
| Total | | | 50,360 |

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**

Generally, the traded wood is not specified to species level, which does not allow species-specific estimations of trade volumes. The wood can be traded with the scientific name or the local common name, but at least 28% of the exports by weight were either only reported as Ipê or another generic common name which did not include any information on the botanical name.

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, resulting in distinguishing species being difficult. *A clear differentiation of the individual species is not possible on either the macroscopic or microscopic level (Koch, in litt., 2022).* There are no identification guides for all species of 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* at 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 (Koch, in litt., 2022).*

Experts from the Thuenen-Institute of Forest Genetics noted that although it is possible to distinguish H. serratifolia from H. impetiginosus using genetic markers their use is not widespread (Blanc-Jolivet, in litt., 2022; Schroeder, in litt., 2022). In addition, molecular techniques were noted to be the only method for identification of these species and required significant expertise (Blanc-Jolivet, in litt., 2022). Identification of origin was noted as only possible using isotope analysis (Blanc-Jolivet, in litt., 2022).

The following species are commonly confused with Ipê in trade: *Dipteryx olifeira*, *Bulnesia sarmientoi* (CITES Appendix II), *Guaiacum* spp. (CITES Appendix II), and *Chlorocardium* spp. (Koch, in litt., 2022). The following species are additionally commonly confused and all are present in range States where the three proposed genera are also present (G. Koch, in litt., 2019): *Acosmium* spp.: in Bolivia and Brazil (Rodrigues and Tozzi, 2009) *Leptolobium* spp. (*Lapachillo*, *lapachin*): from Mexico to Argentina (Rodrigues and Tozzi, 2012) *Dicorynia guianensis* Amsh.: in Suriname and French Guiana (ITTO, 2019) *Dicorynia paraensis* Benth. (Angélique, *basalocus*): in Brazil, Colombia, and Venezuela (Canteiro, 2018) *Dipteryx odorata* (Aubl.) Willd.: in Brazil, Colombia, French Guiana, Guyana, Suriname, and Venezuela (Requena Suarez, 2017) *Dipteryx alata* Vog. (Cumarú): in Brazil (Raquena Suarez, 2021). *Dipteryx* spp. is currently proposed for listing in CoP19 Prop. 48.

Live standing trees are easily distinguished.

One of the Spanish common names used for the genera “Guayacan” is also used to refer to timber of CITES-listed species such as species of the genus *Guaiacum* and *Bulnesia sarmientoi*.

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

Additional information

Threats

The main threats to Ipê are deforestation and logging for both domestic and international trade. Latin America has suffered from very high deforestation rates in the last three decades, reducing the potential suitable habitat for Ipê trees significantly. Whilst settlement and agricultural development are major causes, expansion of road networks and mining are also threats to South American forests.

Locally, *Handroanthus*, *Tabebuia*, and *Roseodendron* species are also used domestically in the construction of houses and bridges, pavements, decks, exterior woods, handicrafts, and for medicinal purposes. Besides timber commodities, herbal products made from the inner bark of some species of *Handroanthus*, including, but not limited to *H. impetiginosus*, are traded internationally as “lapacho”, “pau d’arco” or “taheebo”. However, while trade data are not available for these products, no substantial role in international trade could be identified compared to timber trade volumes, and no information was found to suggest that this use has impacts on wild populations of *Handroanthus*, *Tabebuia*, and *Roseodendron*. A book from 1997 noted that the international market for *Tabebuia* bark was growing and that this was increasing pressure on wild stands of the trees, with overharvesting reportedly occurring in Peru (Sheldon et al., 1997).

Deforestation

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.

Brazil is one of the most extensively forested countries in the world, with 463 million hectares of forests; 90% of them are in the Amazon Basin and the Cerrado. Brazil experienced rapid deforestation with mean annual rates between 0.2–0.4% during 2000–2015. According to Brazil’s National Institute for Space Research, the total area deforested in the Brazilian Amazon in 2011–2012 was 460,000 ha, compared to 2.8 million hectares in 2004. In 2019, an area of 10,129 km² of forest was clear-cut, which is an increase of 34% compared to 2018 (7,536 km²). In 2020, deforestation was estimated to be 11,088 km², representing an increase of 47% and 9.5% compared to 2018 and 2019, respectively. Deforestation was driven primarily by demand for agricultural land with many of the forest conversion being illegal. In recent years, Ipê harvests have declined or ceased in most of the old, well-developed logging centres in eastern Amazonia, but at the same time, new logging frontiers in the remote central and southwestern Amazon region (where access and infrastructure had been poor) were opened up, with Ipê amongst the main species being harvested.

In Mexico, the negative effect of land use change, deforestation, elimination of ecotypes, clandestine logging, selective logging, fires, and introduction of exotic species have been documented to have a negative effect on *Roseodendron donnell-smithii* and to cause severe genetic degradation. Furthermore, Mexico lost 16% of forest cover from 1986–2000, which mainly affected the dry tropical forest with an annual deforestation rate of 3.7%; and forest cover loss increased to 22% between 2000–2011. The state of Michoacán, Mexico, lost almost 525,260 ha over the same 10-year period, which was partially being recovered with the reforestation of *T. rosea*.

In Colombia, the reduction of forest fragments driven by the expansion of areas for agricultural and livestock use have restricted the population of *H. chrysanthus* to the driest zone of transition of xerophytic scrub vegetation in the south of the country.

In Bolivia, based on spatial satellite imagery 276,000 and 281,283 ha were deforested in 2004 and 2005 respectively. Until 2010, approximately 4.6 million ha of forest were lost, corresponding to 10% of the area originally covered by forest.

In Peru, about 12,849 km² of forests are cut down annually—reportedly 80% cut illegally.

In Ecuador, originally about 35% (28,000 km²) of the land surface was covered by dry forest, with 80–90% of the original dry forest vegetation having disappeared due to land use change. The national annual deforestation from 2008–2020 ranged between 214.8 km²/year and 310 km²/year.

In Venezuela, the deforestation between 1990–2010 was 288,000 ha/year.

Logging

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 (Schulze, in litt., 2019). A study by Richardson and 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.

The decline in other tropical timber species such as Big-leaf Mahogany *Swietenia macrophylla* has led to an increase in demand for species of Ipê on the international market, which has led to declines in some species. In Brazil, in particular, it was suggested that exploitation may lead to the extinction of *Handroanthus* species as a result. *Handroanthus* species are vulnerable to logging due to their low natural density and low growth rates.

In Brazil, a study examined the response of *H. serratifolius* and *H. impetiginosus* populations to logging in various locations in the eastern Amazon. The authors reported that the widely held assumption that 30-year cutting cycles combined with a minimal exploitable diameter (MED) of 50 cm dbh and 90% logging intensity (10% of the trees above MED should be left as seed trees) is sustainable is based on a limited number of small plots in Amazonia that do not consider logging impacts on timber species populations or recovery rates. The study projected that under the current logging regime, in some concessions projected volumes for a second logging cycle will be as low as 2–3% for *H. impetiginosus* and 4–12% for *H. serratifolius*. After selective logging, there was no evidence that the composition of the timber species and the overall value of the forest recovered, suggesting that the timber species with the highest commercial value, such as Ipê, would not show sufficient population recovery and become rare or economically extinct in former timber frontiers.

Illegal logging and trade

Illegal logging is reported to be a significant threat to species of Ipê, including *H. serratifolius*. The Supporting Statement notes that in the Brazilian Amazon, illegal timber laundering through overestimating the inventoried timber volume followed by fraudulently obtained official documentation is widespread. Greenpeace-Brazil, in collaboration with the State Secretariat for the Environment and Sustainability of Pará (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 plans listed Ipê in their inventories, approximately 14% of which overestimated the timber volume to be harvested during the logging intervention (3,000 m³ per concession or 60% above the species average of 2.4 m³/ha). Although illegal logging has been reported to have fallen to 54–75% in the Brazilian Amazon from 2003–2013, it still accounts for 35–72% of logging in this area. A comparison of satellite data with official records of licences issued by the SEMA suggested that 78% of the area logged from August 2011 to July 2012 in the state was not licensed. In 2017, 74% of the total volume of 33,389 m³ licensed for logging had a high risk of being overestimated in pre-harvest inventories. Presumably owing to its high value, Ipê was found to be the timber with the highest probability of fraudulent inventory data. The occurrence of illegal harvest suggests that the legally authorised timber volume is not sufficient to meet demand.

The Supporting Statement describes how, in 2016, the Brazilian Institute for the Environment and Renewable Natural Resources (IBAMA) dismantled a criminal scheme for the extraction, transport and commercialisation of

illegal wood in the northern region of Mato Grosso, one of the main timber producing states in Brazil's Central West region. In the action, approximately 350 m³ of sawn Ipê was secured (~18 loaded trucks), which were valued at ~USD567,000. The shipment was destined for the international market, mainly Belgium, the USA, and France. In January 2018, 400 containers containing wood from the Brazilian Amazon were seized by IBAMA and by the Brazilian Federal Police. Among them was 475 m³ of *Handroanthus* sawn timber without legal provenance. More than 43,000 m³ of wood were traded using fraudulent documentation from just one company in 2015, including about 12,000 m³ of Ipê, potentially worth at least USD7 million if processed and exported. Between 2016–2017, 10,171 m³ of Ipê wood from forest management plans with evidence of illegality was imported by 37 American companies. In addition, 11 EU countries, including France, Portugal, Belgium and the Netherlands imported 9,775 m³ in that timeframe, some of which was assumed to have come from an illegal origin. A strong driver for illegal trade appears to be the high prices in international markets for Ipê timber. The high export value of Ipê (up to USD2,500/m³ at export ports) gives loggers and sawmills a motivation to build illegal roads, leading to growing forest degradation and the destruction of biodiversity, and also to obtain official documentation through fraudulent inventory reports to launder and subsequently commercialise illegally harvested *Handroanthus* trees. Brazil's non-integrated forest licensing and control system has been reported as being unreliable, with official documentation considered inadequate, meaning it is almost impossible to distinguish between legal and illegal Ipê timber.

In Venezuela, 65 m³ of wood and 1,062 units of *T. rosea* products were seized between 2013–2018. In Peru, 119.16 m³ of wood, 14.96 kg of bark and 4,738 pieces of *Tabebuia* (potentially *Handroanthus*) were seized during the period 2011–2017. In Colombia, between 2010–2020, a total of 270 m³ was seized, including *H. billbergii* (93 m³), *H. chrysanthus* (59 m³), and *T. rosea* (118 m³), seized due to a lack of permits. Mexico reported 9 seizure events from 2017–2019, involving *H. chrysanthus* (0.485 m³), *R. donnell-smithii* (4.16 m³), and *T. rosea* (28.98 m³ and 208 pieces).

Conservation, management and legislation

About half of the Brazilian forest area (243 million ha) has been identified as PFP "Permanent Forest Property", including public, federal and private forests (Indigenous Lands and Legal Reserves based on long-term land ownership for forest users). Forest management units for timber production within the PFP comprise 34.25 million ha or 14% of the PFP. Owners and users are responsible for management. Forest area that is not classified as PFP is open for conversion to other land uses. A positive example for a Sustainable Use Conservation Unit is the Altamira National Forest in the central-southwestern part of Pará. Its area of 689,000 ha is predominantly covered by dense ombrophilous forest and includes a protected area that represents a significant extension of ancient forests. Altamira is embedded in the so-called Xingu River Basin Corridor, with an area of more than 26 million ha and 18 Indigenous Lands (24 ethnic groups), various sustainable uses and fully protected areas, identified as areas of great importance for the preservation of biodiversity.

Mainly due to management actions, including the declaration of an effectively managed and enforced ban in 1978 and the application of *in-vitro* cultures, the populations of *H. chrysanthus* and *H. billbergii*, have recovered in Ecuador.

There are broad bans/restrictions on timber logging/export in several range States, examples are included in Table 3.

Table 3. Legislation in some of the range States.

| Range State | Legislation |
|-------------|---|
| Brazil | <p>The export of unfinished timber of native species (i.e., destined to be processed abroad) is prohibited according to Normative Instruction 15/2011, amended by Normative Instruction 13/ 2018.</p> <p>A prerequisite in Brazil for timber exploitation is an approved forest management plan that establishes how forestry activities will be carried out in a specific area which must be submitted to the environmental institutions of the Brazilian departments. Once approved, an annual operation plan must be submitted for the following year's harvest, including a forest inventory and a detailed logging map showing the trees to be felled.</p> <p>The minimum exploitable diameter (MED) is 50 cm for all commercially exploitable species, including Ipê, and 10% of the trees above the MED must be left standing as seed trees.</p> <p>The 2006 legislation transferred the responsibility for approval, monitoring and evaluation of forest management plans (FMPs) to individual states. The landowner or company proposing the forest management plan submits it to the authority responsible for timber regulation (State Environmental and Sustainability Secretariat (SEMAS), which</p> |

| Range State | Legislation |
|-------------|---|
| | registers estates and licences, including those for logging, on a computerised system. The registration of timber producers, and the monitoring of the chain of custody through an electronic system intended to track timber and record transactions is the responsibility of state governments. All timber sales and shipments are accompanied by the corresponding quantity of credits entered into this system. |
| Bolivia | Export of unprocessed forestry products is subject to restrictions and highly regulated, mainly through forest certification (from 1996 onwards, last updated 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 industrialisation have been met" (from 2005 onwards, last updated 2016). |
| Mexico | The Environmental Attorney General's Office (PROFEPA) will carry out technical investigation, inspection and surveillance actions in forestry matters, carrying out documentary verifications on forest management programmes, justifying technical studies, the authorisations issued by the Secretariat or the Commission or the reports rendered. During the technical investigation, PROFEPA may obtain any data, information, or indication that it deems necessary to clarify the legality of any operations. They then prepare a report in which the information collected is analysed, to detect cases in which it is appropriate to exercise its powers of inspection, surveillance and sanctions. |
| 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). Reportedly forest concession agreements require reduced impact logging practices, cutting rotations of at least 20 years, and the retention of a minimum of 10% of mature adult trees (seed trees) of each harvested species to enable regeneration. |
| Suriname | Under the Forest Management Law of 18 September 1992, <i>H. serratifolia</i> (<i>Tabebuia serratifolia</i>) is listed as a Category A species (market-worthy timber species), meaning that the minimum exploitable diameter is 35 cm (although exceptions can be made by the Forestry Department). Additionally, a permit is required for the export of "raw wood, round wood, round or felled pole wood, processed wood, wood products and forest by-products". |
| Venezuela | Reportedly, forest concession holders may only extract trees greater than 40 cm dbh (not specific to Ipê). In 2006, Venezuela decreed a ban on <i>T. spectabilis</i> , prohibiting the exploitation, use and any type of intervention on trees of this species throughout the national territory. |

Artificial propagation

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 *T. rosea* in 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) that is managed in commercial plantations, as well as being used to enrich secondary forests and degraded paddocks. *Exploitation of T. rosea in Mexico is reportedly primarily for local use* (Martínez Salas, in litt., 2019). *T. heterophylla* is also grown in plantations in Puerto Rico (USA). *There are no Ipê plantations in Brazil.*

To date, *H. serratifolius* has rarely been utilised in forest plantations due to the lack of information regarding its development in both nursery and field conditions. The costs for the production of seedlings were found to be five times higher than for the production of *Swietenia macrophylla* seedlings.

Several species of *Handroanthus* and *Tabebuia* are commercially nursery-grown for urban landscape planting and street trees in countries with suitable environmental conditions. However, no indication was found on timber plantations outside of South America, Central America, and the Caribbean.

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).

Potential benefit(s) of listing for trade regulation

The absence of international mechanisms to monitor and control international trade in these vulnerable and endangered species is believed to have contributed to their overexploitation, very high international trade volumes as well as laundering and illegal trafficking. A CITES-listing could help to regulate this trade towards a

sustainable volume and in a manner that is not detrimental to the survival of the species. The phenomenon that all 113 species might be traded under the same trade name and the fact that timber of the different species is hardly distinguishable suggests the listing of all species of *Handroanthus*, *Tabebuia*, and *Roseodendron* would be necessary to avoid enforcement problems and loopholes for timber laundering.

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Inclusion of *Rhodiola* spp. in Appendix II with annotation #2

Proponents: China, European Union, Ukraine, United Kingdom of Great Britain and Northern Ireland, United States of America

Summary: *Rhodiola* is a diverse genus of perennial herbs occurring in colder parts of the Northern Hemisphere. They are generally long-lived and slow growing, with some species taking up to 20 years to mature in the wild. The taxonomy is unresolved, but the proponents follow a 2003 review, which recognises 58 species within the genus.

Two species (Rose Root *Rhodiola rosea* and Arctic Root *Rhodiola crenulata*) are proposed for inclusion in Appendix II to address trade threats; the remaining species are proposed as lookalikes. The proposal includes annotation #2: "All parts and derivatives except: a) seeds and pollen; and b) finished products packaged and ready for retail trade." *Rhodiola rosea* is extant in 28 range States across Asia, Europe, and North America, and *Rhodiola crenulata* is extant in Bhutan, China, India, and Nepal where it is confined to altitudes of 2,800–5,600 m above sea level. One species *R. marginata*, endemic to Bhutan, was assessed globally for the IUCN Red List as Least Concern in 2017. Other species have not been globally assessed.

The rhizomes of *Rhodiola* have been used in traditional medicine systems across much of their range. Both domestic and international demand for *Rhodiola* species has increased in the last 20 years accompanied by increased diversity in products available, including teas, pills and herbal medicines, supplements, energy drinks, alcoholic beverages, and cosmetics. Products containing *Rhodiola* have been tested in clinical trials investigating treatment of fatigue, sleep disorders, and depression. *Rhodiola* is an ingredient of one of the most popular Traditional Chinese Medicine (TCM) formulations recommended for treating COVID-19 in China, with approval for this medicine registered in 30 countries.

National red list assessments are available for *R. rosea* for 21 range States. The species has been nationally classified as least concern or secure in eleven countries in Europe and North America and as threatened with extinction or rare in ten. *R. crenulata* was nationally assessed as endangered in China in 2017 and one recent study in Bhutan has found *R. crenulata* to be uncommon and patchily distributed in some areas. According to known occurrence records for *R. crenulata*, the majority of the species' range is in China.

Global trade data are not available for either species but available evidence suggests that *R. rosea* is predominantly wild harvested for international trade in four range States (China, Kazakhstan, Mongolia, and Russian Federation) and *R. crenulata* in one (China). *Rhodiola crenulata* is said to be the most widely traded species in China and has also been found to be traded under the name of *R. rosea*. In both Russian Federation (for *R. rosea*) and China (for *R. rosea* and *R. crenulata*), research indicates that wild harvest to meet commercial demand is leading to population declines.

International exports from China, particularly in the form of extracts, are said to be a key driver of commercial trade in *Rhodiola*. China sources *R. rosea* both domestically through wild harvest, and through imports of raw wild-harvested material (e.g., roots and rhizomes) from Russian Federation, Kazakhstan, and possibly Mongolia. Stocks of *R. rosea* in China have been reported to be declining, with imports from neighbouring regions now needed to meet commercial demand. There is no clear evidence of cross-border trade of *R. crenulata* into China from Bhutan, Nepal, and India although there may be some trade from the Linghsi district of Bhutan.

The Russian Federation also imports roots of *R. rosea* from Kazakhstan and Mongolia, and exports finished products to countries in Europe and Asia. Harvest in Russian Federation is reported to have affected populations in the Altai region in particular. In all four countries with the most evidence of wild harvest for commercial export, the species has been documented in national red list

assessments as either rare (*R. rosea* in Kazakhstan and Russian Federation), vulnerable (*R. rosea* in Mongolia and China), or endangered (*R. crenulata* in China).

Data on volumes in trade are sparse. In Russian Federation, 85 t of dried *R. rosea* rhizomes were reported as exported between 2006 and 2008. A study in 2017 estimated that 500 t of dry rhizomes of *R. rosea* were exported from the Xinjiang region in China each year for manufacturing in eastern China, with most of this material thought to be exported internationally.

All available evidence indicates that the *R. rosea* and *R. crenulata* in commodities exported from China are entirely wild-sourced. Commercial cultivation is known to take place in some countries, including Canada, USA, and possibly Ukraine, and Russian Federation, although this is generally at a small-scale with products intended for domestic use. The relatively long maturation period of the plants (five years) and low profit margins present significant barriers to large-scale commercial cultivation. For *R. crenulata*, cultivation is reportedly additionally challenging due to its high-altitude growing requirements. There are no indications of commercial cultivation of *R. rosea* in Mongolia or Kazakhstan, and no evidence of cultivation of *R. crenulata* in any country.

Permits are required for the wild harvest of both *R. rosea* and *R. crenulata* in China. In Russian Federation, *R. rosea* is nationally protected with wild harvest for commercial purposes illegal in all areas aside from populations in Altai Krai, Krasnoyarsk, Tuva and Magadan. In Kazakhstan, *R. rosea* is thought to be protected, although one source indicates that wild harvest in state forests is allowed and subject to quotas. Regulations for *Rhodiola* harvest are not clear in Mongolia. The species is fully protected in Bosnia and Herzegovina and Bulgaria and offered some form of protection in six other range States, most of which are also in Europe.

It is difficult to identify the species of *Rhodiola* in trade for live plants and dried rhizomes and mixing of different species in trade is thought to occur early within supply chains.

Analysis: There is clear evidence of international trade in and commercial demand for *Rhodiola* products that may be increasing. *Rhodiola rosea* and *R. crenulata* are the species that occur most frequently in trade, with most commercial cultivation (*R. rosea*) small scale or for domestic purposes, and most international exports likely to originate from wild harvest.

R. crenulata populations appear to be wild harvested for commercial international trade in China only, where it is now considered endangered. From known occurrence records, most of the range is also in China. There are indications of depletion from collection in parts of Bhutan and there may be harvesting of the species in India for domestic purposes. Given its relatively restricted distribution and the clear impact of trade in China, *R. crenulata* appears to meet the criteria for inclusion in Appendix II in Annex 2a of Res. Conf. 9.24 (Rev CoP17).

R. rosea has a very wide distribution. Populations have been assessed as rare or vulnerable in countries where they are most heavily harvested (Kazakhstan, Mongolia, Russian Federation and China). In large parts of the remainder of its range there is little indication of extensive harvest and the species has been assessed as not of conservation concern. It seems unlikely therefore that it meets the criteria for inclusion in Appendix II in Res. Conf. 9.24 (Rev.CoP17). However, because of difficulty in distinguishing between species of *Rhodiola* in dried form and the known mixing of products in trade, *R. rosea* along with other members of the genus meet lookalike criteria for inclusion in Appendix II in Annex 2bA of Res. Conf. 9.24 (Rev. CoP17).

Annotation

Annotation #2 would include "All parts and derivatives except: a) seeds and pollen; and b) finished products packaged and ready for retail trade." It is not currently known what proportion of products exported by range States would be considered finished products packaged and ready for retail trade. Res. Conf. 9.24 (Rev. CoP17) states (operative paragraph 7); annotations to proposals to amend Appendix I or Appendix II "should...include those specimens that first appear in international trade as export from range States." If a significant part of the export trade is in finished products inclusion of this annotation, were the proposal to be accepted, would go against the intention of the Resolution, although there may be challenges to implementing their inclusion.

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 taxonomy of the genus remains partially unresolved, with the number of accepted species ranging from 58–90 according to different sources.

No Rhodiola spp. are currently listed in CITES and the proponents follow the taxonomy of Obha (2003), which recognises 58 species in the genus. The EU has submitted a Proposal to CoP19 to include Obha (2003) as the reference for CITES standard nomenclatural reference for Rhodiola spp., should the Proposal to include Rhodiola spp. in Appendix II be accepted (CoP19 Doc. 84.4).

Rhodiola rosea has 55 synonyms and Rhodiola crenulata has 10. These are listed in Annex 1 of the Proposal.

Range

Rhodiola is a diverse genus of perennial herbs with a wide distribution range spanning across the northern hemisphere. Species in the genus are commonly associated with subarctic and alpine areas. The centre of diversity is found in China, which hosts 60% of the world's *Rhodiola* species, according to the Flora of China.

The distribution of the two species most commonly traded and proposed for inclusion in Appendix II for satisfying criterion B of Annex 2a of Res. Conf. 9.24 (Rev. CoP17) is as follows:

R. rosea: Andorra, Austria, Bosnia and Herzegovina, Bulgaria, Croatia, Czech Republic, Denmark (Faroe Islands and Greenland), Finland, France, Germany, Iceland, Ireland, Italy, Norway, Poland, Slovakia, Spain, Switzerland, Ukraine, United Kingdom, China, Democratic People's Republic of Korea, Japan, Kazakhstan, Mongolia, Russian Federation, Canada, and USA.

R. crenulata: Nepal, India, Bhutan, and China.

IUCN Global Category

Rhodiola marginata: Least Concern (assessed 2015, ver. 3.1)

The remaining species in this genus have not been globally 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**
- 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

R. rosea

No data are available on population trends across the whole of its circumpolar distribution.

Out of 28 range States for this species, the Proponents show that national assessments are available for 19, which are summarised in Table 1. An additional national red list assessment was identified for Kazakhstan, where the species is classified as "rare" (Kubentayev et al., 2021), and a NatureServe assessment for the species in the USA, which classified the species as "secure" (Table 1). Ten out of the 21 range States that have national assessments for the species classified it as threatened or rare. Ten, mainly in Europe, classified the species as least concern or secure, and one as near threatened.

China has separately assessed three subspecies of *R. rosea*, and an additional species *R. sachalinensis* that is included taxonomically within the species of *R. rosea* by Obha (2003). Their classifications are as follows: *R. rosea* var. *rosea* (vulnerable, 2017), *R. rosea* f. *purpurascens* and *R. rosea* var. *microphylla* (least concern, 2013) and *R. sachalinensis* (vulnerable, 2017).

Information on populations, or national threatened status could not be found for Andorra, Croatia, Denmark, Italy, Poland, Spain, or Japan, although the species is assessed as least concern on the European Red List of Medicinal Plants in 2014 (Allen et al., 2014).

Table 1. National conservation assessments from range States of *R.rosea*, where available (Sources: SS, * Kubentayev et al., 2021, + NatureServe, 2015).

| Country | National Red Listing Status (year) |
|---------------------------------------|------------------------------------|
| Czech Republic | critically endangered (2012) |
| Bulgaria | critically endangered (2001) |
| Austria | endangered (1999) |
| Mongolia | vulnerable (2018) |
| Bosnia and Herzegovina | vulnerable (2014) |
| Ukraine | vulnerable (2012) |
| Germany | threatened (2018) |
| Russian Federation | rare (2017) |
| Kazakhstan* | rare (2014) |
| Democratic People's Republic of Korea | near threatened (2005) |
| Finland | least concern (2019) |
| France | least concern (2019) |
| Iceland | least concern (2018) |
| Ireland | least concern (2016) |
| Switzerland | least concern (2016) |
| Norway | least concern (2015) |
| Slovakia | least concern (2015) |
| UK | least concern (2014) |
| Denmark; Greenland | not assessed; least concern (2018) |
| Canada | secure (2015) |
| USA | secure (2015)+ |

R. crenulata

The species occurs at high altitudes between 2,800 and 5,600 m above sea level with a fragmented population across the southern parts of the Qinhai-Tibetan Plateau (Brinckmann and Cunningham, 2022). According to combined data from the Global Biodiversity Information Facility Secretariat and the Kunming Institute of Botany iFlora database, the majority of the known occurrence records for this species occur in China (Brinckmann and Cunningham, 2022).

Bhutan: *Rhodiola crenulata* occurs in the regions of Bhumthang, Dagala and Lingshi, in addition to the Jigme Khesar Strict Nature Reserve in Haa District (Brinckmann and Cunningham, 2022). A recent study conducted in Lingshi (a local administrative unit in northeast Bhutan, on the border with China), which assessed priority species to determine whether wild harvest could be sustainable, found the species to be vulnerable and patchy in distribution, critically low in plant density (0.4 plants per m²), and among the rarest of 16 wild-collected medicinal plant species.

China: The species occurs in Qinghai, Sichuan, Tibetan Autonomous Region and Yunnan (Brinckmann and Cunningham, 2022). It is classified as endangered (2017) on the national Red List of China. Relatively large populations of *R. crenulata* can however still be found in remote locations (Cunningham et al., 2020).

India: The species occurs in the Trans Himalayan zone of Ladakh and potentially Sikkim (Brinckmann and Cunningham, 2022).

Nepal: The only known occurrence of the species is in the northeastern part of Nyeshang Valley in Manang District (Brinckmann and Cunningham, 2022).

No population assessments of *R. crenulata* could be located in India or Nepal. Neither quantitative nor qualitative information on current and past trends regarding the abundance of *R. crenulata*, or national Red List assessments, could be found for Bhutan, India, or Nepal.

International trade

The rhizomes of some species of *Rhodiola*, also known as “roseroot”, have historically been part of traditional

medicine systems across the majority of the genus' range; however, industrial use of *Rhodiola* species started in the mid-20th century in the former Soviet Union, and the past 20 years have seen a rapid rise in the diversity of *Rhodiola* products available on the market and an expansion in both domestic and international trade in the genus.

With several clinical trials investigating the effectiveness of *R. rosea*-containing products in treating the effects of fatigue, sleep disorders and depression, international demand is further projected to increase. *Rhodiola* is also an important ingredient of *Lianhua Qingwen*, one of the TCM formulations recommended for treatment of COVID-19 (Zhang in litt., 2022). This product has been authorised by the National Medical Products Administration in China to treat COVID-19 (Shen and Yin, 2021) and according to Yiling Pharmaceuticals, in China the product has obtained registration approval and import licences in 30 countries including Canada, Russian Federation, and Singapore (Yeping and Xinyi, 2022). One source in China stated that sales of this medicine domestically and internationally are a driver of increased demand for *R. rosea* (Yi, 2022).

Trade is currently predominantly in *R. rosea* and *R. crenulata*, which are traded in the form of raw material (as dried rhizomes, chips, or powder) and semi-processed commodities (extracts), and finally sold to end consumers as dried coarse ground and chipped root for teas, as well as pills/tablets, concentrated extracts and powders marketed as herbal medicine, dietary supplements, sports and energy drinks, alcoholic beverages and cosmetic products. Harvesting is based on the exploitation of the large rhizomes (rootstocks) and/or whole plants, and the majority of material currently traded is wild-sourced. Harvest is focused on reproductively mature individuals, so commercial levels of exploitation have an increased potential to impact recruitment and long-term population viability.

Some *Rhodiola* species are also traded as live ornamental plants for gardening purposes. The extent of horticultural trade is difficult to gauge as there are few global data to track such trade for species that are not CITES-listed. However, no specific collectors' market exists and demand is assumed to be generally low. Nurseries and horticultural associations contacted in the UK and Germany reported international trade in live *Rhodiola* plants to be very rare.

Trade volumes for *Rhodiola* are uncertain, and there are no known genus-specific harmonised system tariff codes (HS Code) assigned in any country for any commercial forms of the genus. No imports or exports of species within the *Rhodiola* genus are reported in LEMIS.

R. rosea

Prominent international trade routes and commodities for *R. rosea* identified from information in the proponents' Supporting Statement and in this research are summarised in Table 2.

Table 2 shows that most evidence of exports from wild-harvested forms of the species is restricted to the four range States of China, Kazakhstan, Mongolia, and Russian Federation. Where commodities are mentioned, most exports are in the form of raw materials, with some evidence of exports of end products from Russian Federation (from two companies found in online searches) and China (from research presented in the Supporting Statement). Extraction houses in eastern China are said to process raw materials of *R. rosea* for value-addition and export in the form of bulk extracts as well as end products. With a lack of quantitative trade data, it is not clear what proportion of exports from China or Russian Federation are extracts, raw materials, and finished products.

The proponents state that commercial exploitation of *R. rosea* for medicinal purposes in national and international markets has been reported in Norway, which classified *R. rosea* as least concern in 2015, but no specific details on trade routes or importing countries are given. Collection in European countries is considered to have less economic importance because of the high labour costs and the difficulties of transportation in high mountain areas, and many populations in Europe are also legally protected.

For the other 22 range States of this species, 10 of which have nationally classified *R. rosea* as least concern or secure, it is not clear from the available evidence that there is wild harvest for commercial export in *R. rosea*.

Table 2. Prominent international trade routes and commodities for wild harvested *R. rosea* identified in research (Source SS = Supporting Statement, O = brief preliminary search of online products containing *R. rosea*, Y = Yi, 2022).

| From | To (source, commodity if stated) |
|------------|---|
| China | USA, UK, Australia, Canada, Republic of Korea, Pakistan, and New Zealand (SS, end products). "Internationally" (SS, extracts, end products) |
| Mongolia | Russian Federation (SS, raw materials) |
| Kazakhstan | Russian Federation (SS, raw materials) |

| From | To (source, commodity if stated) |
|--------------------|--|
| Russian Federation | China (SS, rhizomes, Y, not specified) |
| Russian Federation | Germany (O, raw materials), USA (O, raw materials), Spain, Serbia, Montenegro, Bosnia and Herzegovina, Hong Kong SAR (O, finished products), France, Slovakia, Germany, Armenia, Austria, Czech Republic, Georgia, Kyrgyzstan, Uzbekistan, Viet Nam (O, finished products) |
| Kazakhstan | China (SS, raw materials) |
| Mongolia* | China (SS, raw materials) |

* The SS states that raw material (rhizomes) is "possibly" exported from Mongolia to China

China: International exports of *Rhodiola* products from China, especially extracts, are said to be a major driver of commercial trade (Cunningham et al., 2020). China is considered to be the major exporter of value-added *Rhodiola* extracts. According to a 2016 market study, nearly 75% of the world's *Rhodiola* extract volume was manufactured in China, 13% was made in Europe, 5% in the USA, and 7% elsewhere. One source stated most *R. rosea* extract produced in China is exported overseas with smaller amounts utilised domestically (Yi, 2022).

The largest populations of *R. rosea* subject to intensive commercial wild collection are reported to be in China (and Russian Federation) with the population most harvested in China in the Xinjiang Uyghur Autonomous Region. The Xinjiang region is an important trade and processing centre within China and the *R. rosea* from Xinjiang is reportedly considered one of the highest quality products (Cunningham et al., 2020). Both commercial demand and the location of extract manufacturers in China attract cross-border trade in raw *Rhodiola* rhizomes. Raw material that was not collected in China is principally thought to be first exported into the Xinjiang Uyghur Autonomous Region, mainly from harvesters operating in Russian Federation, including the Altai Republic (where the wild harvest is illegal) but also from Kazakhstan (where wild harvest is thought to be illegal, or restricted by quotas) and possibly Mongolia. An additional source stated that some areas in the Tibet Autonomous Region have banned harvest in *Rhodiola* due to consumption of resources, with "large quantities" of *R. rosea* (commodity not stated) imported from Pakistan, Russian Federation, and countries in Central Asia in the last five years to supply domestic demand (Yi, 2022).

From the Xinjiang Uyghur Autonomous Region in China, *R. rosea* is either sold directly to commercial enterprises as raw material or to extraction houses in eastern China for value-addition and export in the form of bulk extracts as well as end products. It was estimated that 500 t of dry rhizomes of *R. rosea* from 4–5 collection sites were sold from the Xinjiang region per year, with the majority sold to manufacturers in eastern China and subsequently sold internationally. Further harvest areas of *Rhodiola* in China are situated in the provinces of Heilongjiang and Jilin; *Rhodiola* specimens from these regions are classified as *R. sachalinensis* by the Flora of China, but Ohba (2003) treats it as a synonym of *R. rosea*.

A recent study into wild harvest and trade of *Rhodiola* in China pointed out that the growing demand for *Rhodiola* products is supplied from wild harvest and that there were "justifiable" concerns about its sustainable use in China (Cunningham et al., 2020). Another source confirmed that China is reliant upon wild supplies of *Rhodiola* to meet commercial demand, with the low profits from cultivation a barrier to farmers (Yi, 2022). In 2021, the Chief Scientific Officer of the American Botanical Council stated that based on discussions with companies in the *Rhodiola* trade, *R. rosea* had become scarce in parts of China (particularly the Xinjiang region) with harvesters being prompted to obtain materials in Russia, especially from the Altai region (Daniells, 2021). As *R. rosea* is included on the national protected plant list of China, permits are required for wild harvest (Zhang in litt., 2022). Despite this, there is some evidence that illegal wild harvest occurs; a recent report interviewed a *Rhodiola* supplier working with harvesters in the Xinjiang Uyghur Autonomous Region who stated that in their experience, harvesters do not obtain permits. Reasons cited were the expense of permits and the low likelihood of being caught harvesting without one (Kilham, 2021).

Based on export sales data from the e-commerce website www.alibaba.com, the principal international trade routes for end *Rhodiola* products from China appear to be the USA, the UK, Australia, Canada and the Republic of Korea, although exports have also been recorded to Pakistan and New Zealand.

Russian Federation: Russian Federation is a major user and exporter of *R. rosea* raw material (Brinckmann et al., 2021). In addition to China, the largest populations of *R. rosea* subject to intensive commercial wild collection are also reported to be in Russian Federation, with the population most harvested in Russia in the Altai Mountains of southern Siberia. A brief preliminary search for *R. rosea* products for sale online identified wild harvested roots from the Altai mountains in Siberia offered for sale on online platforms hosted in the USA and Germany.

The Russian Federation is also considered to be a major importer of *R. rosea* dried root, especially from wild collection operations situated in neighbouring countries of Kazakhstan and Mongolia.

It was estimated that around 85 t of dried *R. rosea* rhizome were exported from the Siberian Federal District (which includes the Republics of Altai and Khakassia, *where wild harvest is illegal*) between 2006 and 2008. This was noted to be four to five times more than the amount traded domestically inside Russian Federation. In 2006 it was noted that natural populations of *R. rosea* in the Altai area of South Siberia remained seriously threatened due to intense collection.

Seizures of illegally harvested *R. rosea* roots have been reported in Russian Federation, with reported quantities increasing from around 1.5 to 3.0 t annually during the 2000s to around 4.5 t seized in 2018 and over 8 t seized in 2019. *In 2022, 12 t of R. rosea were seized in the Altai Republic, said to be destined for a European country (Ren TV, 2022) and in 2021, over 27 t were seized, with the destination not clear (Administration of Maiminsky District, 2022).* Illegal harvest in protected areas has been documented in Russian Federation, including in a reserve in the Altai Republic.

Kazakhstan and Mongolia: Some raw *R. rosea* material not collected in China is principally thought to be imported from Kazakhstan, and possibly Mongolia, and both countries are reported to export wild-harvested raw materials to Russian Federation.

In 2019, a seizure of 42 bags of R. rosea was reported in eastern Kazakhstan, although the report stated that further identification was needed to confirm the plant was this species (Dyusengulova, 2019). The destination of the plant material was not stated. Illegal trade in *Rhodiola* has also been reported in eastern Kazakhstan near the border with Xinjiang and in the Almaty region, with illegal harvest in protected areas also reported in the country.

Other range States: Illegal trade in *Rhodiola* has also been reported in Kyrgyzstan, where harvest is reported to be restricted. Illegal harvest in protected areas has also been documented in Bulgaria, where collection from the wild is prohibited. *One source reported exports of R. rosea from Pakistan to China (Yi, 2022).*

R. crenulata

China: Less information is available regarding the key trade routes for *R. crenulata*, but the species is thought to be mainly harvested in the Qinghai-Xizang Plateau and Sichuan. The species is known to be misrepresented and traded as a substitute for *R. rosea*. *Rhodiola crenulata is the species most widely traded in China and the highest quality products from this species are considered to be those sourced from the Tibet Autonomous Region (Cunningham et al., 2020).*

Based on a study of *R. crenulata* at 23 locations in four provinces in China, it was considered that accessible populations had been “significantly reduced” due to destructive harvesting. This downward trend fits with the observations of three other studies, which noted a worsening situation for *R. crenulata* populations since the 1980s. Yan *et al.* (2003) reported that the economic value of *R. crenulata* had resulted “in heavy collections in recent years, which induced its limited distribution and fragile habitat”. Lei *et al.* (2006) considered that “*Rhodiola* natural resources have decreased remarkably recently, owing to overexploitation for medicine and shrinkage of their natural habitat”. Finally, Zhang *et al.* (2018) stated that “since the 1980s, the accelerated and uncontrolled use of *R. crenulata* in China has severely reduced its population”.

Bhutan: A recent study conducted in Lingshi (a local administrative unit in northeast Bhutan, on the border with China), which assessed priority species to determine whether wild harvest could be sustainable, found *R. crenulata* to be vulnerable and patchy in distribution, critically low in plant density (0.4 plants per m²), and among the rarest of 16 wild-collected medicinal plant species. *The authors of the study suggest some illegal cross-border trade in medicinal plants may be occurring, but do not specify R. crenulata in this context (Lakey and Dorji, 2016).* Lingshi is considered to have served as the main source of high-altitude medicinal plants for the Institute for Traditional Medicine Services in Bhutan for over 20 years, *which indicates that harvest may be primarily for domestic use.*

Nepal: A study published in 2021 combined primary and secondary sources to collate a list of plant species traded from Nepal to China; they included *R. crenulata* in their results based on secondary sources published in 2009 and 2013, but did not give details on the extent or trends of trade in this species (Chapagain *et al.*, 2021). Another study published in 2018 combined interviews with traders and wholesalers in Nepal and Tibet Autonomous Region with secondary sources from government documents including customs data to identify the most common medicinal and aromatic plant species traded from Nepal to China (Jun *et al.*, 2018). *Nepalese customs reported 17 species commonly traded with China but surveys along the border suggested that only seven species are commonly regionally traded, with results from Tibet Autonomous Region’s Commercial Bureau also showing that these seven species comprise over 90% of trade according to volume. R. crenulata was not mentioned in either source. Cunningham et al. (2020) mention trade in Rhodiola species from Nepal into China based on information on sourcing from TCM markets in Yunnan, Sichuan, and Qinghai but do not specifically mention which species this includes.*

India: No information could be found for evidence of wild commercial harvest or exports of *R. crenulata* in India but one source stated that *Rhodiola heterodonta* has been recently promoted as a “wonder herb”, with plans for the cultivation and propagation of this species in Ladakh (Rawat in litt., 2022). The source stated this has led to increased collection of *Rhodiola* rhizomes and plants from mountains in Ladakh on behalf of research institutes and industrialists with the aim of planting them at lower altitudes, with additional reports of individuals collecting *Rhodiola* from remote parts of Ladakh. Although the species involved were not confirmed as *R. crenulata*, this source stated that it was likely that different species of *Rhodiola* are mixed in India as in other regions.

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

Identification of individual *Rhodiola* species can be challenging even for live plants or dried rhizomes; it becomes increasingly difficult as the species undergoes processing and, at later stages, is not considered possible without molecular techniques such as DNA barcoding.

Mixing of different species is thought to occur at an early stage of the supply chain, thus presenting a challenge to enforcement if only a selection of species is included in the CITES Appendices. For example, research published in 2015 used DNA barcoding to show that in China, only 40% of 36 samples of products labelled as *R. crenulata* were authentic, with the remainder consisting of *R. serrata*, *R. rosea*, and three other species. Research published in 2016 additionally found that commercial samples labelled as containing *R. rosea* frequently contained other species, particularly *R. crenulata*, as well as *R. serrata*. In addition, the high rates of species interchangeability mean that there is a danger that harvesting pressure might readily shift to other species if only a subset is protected.

Additional Information

Threats

Rhodiola's large distribution means that it is not possible to make general statements regarding habitat trends for the genus. However, infrastructure development associated with tourism and the oil and gas industry, soil erosion, water abstraction and grazing have been highlighted as issues in discrete parts of *R. rosea*'s range. In addition, forest fires as well as anthropogenic threats such as illegal clear cutting are reported to have caused fragmentation and habitat loss in Russian Federation's Far East, negatively impacting wild medicinal plant populations, including those of *R. rosea*.

Climate change is considered to be a significant threat to some *Rhodiola* species, especially those limited to high-altitude habitats. In general, hotter, drier habitats are considered to be challenging for species such as *R. rosea*, which are dependent on relatively stable water supplies.

Conservation, management and legislation

Legislative measures to protect national populations of *R. rosea* and *R. crenulata* are in place in a number of range States.

A summary of national protection in the four range States in which there is the most evidence that commercial trade from wild-harvested R. rosea and R. crenulata occurs is presented in Table 3.

Based on the summary of national legal protection in Annex 5 of the Proposal, R. rosea is additionally fully protected in Bosnia and Herzegovina, and Bulgaria. In six further range States, four of which are in Europe, R. rosea has some form of protection, for example, permit requirements for harvest, quotas for harvest, or the protection of specific populations thought to be most at risk.

Table 3. A summary of legal national protection measures for the four range States with the most evidence for international commercial trade in wild-harvested *R. rosea* (all countries) or *R. crenulata* (China).

| Country | Legal national protection and/or regulations |
|------------|---|
| China | Both species are listed as protected wild plants in China. Ten species of <i>Rhodiola</i> need a permit for wild harvest. These include <i>R. rosea</i> and <i>R. crenulata</i> (Zhang in litt., 2022). |
| Kazakhstan | There were harvest quotas for <i>R. rosea</i> in certain areas in 2016 but recent harvest quotas are unclear. One source states that species (including <i>R. rosea</i>) included in the Red Book of Kazakhstan are automatically protected (Atomyime, 2022) but another source states <i>R. rosea</i> can be collected, although quotas may apply (Ridder City Portal, 2022). |

| | |
|--------------------|--|
| Mongolia | No information was given in the proponents' Supporting Statement or found during this research. |
| Russian Federation | <i>R. rosea</i> is nationally protected through the Red Data Book, with wild harvest for commercial purposes not allowed (Ivannikova in litt., 2022). Populations in Altai Territory, Krasnoyarsk Krai, Tuva Republic and Magadan Oblast are excluded from this protection, although regional bans or zero harvest quotas may be implemented in these populations (Ivannikova in litt., 2022). |

There are no known international instruments or controls in place that specifically relate to *Rhodiola* spp. either to protect or regulate the use of the species across international borders.

No management plans could be located for either of the two most traded species although *R. rosea* has been recorded in protected areas in Bulgaria, Croatia, Norway, Russian Federation, and China, and *R. crenulata* in Bhutan, China, and Nepal.

Artificial propagation

The proponents state that *R. rosea* is now being cultivated experimentally and sometimes commercially (albeit at a relatively small scale) in several countries including Bulgaria, Canada, China, Finland, Germany, Italy, Latvia, Lithuania, Moldova, Mongolia, Norway, Poland, Russian Federation, Slovenia, Sweden, Switzerland, the UK, the USA and possibly Kyrgyzstan.

Several other *Rhodiola* species are being cultivated experimentally in China, including *R. crenulata*, however these are not yet commercially available. *R. sachalinensis* (which is considered a synonym of *R. rosea* in this Proposal, following Ohba (2003)) is reportedly cultivated extensively as a field crop in northeast China. Cultivation of high-altitude *Rhodiola* species such as *R. crenulata* is also considered to be challenging due to its unique growing requirements. The current market for *Rhodiola* products in China is reported to be supplied entirely from wild-collection.

This research identified some additional evidence of commercial cultivation, although most of this was small-scale and for a domestic market:

Russian Federation

According to one expert, a large national company "Evalar" states online that it cultivates *R. rosea* on its own plantations on an industrial scale without the harvest of wild populations (Ivannikova in litt., 2022). A search on their online platform shows they have supplements containing *R. rosea* for sale and that products are available in ten countries in addition to Russian Federation, with some in Europe and Asia (Evalar, 2022).

A search for *Rhodiola* farms in Russian Federation found an organic farm in Khakassia that states they have cultivated 160,000 m² of land and planted *R. rosea* in addition to other Siberian herbs (Nature Siberica, 2022). The company also states they wild harvest *R. rosea*, with the map of harvest locations appearing to show wild harvest in Altai Krai (where commercial wild harvest is legal). This company sells its products in physical stores across Russian Federation and in Spain, Serbia, Montenegro, Bosnia and Herzegovina, and Hong Kong SAR and advertises several cosmetic products containing *R. rosea*.

Anecdotal evidence from interviews with companies selling *Rhodiola* products at an international trade fair indicates that several companies source cultivated *R. rosea* roots directly from Russian Federation. One company based in Latvia stated they sourced cultivated *R. rosea* from Russian Federation and another company based in the USA stated they sourced "wild cultivated" *R. rosea* from northern Russia (Timoshyna in litt., 2022).

In 2020, it was reported that the development of methods for industrial cultivation of *R. rosea* had begun in Russian Federation in 2019 with the participation of farms in Buryatia with support from scientific researchers and a grant-funded project (Pacific Russia, 2020). It is thought that despite some cultivation in Russian Federation, most specimens in exports likely originate from the wild, with cultivation efforts producing only small volumes, mainly for a domestic market, and volumes difficult to quantify (Ivannikova in litt., 2022).

USA

A company based in Alaska that sells *Rhodiola* products from a co-operative of farmers states that its *R. rosea* is cultivated in Alaska utilising organic methods (Alaska *Rhodiola*, 2022). The owner of the company stated in 2021 that they were seeing an increased demand for their *R. rosea* (Daniells, 2021). The online platform for this company only allowed shipments domestically. In 2015, it was stated that only around five acres of *R. rosea* have been planted in Alaska (Fessenden, 2015).

Canada

In Alberta, Canada, an *R. rosea* growers' organisation established in 2007 was said to have 140 shareholders in 2010, each a grower of *R. rosea* (Ampong-Nyarko, 2010). It was stated that the species was being cultivated in several locations, with at least 300,000 m² of *R. rosea* in production in various stages of growth (Ampong-Nyarko, 2010). In 2021, the manager of the co-operative stated that they were collaborating with growers in other parts of Canada to encourage the acceleration of commercial cultivation, with some increase in demand for sustainability cultivated *R. rosea* in North America, Europe, Australia, and parts of Asia (Daniells, 2021). A herbal company based in the UK and Germany stated in 2014 that they own and manage farms in Canada for cultivation of *R. rosea* utilised in their supplements (Schwabe Pharma, 2014).

Ukraine

The Deputy Chairman of the Association of Producers of Medicinal Raw Materials of Ukraine stated that *R. rosea* could grow "almost anywhere" in Ukraine, with wholesale prices per kg close to 10 times lower than those exported from China (Seeds, 2020). The extent to which commercial cultivation operations for international export were currently operating is not clear from this source.

One company based in Latvia stated they sourced cultivated *R. rosea* from Ukraine (Timoshyna in litt., 2022).

This research identified several barriers to large-scale commercial cultivation of *Rhodiola* species:

In 2021, the Chief Scientific Officer of the American Botanical Council stated that, based on discussions with companies in the *Rhodiola* trade, the costs of wild-harvested *Rhodiola* roots and rhizomes are "substantially lower" than cultivated, which deterred companies from purchasing cultivated varieties (Daniells, 2021). That labour costs relating to cultivation can act as a deterrent was reiterated by the manager of a *R. rosea* growers' organisation in the state of Alberta. One company in the USA cultivating the species also stated that factors including the plant needing five years to reach maturity, and questions regarding best agricultural practice, can deter farmers from choosing the crop due to concerns over financial profit (Daniells, 2021) and a report in China additionally stated that profits from cultivation of *R. rosea* were low and a deterrent to farmers (Yi, 2022).

Experts in the Ukraine and Canada also stated that it may take 4–5 years to achieve financial results from growth of *R. rosea* (Ampong-Nyarko, 2010; Seeds, 2020). This is because the desired chemical compounds—rosavins, salidroside and tyrosol—in *Rhodiola* roots reach their maximum when plants are of this age (Alaska *Rhodiola*, 2022, Ampong-Nyarko, 2010). A researcher in Canada stated that because of establishment and maintenance costs, yields of around 2,500 kg per hectare were needed for crops to become profitable (Ampong-Nyarko, 2010). Cultivation of *R. sachalinensis* (recognised as *R. rosea* by Ohba (2003)) has also been noted to be challenging due to low salidroside levels in cultivated plants, root rot, and leaf wilt. Cultivation of high-altitude *Rhodiola* species such as *R. crenulata* is also considered to be challenging due to its unique growing requirements (Lakey and Dorji, 2016).

Other comments

The proponents state that the suggested annotation would include commodities from *Rhodiola* that first appear in international trade, referring to raw material and semi-processed commodities such as extracts. Although the evidence indicates that in most range States, raw material and semi-processed commodities do appear first in trade, the available evidence presented above indicates an unknown proportion of exports from both Russian Federation and China is in finished products, with some of these likely produced from domestic wild harvest.

Final value-addition is not anticipated to increase in range States or to shift from importing countries to range States as a putative reaction of annotation #2, because in many instances production facilities would first have to be established.

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Inclusion of all African populations of African mahogany *Afzelia* spp. in Appendix II with annotation #17

Proponents: Benin, Côte d'Ivoire, European Union, Liberia, Senegal

Summary: *Afzelia* is a genus of African and South-East Asian trees in the family Leguminosae. The seven African species are an important source of high-quality timber traded internationally as *Afzelia*, Doussie, Pod Mahogany, or African Mahogany. Two species are found in the Zambezi region, one in the Sudanian region and four in the Guinea-Congo region. They are typically slow-growing and occur at low densities. There are not known to be introduced populations or plantations of these species outside Africa.

The timber of different *Afzelia* species is difficult to distinguish and is marketed under the same commercial names. It has a wide range of uses as decorative veneer, flooring, door frames, staircases, docks, boatbuilding, exterior millwork and construction, furniture, musical instruments, turned objects, inlays, and other small speciality wood items. Exports from Africa include logs and sawn timber. *Afzelia* are also important locally for a wide range of subsistence uses. They provide traditional medicine ingredients, livestock fodder, wood for construction, charcoal, and fuelwood.

Five species are currently identified as of major commercial importance: *Afzelia africana*, *A. bella*, *A. bipindensis*, *A. pachyloba*, and *A. quanzensis*. Despite the paucity of inventory data, a global population decline has been noted as a result of timber harvesting for international trade for *A. africana*. The population of *A. quanzensis* is suspected to be decreasing as this species is becoming locally threatened in some areas due to selective logging for its timber. The proposal is to include all African populations of *Afzelia africana*, *A. bipindensis*, *A. pachyloba*, and *A. quanzensis* in Appendix II owing to concerns about unsustainable trade, and, because of the similarity of appearance of their timber, all other African populations of the genus *Afzelia* (i.e., *A. bella*, *A. parviflora*, and *A. peturei*) in Appendix II, under lookalike criteria.

In general, there appears to be very limited data on the level of international trade in *Afzelia* spp. The Supporting Statement provides information on trade in African Mahogany noting that this may refer to *Khaya* spp. (also proposed for listing in CITES Appendix II, see CoP19 Prop.51) and other species in addition to *Afzelia* spp. *Afzelia africana*, *A. bipindensis*, *A. pachyloba*, and *A. quanzensis* are widespread species that are considered to be in high demand for international trade. Although global population data are not generally available for these species, significant population declines are noted at African, national and local levels. The rare species *A. peturei* is not thought to be in trade; its timber properties are unknown.

- ***Afzelia africana*** is widespread but considered Vulnerable (IUCN Red List, 2019) with intensive and unsustainable harvesting resulting in a population reduction of at least 30% over the past three generations (150 years). Threats are still ongoing. Intensive exploitation of this species for timber used in the international market is a significant threat. *Afzelia africana* is exported from Ghana where there has been no recent official inventory of the species. The national population is, however, suspected to be decreasing due to intensive annual fires in forest savanna ecotones of the country.
- Both ***Afzelia bipindensis*** and ***A. pachyloba*** were assessed as Vulnerable in 1998 due to population decline. IUCN Red List reassessments for these two species are in progress. *Afzelia bipindensis* and *A. pachyloba* have been reported as the most commonly traded African *Afzelia* spp. with Cameroon noted as the main African exporter of the genus. Côte d'Ivoire and Ghana are also major exporters. *Afzelia bipindensis*, *A. pachyloba* and an additional species, *A. bella*, are all industrially exploited in the Congo Basin.
- ***Afzelia quanzensis*** is considered to be locally threatened in various countries due to depletion from unsustainable and illegal logging but was assessed in 2019 on the IUCN Red List as Least Concern. In Mozambique it is one of the three main timbers harvested (by volume) and

one of five major timber species exported with China being the major export destination. It is one of the main species harvested and traded in Angola. *A. quanzensis* is considered to be a potential replacement timber for the Appendix II listed *Pterocarpus erinaceus*.

- ***Afzelia peturei*** is a restricted range species occurring in Democratic Republic of the Congo (DRC) and Zambia close to the border between the two countries. It is considered Vulnerable (IUCN Red List, 2019) due to its restricted range and human disturbance. It is not known to be in trade.
- The wood of ***A. parviflora*** (Least Concern, IUCN Red List, 2019) is harvested but it is unclear whether it is traded internationally.
- ***A. bella*** is also widespread and in international trade but there is not currently thought to be a significant population decline.

Analysis: African *Afzelia* spp. produce high quality timber that is valued in the international market for its durability and aesthetic appearance. Four of the seven currently recognised African species (*Afzelia africana*, *A. bipindensis*, *A. pachyloba*, and *A. quanzensis*) are widespread African trees that have been heavily harvested in at least parts of their range for their timber. There are reports of declining populations as a result of harvest in a number of different range States. As a result, three of these species have been classified as Vulnerable on the IUCN Red List (*A. africana* in 2019 and *A. bipindensis* and *A. pachyloba* in 1998). Harvesting and export has continued, which is likely to have led to further depletion and in some cases exhaustion of harvestable stocks. The fourth (*A. quanzensis*) was assessed as Least Concern in 2019 but is known to be widely harvested in at least one important range State (Mozambique). There are no known national population estimates or stock assessments for any of the species. There are strong indications that all four of these species are currently harvested unsustainably in sometimes large parts of their range increasing their vulnerability to other important threats, therefore meeting criteria for inclusion in Appendix II as set out in Criterion B of Annex 2a, Res. Conf. 9.24 (Rev. CoP17).

Furthermore, given that it is difficult to distinguish between the timber of different African *Afzelia* spp., the other African members of the genus would appear to meet the lookalike criteria for listing in Appendix II in Annex 2b of the Resolution.

Annotation

The timber of these species is mainly exported as logs and sawn timber by African countries to be processed elsewhere for a range of uses. Therefore, Annotation #17 which designates “Logs, sawn wood, veneer sheets, plywood and transformed wood”, appears to be appropriate as it includes those specimens that first appear in 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

Seven African species of *Afzelia* are currently recognised. *Afzelia africana*, *A. bella*, *A. bipindensis*, *A. pachyloba*, *A. parviflora*, *A. peturei* and *A. quanzensis*.

The genus *Afzelia* also occurs in South-East Asia, although for these species the taxonomy is more uncertain.

IUCN Global Category, Range and Population Trend

Table 1. IUCN Red List assessment, distribution range and population status of the seven African species of the genus *Afzelia*: *A. africana*, *A. bella*, *A. bipindensis*, *A. pachyloba*, *A. parviflora*, *A. peturei*, and *A. quanzensis* (Sources: SS and IUCN Red List assessments)

| Species | IUCN Global Category | Range | Extent of occurrence (km ²) | Population trend |
|-------------------------|--|--|---|------------------|
| <i>Afzelia africana</i> | Vulnerable A2cd (assessed 2019, ver 3.1) | Benin, Burkina Faso, Cameroon, Central African Republic, Chad, Congo, Democratic | 4,850,397 | Decreasing |

| Species | IUCN Global Category | Range | Extent of occurrence (km ²) | Population trend |
|----------------------------|---|---|---|------------------|
| | | Republic of the Congo; Côte d'Ivoire, Ghana, Guinea, Guinea-Bissau, Mali, Niger, Nigeria, Senegal, Sierra Leone, South Sudan, Sudan, Togo, Uganda | | |
| <i>Afzelia bella</i> | Least Concern (assessed 2019, ver 3.1) | Angola, Cameroon, Central African Republic, Congo, Democratic Republic of the Congo, Côte d'Ivoire, Equatorial Guinea, Gabon, Ghana, Guinea, Liberia, Nigeria, Togo | 5,014,355 | Stable |
| <i>Afzelia bipindensis</i> | Vulnerable A1cd (assessed 1998, ver 2.3) | Angola, Cameroon, Central African Republic, Congo, Democratic Republic of the Congo, Gabon, Nigeria, Uganda | - | Unspecified |
| <i>Afzelia pachyloba</i> | Vulnerable A1d (assessed 1998, ver 2.3) | Angola, Cameroon, Congo, Democratic Republic of the Congo, Gabon, Nigeria | - | Unspecified |
| <i>Afzelia parviflora</i> | Least Concern (assessed 2018, ver 3.1) | Angola, Côte d'Ivoire, Guinea, Liberia, Sierra Leone | 296,562 | Stable |
| <i>Afzelia peturei</i> | Vulnerable B2ab(iii) (assessed 2019, ver 3.1) | Democratic Republic of the Congo, Zambia | 378,492 | Decreasing |
| <i>Afzelia quanzensis</i> | Least Concern (assessed 2019, ver 3.1) | Angola, Botswana, Burundi, Democratic Republic of the Congo, Eswatini, Kenya, Malawi, Mozambique, Namibia, Somalia, South Africa, United Republic of Tanzania, Uganda, Zambia, Zimbabwe | 6,290,416 | Decreasing |

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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

Information available for each of the seven African *Afzelia* species:

Afzelia africana

This species is widespread in Africa, with an estimated extent of occurrence (EOO) of 4,850,397 km² (Table 1). The species occurs in Benin, Burkina Faso, Cameroon, Central African Republic, Chad, Congo, Côte d'Ivoire, DRC, Ghana, Guinea, Guinea-Bissau, Mali, Niger, Nigeria, Senegal, Sierra Leone, South Sudan, Sudan, Togo, and Uganda. As reported in the Supporting Statement, *Afzelia africana* populations were suspected to have undergone a decline of 30% over three generations (150 years) as a result of intensive and unsustainable harvesting, and the species showed a decreasing population trend as of 2020 due to ongoing harvest pressure. *A. africana* is considered endangered in Benin due to unsustainable harvest resulting in the decline of natural populations. In Burkina Faso, *A. africana* is considered nationally threatened due to overharvesting and is abundant only in protected areas and in sacred groves. It is not known for these two countries what proportion of the harvesting is for the export trade. *A. africana* was categorised as nationally vulnerable in Cameroon in 2011 due to exploitation for the international timber trade as well as deforestation for agriculture. In Sudan, *A. africana* was considered

“endangered” as a result of intensive and increasing exploitation for timber and fuelwood, encroachment of agricultural and urban areas, overgrazing, and the species’ intrinsic poor natural regeneration. *A. africana* is reported to be under “immense logging pressure” in South Sudan, with large trees selectively targeted for harvest; the rate of felling is reported to exceed the species’ rate of natural regeneration, and there is a risk of extirpation at a local level.

Afzelia africana is considered to be endangered in Uganda where the species has declined dramatically in West Nile in recent years because of charcoal use and carvings for the tourist market. It is now restricted to three locations with ongoing threats. However, it is still confirmed in Murchison where it is relatively protected (GlobalTree Portal 2022). In the last two years, Uganda has experienced rampant illegal harvest of the species in protected areas and private land mainly for timber export to Asia. This prompted a government ban on the harvest and trade in 2017, though this did not stop the export trade (Hills, in litt., 2022).

Afzelia africana is exported from Ghana where there has been no recent official inventory of the species. The national population is however suspected to be decreasing due to intensive annual fires in forest savanna ecotones of the country (Ofori, in litt., 2022).

Afzelia africana is considered to be a possible substitute species for *Pterocarpus erinaceus* (Treanor, 2022).

Afzelia bella

This species is widespread, with a large EOO (>5 million km²) spanning Angola, Cameroon, Central African Republic, Congo, Côte d’Ivoire, DRC, Equatorial Guinea, Gabon, Ghana, Guinea, Liberia, Nigeria, and Togo (Table 1). According to trade information in the Supporting Statement there was a significant volume of trade in 2003–2005 exported from Cameroon.

Afzelia bipindensis

This species occurs in Angola, Cameroon, Central African Republic, Congo, DRC, Gabon, Nigeria, and Uganda (Table 1). The CITES Management Authority (MA) of Zambia reports the species in Zambia. *A. bipindensis* was categorised as nationally vulnerable in Cameroon in 2011 due to heavy exploitation for the international timber trade. The species was listed as nationally endangered in DRC in 2014, as a result of unsustainable logging, deforestation caused by urbanisation and agriculture, and tree bark damage by Forest Elephants. Overexploitation of the species in DRC has led to a “continued and accelerated decline” considered “irreversible” due to illegal logging, poor forest governance, and slash and burn agriculture negatively impacting regeneration. The authors noted an increase in harvestable volume permitted by the DRC forestry administration of 8 m³ to 1,836 m³ over five years (assumed to be 2009–2014), with no accompanying conservation measures for the species.

Afzelia bipindensis is considered to be vulnerable in Uganda where it is rarely seen and is harvested for timber. There is likely to be a small population in Uganda with only three sites noted in WCS surveys (GlobalTree Portal 2022). It is also considered to be threatened in Zambia (GlobalTree Portal 2022).

Afzelia pachyloba

This species is native to Angola, Cameroon, Central African Republic, Congo, DRC, Gabon, and Nigeria (Table 1). Doucet (in litt., 2022) notes that *A. pachyloba* is not present in the DRC. The distribution given in Plants of the World Online (POWO) does not include DRC.

Afzelia pachyloba was categorised as nationally vulnerable in Cameroon in 2011 due to heavy exploitation for the international timber trade. In DRC, the species is restricted to the extreme southwestern region where harvest pressure for timber, subsistence uses and cabinet making is considered to be contributing to its local disappearance; however, the species was categorised as nationally least concern in 2014. Doucet (in litt., 2022) notes that *A. pachyloba* is not present in DRC.

In Angola, *Afzelia* spp. including *A. pachyloba* are considered “still in relative ecological balance, considering the volumes licensed annually [for harvest]”

Afzelia parviflora

This species is found in Angola, Côte d’Ivoire, Guinea, Liberia, and Sierra Leone, with an EOO of 296,562 km² and an area of occupancy (AOO) of 132 km² (Table 1).

Afzelia peturei

This species is restricted to DRC and Zambia (Table 1), with an estimated EOO of 378,492 km² and an estimated AOO of only 28 km² (Kamau et al., 2021). *Afzelia peturei* is not thought to be in trade but would be negatively impacted if included in mixed timber consignments.

Afzelia quanzensis

This species is widespread, occurring in Angola, Botswana (from the extreme northeast to the periphery of the Okavango Delta in the northwest), Burundi, DRC, Eswatini, Kenya, Malawi, Mozambique, Namibia, Somalia, South Africa, Uganda, United Republic of Tanzania (henceforth Tanzania), Zambia, and Zimbabwe, and with a large estimated EOO of >6 million km². Although widespread, the global population of *A. quanzensis* is suspected to be decreasing due to local declines from unsustainable and illegal harvest. *A. quanzensis* has also been reported to be locally threatened in Angola, Mozambique, and Somalia by timber logging and local use, and the species is listed in the national red lists of Malawi and Mozambique as vulnerable and near threatened respectively. The species' population in Malawi has declined over the past 30 years, from once being "conspicuous along riverbanks" to a level where trees can "hardly be found". The country's *A. quanzensis* population is considered to be threatened at present due to high demand for its timber and timber products. A 2012 assessment of Burundi's forest genetic resources determined that the species was nationally threatened, and mature trees are reported to be rare in Zambia. In eastern Tanzania *A. quanzensis* is considered at risk of becoming commercially extinct due to harvest pressure, with traders in some regions reportedly resorting to harvesting diseased and irregular shaped specimens for production of short planks. However, a study published in 2009 [exact fieldwork date unknown] noted that illegal logging of remaining *A. quanzensis* trees of >20 cm dbh (diameter at breast height) was still evident in Tanzania's north-eastern coastal woodlands. In Botswana, the species is "under excessive pressure" due to high demand for its wood, as well as subsistence harvest of leaves, bark and roots, forest fires and destructive foraging by elephants.

In Tanzania, a 60% decline in the population of A. quanzensis is estimated based on seed collection data between 2000–2020 with an estimated 75% decline in use and trade in 94 local timber yards surveyed (Mashimba, in litt., 2022).

In Zimbabwe, Zambia, and Mozambique A. quanzensis is of scattered occurrence, found throughout miombo woodland and also in drier areas. It is not scarce or even particularly uncommon. Sometimes a few individuals occur together in somewhat rocky areas with stands of more than five individuals rarely seen, unlike many other timber trees. Bigger, well-formed trees are often harvested for carving or higher quality timber if close to motorable tracks, especially in Mozambique. The timber is little used for general construction and not for firewood but is certainly sought after where available and accessible. Owing to its scattered distribution it is not easy to harvest systematically, so it tends to be cut down as individual trees are encountered (Timberlake in litt., 2022).

Afzelia quanzensis is considered to be threatened in Zimbabwe based on a SANBI 2001 assessment. The Fifth National Report to the Convention on Biodiversity (CBD) by Zambia notes that Afzelia quanzensis is locally threatened due to exploitation with mature trees rare as a consequence (Nott et al., 2020). Although Mozambique has more than 100 forest species with potential for wood production, A. quanzensis (chanfuta) is one of the five species that dominate the trade due to demand in the national and international markets and is one of three species that together account for more than half the volume of all timber harvested (the others being Pterocarpus angolensis and Millettia stuhlmannii). Most of the exports of the main timber species exported from Mozambique are logs or low value-added products. Afzelia quanzensis is one of the species that has been most affected by irregularities during the harvesting process, such as cutting trees that are smaller than the minimum permitted diameter, cutting outside of licensed areas and illegal cutting. Trees larger than the minimum permitted diameters have become difficult to find due to the pressure on logging, which incentivises cutting trees below these limits. Lack of information is also a problem since local tree fellers cut trees first and try to sell to those who show interest afterwards. Any wood not bought by foreign, often Chinese operators, is bought by local carpenters for furniture production, except curved or split wood, which is often abandoned in the forest. Forest and wildlife regulations state that 15% of the timber harvesting licence fee is intended for reforestation by the government. However, almost no replanting occurs in licensed areas. (MacQueen, 2018).

Afzelia quanzensis is one of the species exploited in Angola (Nott et al., 2020).

At the genus level, the three major *Afzelia* spp. exporting range States are Cameroon, Ghana, and Côte d'Ivoire. *Afzelia* spp. produce high-quality timber with properties comparable to *Tectona grandis* (Teak) and *Tieghemella* spp. The timber of *Afzelia* spp. is termite resistant, has a neutral pH, and is durable for use in permanent humid conditions, making it highly sought after on the international market for various construction and industrial uses including boat building and precision machinery. *Afzelia* wood is also prized for its aesthetics, and the species are also traded for use in furniture, flooring, veneer and musical instruments. As a result of their stability, durability and decorative appearance, *Afzelia* timber have high economic value and are considered one of "the very best timbers in the world".

Afzelia bipindensis and A. pachyloba are traded as Doussie, one of the 24 timbers currently harvested in Gabon according to the Timber Trade Portal.

The following table summarises data extracted from the Proposal on the volumes of legal trade recorded for *A. africana*, *A. bella*, *A. bipindensis*, *A. pachyloba*, and *A. quanzensis*.

Table 2. Annual and total volume of legal international trade of *A. africana*, *A. bella*, *A. bipindensis*, *A. pachyloba*, and *A. quanzensis* as reported by the Proposal. No data for *A. parviflora* and *A. peturei* are available. Extent of occurrence (EOO) is not known for all species (Source: SS).

| Species | Extent of occurrence / current IUCN Red List assessment | Purpose of trade | Total volume of legal trade (m ³) | Period | = Annual volume of trade (m ³) |
|-----------------------|---|--|---|-----------|--|
| <i>A. africana</i> | 4,850,397 km ² / VU (2019) | Construction or structural materials, horticulture, handicrafts, jewellery | 75,897 | 2003–2017 | 5,060 |
| <i>A. bella</i> | 5,014,355 km ² / LC (2019) | Construction or structural materials | 57,650 | 2003–2005 | 19,217 |
| <i>A. bipindensis</i> | VU (1998) | Unknown | 1,450 | 2019–2021 | 483 |
| <i>A. pachyloba</i> | VU (1998) | Unknown | 44,275 | 1991–2021 | 1,428 |
| <i>A. quanzensis</i> | 6,290,416 km ² / LC (2019) | Construction or structural materials | 4,425 | 2021 | 4,425 |

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

It is considered difficult to distinguish between the timber of different African *Afzelia* spp. using morphological characteristics and timber of different species is traded under the same trade names.

Additional information

Threats

Local subsistence harvest for fuelwood, charcoal, livestock fodder, and traditional medicine ingredients put additional pressure on *Afzelia* spp. Seedlings are susceptible to fire, browsing and drought. Additionally, numerous *Afzelia* spp. range States are undergoing significant deforestation.

Conservation, management and legislation

Legislative measures to protect national populations of *Afzelia* spp. are in place in a number of range States. *Cameroon, Gabon, Central African Republic, DRC, and Congo have all defined minimum cutting diameters for Afzelia species. Minimum recovery rates are also imposed (Doucet in litt., 2022).*

Artificial propagation

Preliminary results of mixed stand plantation trials in Côte d'Ivoire recommend mixed forest plantations of 13 timber species, including *A. africana*, for wood production and provision of other ecosystem services. In Sudan, *A. africana* is reportedly grown in nurseries and has been cultivated along rivers in several regions.

In Burkina Faso, agroforestry management practices for *A. africana*, such as assisted natural regeneration, seedling or sapling transplantation within farmlands, have not been practised. This might be due to the magic or taboo status associated with *A. africana* that poses major challenges to its domestication and sustainable conservation within the country (Balima et al., 2018).

Plantation trials of *A. quanzensis* were established in Mozambique in the early 1930s; after 60 years, plantation trees were found to have shorter trunks and lower branches than wild specimens. It was noted that plantation production of slow growing hardwood species such as *A. quanzensis* represented a poor economic return and does not reduce harvesting pressure on wild stocks.

Implementation challenges (including similar species)

The implementation of CITES for valuable African timbers has proved challenging and it has been suggested that substantial funds may be required to produce non-detriment findings (NDFs) (ATIBT Flash News, 2022).

The volume of re-exports of worked products of Afzelia produced in non-African countries could produce a significant implementation burden.

Trade in African mahogany may refer to both Khaya and Afzelia species, both of which are proposed to be included in Appendix II with Annotation #17 at CoP19. If one is listed and not the other, this could create challenges for customs officials.

Potential risk(s) of a listing

A potential risk of listing is to shift the burden of international trade to related species with desirable timbers including Asian Afzelia spp. The two genera Afzelia and Intsia are closely related and wood of the two can easily be confused: careful examination is needed to distinguish *I. bijuga* from Afzelia spp. timber. Similarly, expert knowledge and chemical tests are needed to differentiate between timber of the Asian *Intsia palembanica* and Afzelia spp. Neither *I. palembanica* nor *I. bijuga* naturally occur in any of the African Afzelia spp. range States; *I. bijuga* is native to Madagascar but has been planted in Tanzania.

Potential benefit(s) of listing for trade regulation

Forest concessions in the Congo Basin cover over 50 million ha. Most are managed under a management plan, which has proved to be the best tool for planning timber harvests, even if readjustments are needed. Mechanisms to verify that production is legal have been set up in recent years (notably through Voluntary Partnership Agreements (VPAs) and the EU Timber Regulation (EUTR), as have voluntary certification standards. A ban on export of logs for all the Central African countries is forthcoming. The States of the region are preparing to support in-depth transformation of the sector with a set of reforms. A different challenge is oversight of the timber sector in local markets, which accounts for a significant portion of timber harvest, but which jeopardises the sustainability of forest resources and produces no direct benefits for nation states. Formalisation of this sector will require adaptation of national regulatory frameworks and the development of transactions within a reorganised sector (Eba'a Atyi et al., 2022).

Efforts to improve sustainable forest management and promote certification such as those for the Congo Basin could potentially be reinforced by the CITES listing. Furthermore, it could establish necessary controls needed for more effective data gathering on trade and support regional collaboration in data sharing and transparency as recommended by Mahonghol et al. (2020).

Other comments

Afzelia is within the scope of certification schemes as the FSC Certificates Public Dashboard records certificates for the genus relating to Cameroon, Gabon, and Ghana.

The proposal notes that Afzelia spp. are nitrogen fixing legumes. This is not the case (Doucet in litt., 2022; Timberlake in litt., 2022).

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Deletion of North Indian Rosewood *Dalbergia sissoo* from Appendix II

Proponents: India, Nepal

Summary: North Indian Rosewood *Dalbergia sissoo* is a fast-growing perennial tree, native to Afghanistan, Bangladesh, Bhutan, India, Myanmar, Nepal, and Pakistan, and 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 provide resilience to this threat. In Bangladesh, India, Nepal, and Pakistan the species is widely cultivated and has also successfully naturalised following afforestation programmes. The species was assessed by IUCN as Least Concern in 2019.

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 used plantation tree species in the Indian subcontinent where it is economically important for its value in forestry, agroforestry, and horticulture.

The entire genus *Dalbergia*, apart from those species already included in Appendix I, was included in Appendix II at CoP17 (2016) with annotation #15. 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 reliably between the various species. At CoP18, an unsuccessful proposal (Prop. 51) was submitted to delete *Dalbergia sissoo* from the Appendices. India as one of the proponents raised particular concerns over the impact that the listing of *Dalbergia sissoo* had had on its handicraft industry. However, at CoP18 (2019), annotation #15 was amended to include an exemption for wood products under 500 g. It was believed that this might mitigate some of the impacts on the handicraft industry, although it is unclear if this has been the case.

India has had a reservation in place for the genus since 2017, as well as stricter domestic measures banning the export of all wild specimens of all species, with a few exceptions including trade in *Dalbergia sissoo*.

From 2017 to 2020, the predominant commodities of *D. sissoo* reported in direct CITES trade were wood products (~19.5 million kg, plus ~1.5 million items) and carvings (~6.3 million kg plus ~40,000 items), reported by importers. Most were reported as sourced from artificial propagation (74% of items reported by weight, and 80% of those reported by number), and the remainder were declared as from Pre-Convention and wild sources, with sources changing from primarily Pre-Convention in 2017 to artificial propagation. The majority of trade was reported as from India, and trade was stable between 2017 and 2020, with ~6.5 million kg reported as imported from India annually (India did not report this trade). Importers were primarily the EU, the USA, and the UK.

Many experts acknowledge that, without the use of technology, it is difficult for non-experts to readily identify *Dalbergia sissoo* once made into finished products, which 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.

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 artificially propagated populations. While the species does not meet the criteria for inclusion in Appendix II in Annex 2a of Res. Conf. 9.24 (Rev. CoP17), differentiating this species in trade from all other *Dalbergia* species does, at present, remain an implementation challenge. While methods exist to differentiate *D. sissoo* from other members of the genus in international trade, these require expertise and technology not currently available globally. The species therefore meets the criteria in Annex 2bA.

If this proposal were accepted *Dalbergia sissoo* would be the only *Dalbergia* species not included in the Appendices.

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 is no CITES Standard Reference for Dalbergia.

The CITES Dalbergia Checklist (Cowell et al. 2022) will be proposed as the CITES Standard Reference at CoP19 (CoP19 Doc. 84.1).

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*). The IUCN Red List assessment does not include Iraq in its range (Lakhey et al., 2020).

Introduced: Antigua and Barbuda, Australia, Benin, Brazil, Burkina Faso, Cameroon, Chad, China, Cyprus, Dominican Republic, Ethiopia, French Polynesia, Ghana, Guinea Bissau, Indonesia, Israel, Kenya, Malawi, Malaysia, Mauritius, Mozambique, New Caledonia, Niger, Nigeria, Oman, Pakistan, Panama, Paraguay, Philippines, Puerto Rico, Senegal, Sierra Leone, South Africa, Sri Lanka, Sudan, Thailand, Togo, Uganda, United Republic of Tanzania, Uganda, United States of America, Virgin Islands of the USA, Zambia, Zimbabwe.

IUCN Global Category

Least Concern (assessed 2019, criteria version 3.1)

Biological and trade criteria for retention in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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 a fast-growing tree 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, 2022). 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, 2013; 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). The global Extent of Occurrence (EOO) was estimated at 4,074,747 km² (Lakhey et al., 2020).

Within India the extent of occurrence is at least 198,974 km² 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.

Wild subpopulations in different parts of India comprise medium-sized, sometimes large trees, unevenly distributed, with a reported 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 monospecific plantations. In 2021 the number of *D. sissoo* trees outside of forests was estimated to be over 75 million trees in India, an increase of 1 million since 2017. Although 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. Harvest or trade primarily utilises cultivated trees, although wild exports have been reported (see below).

The Supporting Statement of CoP18 Prop. 51 reported 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. The Supporting Statement of the current CoP19 proposal additionally states that the price per cubic foot of wood

ranged between INR400–INR750 (~USD5–USD9), and that of logs ranged between INR800–INR4,500 (~USD10–USD56) according to online advertisements in May 2022⁹.

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 2021. There were significant discrepancies reported by exporting countries and importing countries, with importers reporting far more than exporters (see Table 1). India (the main reported exporter) has submitted all CITES annual reports for 2017–2020, and the main importers (EU Member States¹⁰ and the UK) have submitted annual reports for 2017–2020, except for the USA (at the time of writing 2019 and 2020 had not yet been received). According to LEMIS data, the USA imported 150 D. sissoo wood products and refused the import of a further 245 (242 of which were subsequently seized) originating from Pakistan and India, between 2017 and 2019. India has taken a reservation on *Dalbergia* spp. (since January 2017) and a Notification (2018/031) states that India 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*, explaining some of the discrepancies between importer and exporter reported data.

Table 1. Global CITES trade in *Dalbergia sissoo* 2017–2020 reported by importing and exporting countries, by trade term and source.

| Term | Source | Number of items | | Weight (kg) | |
|--------------|--------|-------------------|-------------------|-------------------|-------------------|
| | | Importer-reported | Exporter-reported | Importer-reported | Exporter-reported |
| Carvings | A | 37,139 | 199 | 1,100,903 | |
| | O | 3,048 | | 5,221,589 | |
| | W | 186 | | 3,949 | |
| Derivatives | A | 1,182 | | 444 | |
| | O | 1,080 | | 7,958 | |
| Timber | O | | | 33,152 | |
| Wood product | A | 1,164,898 | 114,292 | 17,959,203 | |
| | O | 89,711 | 236 | 1,497,947 | |
| | W | 199,462 | 110 | 83,223 | 14,896 |
| Total | | 1,496,706 | 114,837 | 25,908,368 | 14,896 |

Artificially propagated ("A"), Pre-Convention ("O"), and Wild-sourced ("W"; CITES Trade Database, 2022).

Nearly all trade reported by importers was sourced from artificial propagation (74% of items reported by weight, and 80% of those reported by number), and the remainder was from Pre-Convention and wild sources, with the source changing from primarily Pre-Convention in 2017 to artificial propagation. The majority of trade was reported as from India, and trade was stable between 2017 and 2020, with ~6.5 million kg reported as imported from India annually (India did not report this trade) and Pakistan and Egypt exporting low volumes. Virtually all trade was for commercial purposes (>99%). The main importers were the EU (predominantly France but also others including Poland and the Netherlands), the USA, and the UK.

In the years 2017 and 2018, 2,206 import permits for ~12 million kg of furniture made of *Dalbergia sissoo* were issued by the German Management Authority (in litt., 2019). In comparison, only ~5 million kg of other small wood products (most of them chess boards/pieces) were imported with 28 import permits over that period.

Between 2017 and 2020, seizures of *Dalbergia sissoo* were reported by the EU (including reports made by the UK prior to 2019), primarily because they were shipped without CITES permits, and totalling 29,738 items including wood products, carvings, and pieces of timber (EU-TWIX, 2022). The majority were reported as originating from India (96%), with the remaining from unknown origin, Nepal, Pakistan, and the USA (EU-TWIX, 2022). Further *Dalbergia* seizures recorded to genus level may also include *D. sissoo*. A total of 392 seizure events involving the genus *Dalbergia* were reported in WiTIS between 2006 and 2022.

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

⁹ Conversion INR to USD on 1st May 2022 was INR 0.012905 to USD 1, according to www.xe.com, accessed 22nd July 2022.

¹⁰ With the exception of Luxembourg for which no CITES annual report was available for 2020 at the time of writing.

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 Supporting Statement notes that living *Dalbergia sissoo* is easy to identify and is 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, 2022; Sivadas, in litt., 2019; Vlam and Zuidema, in litt., 2019; Ahmedullah, in litt., 2022).

Koch (in litt., 2022), 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 differentiation, particularly if the origin of the wood is unknown. *D. oliveri* was assessed as Critically Endangered (A2cd+3cd+4cd) in the IUCN Red List in 2020 (Barstow et al., 2022).

A range of techniques to enable the identification of *Dalbergia sissoo* in trade is available (Hartvig et al., 2015; Espinoza, in litt., 2019; Koch, in litt., 2019; Vlam and Zuidema, in litt., 2019; Ahmedullah, in litt., 2022; Sivadas, in litt., 2022):

Macro- and microscopic visual methods

Macroscopic visual identification methods using identification guides (such as CITESwoodID) can be utilised by non-experts to identify *Dalbergia sissoo* to genus level (Koch, in litt., 2022). 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., 2022). The only exception is for distinguishing between *D. sissoo* and *D. latifolia* when the origin country is known, for which 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. In addition, the increasing artificial propagation of *D. sissoo* outside of its native range, results in wood that resembles juvenile wood (for example, wider growth rings and lower wood density), and is therefore distinguishable as likely artificially propagated (Koch in litt., 2022).

Near-Infrared and Raman Spectroscopy

The Supporting Statement notes that the Indian Institute of Science, Bengaluru uses Raman Spectroscopy and the Institute of Wood Science and Technology (IWST), Bengaluru uses Near Infrared Techniques to differentiate various wood samples. The IWST has developed near-infrared and Raman Spectroscopy techniques to differentiate wood samples, however an extensive library of reference materials are needed from across the *Dalbergia sissoo* range to implement (Ahmedullah in litt., 2022). If developed, portable equipment (Raman Spectrometer) would aid customs agencies to identify wood samples (Ahmedullah, in litt., 2022).

DART TOFMS

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, in litt., 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 (USD250,000 to install and, in the US, USD250 to process each sample in 2019) may currently be a barrier to its implementation, and to date, uptake of the system by global enforcement agencies has been low (Espinoza, in litt., 2019), with only very few specialised laboratories operating the method for the purpose of customs identification (Koch, in litt., 2022). It is not widely known to be used in India (Ahmedullah, in litt., 2022).

DNA barcoding

DNA barcoding is another technology available for identification and has been demonstrated as capable of identifying *Dalbergia sissoo* to species level (Hartvig et al., 2015). However, to be used practically, this technique first requires the creation of reference datasets, or species-specific assays. DNA extracted from timber may also be of poor quality, which can hamper the process (Hartvig et al., 2015). This technology has also been used to

differentiate eight endangered *Dalbergia* timber species from China, South-East Asia, Africa and South America (He et al., 2018).

At the present time, therefore, a considerable gap remains between the potential and realised application of such methodologies (Dormontt et al., 2015).

Additional information

Threats

The main impacts on wild and cultivated populations of *Dalbergia sissoo* are fungal and bacterial diseases (root rot, wilt, 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, and older trees are more susceptible to the disease. 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 frequency of mortality due to diseases is also lower in wild subpopulations than in cultivated plantations. The species' high regeneration and growth rate, however, reduces their impact upon the species as a whole.

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 Supporting Statement also notes that within India, wild populations of *Dalbergia sissoo* are found within several protected areas where it is prohibited to remove trees, in accordance with the Wild Life (Protection) Act, 1972. 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.

Apart from the CITES Management Authority of India, the Export Promotion Council for Handicrafts (EPCH) is also authorised to issue CITES comparable documentation for export of *D. sissoo* specimens from India. EPCH has developed the "Vriksh standard Timber Legality Assessment and Verification Scheme". EPCH issues "Vriksh Shipment Certificates" for exporting goods containing *D. sissoo* by verifying various documents ensuring legal acquisition.

Artificial propagation

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 widely available and possible from almost all common practices such as: sowing seeds; planting stumps, root sections, and stem cuttings; cloning cuttings; and entire transplanting. Stump and root planting was said to be the most effective method of artificial regeneration. Artificial propagation in India is frequently carried out by the Forest Departments of almost all Indian States and Union Territories. Commercial plantations exist in both the area of natural distribution (Indian subcontinent), as well as in China and some African countries. A new source code "Y" was adopted at CoP18 for assisted production which would seem more appropriate for India's production systems (see Res. Conf. 11.11 (Rev. CoP18)).

Implementation challenges

Dalbergia sissoo is easy to identify as a whole tree with leaves (e.g. live), however the species is not commonly traded as live or as a whole organism (CITES Trade Database, 2022). In India, *Dalbergia sissoo* is primarily harvested for its timber which is used in the making of handicraft items, furniture, boats, carts, carriages, gun handles, rail-sleepers, cabinets, decorative veneer, ornamental turnery, plywood, musical instruments, skis, carvings, boats, tool-handles, floorings, etc.

Identification of timber from other species of *Dalbergia* requires analysis of anatomical features, genetic sequencing, or DART TOFMS, Near-Infrared and Raman Spectroscopy. It is unclear how many range States of *Dalbergia sissoo* have access to this technology and whether the capacity exists to implement at a wide scale. It is unclear whether artificially propagated and wild-sourced *D. sissoo* wood can be distinguished (Chen, in litt., 2022).

Potential risk(s) of a deletion

A lack of enforcement capability would leave open the possibility of other rosewood species being misdeclared 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 misdeclaring *D. retusa*, as the then unlisted and similar-looking, *D. bariensis* (EIA, 2016).

There was no mention in the Supporting Statement of whether wild-sourced trees are favoured over artificially propagated individuals, or vice versa, for their timber quality or other reasons. The deletion of the species from the appendices may have implications for wild populations (Chen, in litt., 2022).

Potential benefit(s) of deletion for trade regulation

The Supporting Statement notes that since the listing of *Dalbergia* in CITES Appendix II in 2016, the value of exported furniture and handicrafts has decreased from ~USD129 million per annum before listing to ~USD64–USD77 million after listing. Additionally, it was noted that the share of *D. sissoo* products in total woodware exports from India decreased from 25% (before the 2016 listing) to 7.29% after the listing (2021–2022). It was noted that this has affected the livelihoods of around 50,000 artisans working with the species, as well as other stakeholders involved in the supply chain.

It is possible that deleting *Dalbergia sissoo* from Appendix II would mitigate the negative impacts that this listing has reportedly had on some areas of international trade, particularly the negative impacts on the trade in wooden handicrafts and furniture 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 overexploitation (Jenkins et al., 2012). The Genus *Dalbergia* currently includes 269 accepted species according to Kew Plants of the World (POWO, 2022). The IUCN Red List has assessed 270 species, with 20 Critically Endangered, 55 Endangered, 55 Vulnerable, 18 Near Threatened, and the remainder Data Deficient (26) and Least Concern (96). One of the assessed species, *D. gloveri*, is not accepted by POWO.

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Inclusion of Cumaru *Dipteryx* spp. in Appendix II with new annotation designating logs, sawn wood, veneer sheets, plywood and transformed wood, and seeds

Proponents: Colombia, European Union, Panama

Summary: *Dipteryx* is a taxonomically complex genus encompassing 14 species of large, canopy emergent, slow growing trees distributed across Central and South America and occurring in tropical rainforest, seasonally dry forests, and woodlands. The genus is targeted for its valuable hardwood timber (traded as Cumaru, Shihuahuaco, and Brazilian Teak), as well as its seeds (known as tonka beans), which are traded internationally for use in the fragrance, tobacco, and food industries. In several range States *Dipteryx* spp. are also locally important for food, traditional medicine, charcoal, oil, as shade trees in cocoa agroforestry systems, and providing livelihoods to indigenous and local communities involved in the tonka bean supply chain. *Dipteryx* spp. face deforestation and habitat degradation throughout their global range, and logging adds to the pressure on wild populations.

The proposal is to include the species *Dipteryx alata*, *D. micrantha*, *D. odorata*, and *D. oleifera* in Appendix II of with Criterion B of Annex 2a of Resolution Conf. 9.24 (Rev. CoP17), and to include the remaining species of the genus *Dipteryx* in Appendix II of CITES for lookalike reasons, satisfying Criterion A of Annex 2b, with a new proposed annotation: “Logs, sawn wood, veneer sheets, plywood, transformed wood, and seeds” (current Annotation #17 with the addition of seeds).

- ***Dipteryx alata*:** Assessed as Vulnerable on the IUCN Red List in 2017 with a decreasing population trend, but no population estimate available. Occurs in Brazil, Paraguay, Plurinational State of Bolivia (henceforth Bolivia), and possibly Peru with an estimated extent of occurrence of seven million km². The species has undergone declines estimated between 30–50% over three generations in parts of its range. The seeds are harvested for tonka beans (see below).
- ***Dipteryx micrantha*:** Assessed as Data Deficient on the IUCN Red List in 2017 with a decreasing population trend. Occurs in Brazil, Ecuador, Peru, and possibly Bolivia, and Colombia. *D. micrantha* is reported to reach reproductive maturity at a minimum of 40 cm dbh (diameter at breast height) and live over 1,000 years. Considered in decline due to overharvest for timber, particularly in Peru where large volumes have been reported in exports: over 82 million kg (~ 76,000 m³) reported between 2018 and 2021 with 51 million kg (~7,000 m³) going to China, 19 million kg (~ 17,000 m³) to the EU and 1.8 million kg (~ 1,700 m³) to the USA. The population is estimated to have undergone a 33% decline in Peru between 2000 and 2020 with larger future declines projected.
- ***Dipteryx odorata*:** Assessed as Data Deficient on the IUCN Red List in 2017 with a decreasing population trend. Occurs in Bolivia, Bolivarian Republic of Venezuela (henceforth Venezuela), Colombia, French Guiana, Guyana, Honduras, Suriname, and possibly Peru. Introduced in Bahamas, Dominica, and Trinidad and Tobago. Nationally assessed as vulnerable in Colombia. The species is very slow growing and reaches maturity at 39 cm dbh. Timber harvest is believed to have major impacts on the species. High levels of timber trade are reported from Brazil and Colombia. The seeds harvested for tonka beans.
- ***Dipteryx oleifera*:** Least Concern on the IUCN Red List in 2020 and occurring in Colombia, Costa Rica, Nicaragua, Panama, Ecuador, and Honduras. Nationally assessed as vulnerable in Costa Rica, Colombia, and Panama. Currently listed in CITES Appendix III (under synonym *D. panamensis*) by Costa Rica (since 2003) and Nicaragua (since 2007), with a zero export quota for Nicaragua in 2022. The majority of trade since then has been reported by importers, originating from Panama (~ 51,000 kg, or ~ 47 m³), just under half of which was declared as from seized and/or confiscated sources.

Europe, the USA, and China are key importers of *Dipteryx* timber. Bolivia exported ~3.5 million kg of Cumaru to the European Union (EU) in 2019, and exports from Brazil to the USA and EU in 2018–2021 were around 11 million kg and 7 million kg, respectively. The genus *Dipteryx* comprised 80% of all wood exports from Peru in 2015. The timber of different *Dipteryx* species is not easily distinguished and is often traded under the genus, trade names, and common names comprising multiple species. The main products in trade appear to be logs, sawn wood, strips and joinery, decking.

Tonka beans are primarily harvested from *D. punctata*, *D. odorata*, and *D. alata*. *Dipteryx punctata* and *D. alata* have the shortest reported time to reach maturity of the *Dipteryx* species, estimated at 5–6 years. International trade in tonka beans boomed in the early 20th century, with intensive wild harvesting taking place, and declined in the 1940s. Current trade levels are reportedly a fraction of what they once were due to regulatory and voluntary restrictions on use of coumarin as an additive in the food and tobacco industries over recent decades. Presently, Brazil and Venezuela are the main range States supplying wild-sourced tonka beans to the international market for use in the perfumery and the food industry. Harvesting in Venezuela and Brazil is reportedly primarily carried out by indigenous communities, being an integral part of livelihoods and providing an alternative to involvement in extractive industries. Some experts note that the exploitation of tonka beans is driving conservation efforts for the species involved. Seeds are proposed for inclusion in Appendix II as a Precautionary measure because the impact of current tonka bean trade remains unknown. It has also been argued that, were onerous harvesting restrictions on seeds to be imposed, those who rely on the harvest for their livelihoods might turn to other, possibly destructive, uses of the parent trees, thereby having a negative impact on tree populations.

Several species of *Dipteryx* are traded under the common names Cumaru or Shihuahuaco and are neither distinguishable nor identified to species level in trade. It is not possible to distinguish the individual species within the genus *Dipteryx* using macroscopic or microscopic identification of wood anatomy, although identification of *D. alata*, *D. ferrea*, *D. micrantha*, *D. odorata*, and *D. punctata* using genetic markers is currently possible. Furthermore, *D. alata* and *D. odorata* are “commonly confused” in trade with *Handroanthus* spp., *Tabebuia* spp., and *Roseodendron* spp. traded as “Ipê” and the subject of Prop. 44.

Analysis: Trees in the genus *Dipteryx* are generally slow growing, and most species take a long time to mature (46–177 years), while *D. alata* and *D. punctata* have a faster age of maturity (5–6 years). Due to the slow growth of the main species in trade, the genus is particularly vulnerable to overexploitation, and the primary identified threat to *Dipteryx* is logging for timber. International trade in *Dipteryx* timber appears to be increasing. The seeds of *D. punctata*, *D. odorata*, and *D. alata* are also in trade primarily from Brazil and Venezuela as tonka beans, and it is unclear whether the harvest of seeds negatively impacts the species.

Dipteryx alata appears to meet Criterion B of Annex 2a of Res. Conf. 9.24 (Rev CoP17) on the basis of ongoing and historic decreases of over 30% over three generations driven by deforestation. While *D. odorata* and *D. micrantha* are both assessed as Data Deficient and do not have enough population information available to infer global population trends, they are both perceived to be decreasing and are assessed as threatened in parts of their range, with reported timber exports from Peru being a particular cause for concern, as well as very slow growth rates and age of maturity for both species. *D. oleifera* has been nationally assessed as vulnerable in three range States. Since its listing in Appendix III by Costa Rica (2003) and Nicaragua (2007) low levels of trade have been reported; however, given the generally poor reporting of trade in Appendix III listed species, this may not be an accurate representation of global levels of trade. Based on available information, it is unclear whether or not *D. odorata*, *D. micrantha*, and *D. oleifera* also meet Criterion B of Annex 2a. However, due to the difficulties of identification of timber and trade under the same name, these species meet Criterion A of Annex 2b.

While the remaining species also do not have sufficient population data to determine whether or not they meet the criteria for listing, due to significant taxonomic uncertainty, issues of timber identification, reporting of trade under generic and common names, uncertain distributions in range States, and nationally assessed threat levels, the genus appears to meet Criterion A of Annex 2b. *Dipteryx alata* and *D. odorata* are “commonly confused” with *Handroanthus* spp., *Tabebuia* spp., and

Roseodendron spp. (known as “Ipê”) which are proposed for listing in Proposal 44, therefore they would also meet the criteria in Annex 2bA were that proposal to be accepted.

Annotation

From available trade data the products most in trade from range States are timber and timber products and would be covered by the proposed new annotation or by #17. The overall impact of tonka bean harvesting remains unclear, with some considering its harvest contributes to the conservation and management of the species. Therefore, Annotation #17 without the addition of seeds may be more appropriate until strategies to mitigate potential negative impacts on livelihoods, and knock-on effects on forest cover, are developed in accordance with Res. Conf. 16.6 (Rev. CoP18) on *CITES and livelihoods*.

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 follows the taxonomy according to Carvalho *et al.* (2020), as outlined in CoP19 Doc. 84.2.

Fourteen species of the genus *Dipteryx* are currently recognised, following a nomenclatural review conducted in 2020 (Carvalho *et al.*; Table 1). However, there is considerable taxonomic uncertainty regarding *Dipteryx*.

In 2021, Carvalho *et al.* (2021) published a proposal to rename *D. oleifera* under its synonym “*Coumarouna panamensis* (*D. panamensis*)”, on the basis that *C. panamensis* and *D. panamensis* are more well-established names for the species and have been used in numerous national and regional floras, government publications and long-term ecological and socioeconomic studies. *Dipteryx panamensis* (*Dipteryx oleifera* according to the proposed standard reference) is currently listed in CITES Appendix III by Costa Rica (since 2003) and Nicaragua (since 2007), with a zero-export quota for Nicaragua in 2022.

Table 1. Taxonomy of *Dipteryx* according to various sources. An asterisk (*) indicates the species being proposed under Annex 2a Criterion B, dagger (†) indicates a species listed in CITES Appendix III.

| Species according to the proposed standard reference (<i>Dipteryx</i> , Carvalho <i>et al.</i> , 2020) | Kew Plants of the World Online (POWO, 2021) | Botanical Gardens Conservation International Global Tree Search (BCGI, 2021) | Systematic revision by Carvalho <i>et al.</i> (in prep.) | Pariente Mondragon (2018) |
|---|---|--|--|---------------------------------|
| <i>D. alata</i> Vogel (1837)* | | | | |
| <i>D. casiquiarensis</i> (Pittier) Lewis and Gasson (2000) | Not accepted | Not accepted | Synonym of <i>D. magnifica</i> | |
| <i>D. charapilla</i> (J. F. Macbr.) Ducke (1948) | | | | |
| <i>D. ferrea</i> (Ducke) Ducke (1940) | | Not accepted | | Synonym of <i>D. charapilla</i> |
| <i>D. lacunifera</i> Ducke (1948) | | | | |
| <i>D. magnifica</i> (Ducke) Ducke (1940) | | | | |
| <i>D. micrantha</i> Harms (1926)* | | | | Synonym of <i>D. alata</i> |
| <i>D. odorata</i> (Aubl.) Forsyth f. (1794)* | | | | |
| <i>D. oleifera</i> Benth. (1850)* † | | | | |
| <i>D. polyphylla</i> Huber (1913) | | | | |
| <i>D. punctata</i> (S.F.Blake) Amshoff (1939) | | | | |
| <i>D. rosea</i> Spruce ex Benth. (1860) | | | | Synonym of <i>D. charapilla</i> |
| <i>D. tetraphylla</i> Benth. (1860) | | Not accepted | Synonym of <i>D. odorata</i> | |
| <i>D. trifoliolata</i> (Ducke) Ducke (1940) | Not accepted | Not accepted | Synonym of <i>D. punctata</i> | |

Range and IUCN Global Category

Dipteryx spp. occur in Central and South America (Table 2). The presence of *Dipteryx* spp. in the Bahamas, Dominica, and Trinidad and Tobago as native or introduced populations remains uncertain, as well as anecdotal evidence of introduced populations or plantations in Nigeria and Kenya, evidenced through mentions of Nigeria as a major exporter of tonka. No evidence for these exports could be found, and the genus was not thought to occur in Kenya (Luke, in litt., 2022).

Table 2. Range and IUCN Global Category. “(?)” in the Range column indicates there is uncertainty. An asterisk (*) indicates the species being proposed under Annex 2a Criterion B, dagger (†) indicates a species listed in CITES Appendix III. IUCN Red List population trends: –: stable, ▼: declining and ?: unknown. EOO: extent of occurrence, AOO: area of occupation.

| Species | Range | IUCN Red List Assessment | National threat status | EOO (km ²) | AOO (km ²) |
|--------------------------|--|--|--|-------------------------------|------------------------|
| <i>D. alata</i> * | Brazil, Bolivia, Paraguay, Peru (?) <i>De Carvalho (in litt., 2022) noted that this species does not occur in Peru.</i> | Vulnerable (A2cd+3cd+4cd, assessed 2017, version 3.1) ▼ | Brazil: least concern (2012) <i>Peru: the species appears to meet the criteria for endangered based on an independent assessment (may refer to D. micrantha, Pariente Mondragon, 2018).</i> | 6,922,000 | |
| <i>D. casiquairensis</i> | Venezuela, (?) | | | | |
| <i>D. charapilla</i> | Brazil, Peru | Vulnerable (D2, assessed 1998, version 2.3) | <i>Peru: the species appears to meet the criteria for endangered based on an independent assessment (Pariente Mondragon, 2018).</i> | | |
| <i>D. ferrea</i> | Bolivia, Brazil (?), Peru | | An independent assessment suggested the species meets the IUCN Red List criteria for endangered in Peru. | | |
| <i>D. lacunifera</i> | Brazil | Least Concern (Assessed 2018, version 3.1) – | | | |
| <i>D. magnifica</i> | Brazil, Colombia, Venezuela, Ecuador (?), Peru (?), | | | | |
| <i>D. micrantha</i> * | Bolivia (?), Brazil, Colombia (?), Ecuador, Peru | Data Deficient (Assessed 2017, version 3.1) ▼ | An independent assessment suggested the species meets the IUCN Red List criteria for endangered in Peru. | 2,258,500 Peru: 22,516 | |

| Species | Range | IUCN Red List Assessment | National threat status | EOO (km ²) | AOO (km ²) |
|------------------------|--|--|--|-----------------------------------|------------------------|
| <i>D. odorata</i> * | Brazil, French Guiana, Guyana, Suriname, Venezuela Bolivia, Colombia, Honduras, Peru (?) Introduced: Bahamas, Dominica, Trinidad and Tobago <i>Pariente Mondragon (2018) notes that D. odorata does not occur in Peru</i> | Data Deficient (Assessed 2017, version 3.1) ▼ | Colombia: vulnerable | 9,670,000 | |
| <i>D. oleifera</i> *† | Colombia, Costa Rica, Nicaragua, Panama Ecuador, Honduras | Least Concern (Assessed 2020, version 3.1) ? | Colombia: vulnerable Costa Rica: vulnerable Panama: vulnerable | 541,000 Costa Rica: 10,180 | |
| <i>D. polyphylla</i> | Brazil, Colombia | Near Threatened (B2ab(iii), assessed 2020, version 3.1) ? | | 690,946 | 100 |
| <i>D. punctata</i> | Brazil, Colombia, French Guiana, Guyana, Suriname, Venezuela | Least Concern (Assessed 2018, version 3.1) – | | | |
| <i>D. rosea</i> | Brazil, Peru, Venezuela, Colombia <i>De Carvalho (in litt., 2022) noted that this species does not occur in Peru.</i> | | | | |
| <i>D. tetraphylla</i> | Brazil (POWO, 2022) | | | | |
| <i>D. trifoliolata</i> | Brazil, (?) | | | | |

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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 Supporting Statement notes that the market for timber and seeds is both national and international, but the international market for timber was noted to be expanding, in part due to increased scarcity and protection afforded to other Amazonian hardwoods, such as *Swietenia macrophylla* and *Cedrela odorata*. In addition, substantial illegalities have been identified in the timber industries in some of the species' range States, and regulation of trade was noted as necessary to reduce the increasingly detrimental pressure of harvest on the species' survival in the wild. Ensuring that trade is sustainable is critical for the survival of the genus and to maintain the important ecological role of the species as providers of food and habitat to a number of threatened species of fauna.

Dipteryx spp. are slow growing and long-lived, which makes them inherently vulnerable to overharvesting. There are no population data for any species of *Dipteryx* across their entire range. However, while there is little information regarding the current number of mature individuals, the growth of these species is very slow, and they generally occur at low densities. *Experts noted that in Brazil Dipteryx spp. are rare in forests, and that even in well-managed forests Dipteryx spp. did not recover well after logging (Menu, in litt., 2022).*

The Supporting Statement notes that Brazilian authorities reported that none of the *Dipteryx* spp. were threatened in the country as of 2022. *Dipteryx* spp. are threatened by targeted logging and in some regions also illegal harvest. The timber of several species is in great demand due to their dense, hard wood. Cumaru timber (*D. odorata*) has a reported average dried weight of 1,085 kg per m³ (The Wood Database, 2022). Cumaru decking has a market value of USD1,204–1,237/m³ in the USA and USD1,093–1,119/m³ in Asia, making it one of the most expensive timbers in the global market. Table 4 includes a summary of available information on *Dipteryx* spp. timber trade, and Table 5 on tonka bean trade.

WiTIS holds records on six seizure events and one enforcement action from 2016–2022. Three seizures involving *Dipteryx* were reported in Peru between 2019–2022. In 2019, an undisclosed amount of timber was seized due to a lack of documentation and led to two arrests; in 2021, just over 6 m³ of *D. odorata* timber valued at ~USD12,790 was seized due to invalid documents; and in 2022, a seizure of 11,250 kg of charcoal was made alongside an undisclosed amount of forest products including some originating from *D. micrantha*. Additionally, 19 members of an illegal logging ring in Ucayali, Peru were arrested in 2016, their main timber target being *D. micrantha*, with the main destinations identified as China, Mexico, and the USA. Prosecutors estimated that the gang exported 300 m³ per month over six years. In 2020, 8.68 m³ of *Dipteryx* timber was seized in Mexico due to a lack of proof of legal acquisition. In southeastern Pará, Brazil, illegal logging of *Dipteryx* was reported in 2019. In 2022, 26.66 m³ of *D. odorata* being transported in Acre, Brazil, without documentation were seized.

Dipteryx alata

Biological characteristics: One study reported a maturation time of six years and productive lifespan of 50 years for *D. alata* in agroforestry systems. Seeds are dispersed by mammals including bats and monkeys.

Population status: *Dipteryx alata* was reported to be “abundant and well preserved” in 2018 in eastern Bolivia, and to occupy approximately 72% of the Brazilian Cerrado ecoregion. The global population of *D. alata* was also suspected to have undergone a 30–50% population decline over the past three generations, mainly as a result of habitat conversion in the Cerrado and exploitation for timber. *Pariante Mondragon (2018)* reported that in Peru, the species meets the criteria for endangered status according to IUCN criteria.

Threats: *Dipteryx alata* is considered in decline due to previous and ongoing timber harvest. The global *D. alata* population has intrinsic low genetic diversity, resulting in less resilience to ongoing (modern) population fragmentation resulting from habitat loss, such as in the Brazilian Cerrado. Habitat fragmentation has been noted to hinder mammalian seed dispersal of *D. alata*. A study on *D. alata* gene flow in Brazil in 2014 found that deforestation had led to fragmentation of subpopulations within forest fragments and pastures, leading to isolation of individuals and higher levels of inbreeding than seen in higher density populations, potentially leading to inbreeding depression resulting in reduced seed germination and seedling survival rates.

Dipteryx micrantha

Biological characteristics: In the Madre de Dios region of Peru, an average annual growth rate of 2.77 mm over the first 100 rings/years and 0.86 mm after 300 rings/years was recorded for *D. micrantha*, indicating extremely slow growth. *Dipteryx micrantha* is reported to reach reproductive maturity at a minimum of 40 cm dbh and live over 1,000 years. Taken together, these measurements could indicate that the species’ maturation takes longer than 100 years.

Population status: The global population was assessed as Data Deficient and in decline in the IUCN Red List in 2017. An independent assessment suggested the species meets the IUCN Red List criteria for endangered in Peru, based on an estimate of 33% decline between 2000 and 2020 and a projected decline of 66% by 2036. Data from 356 permanent forest plots in primary forests across a number of locations in the Peruvian Amazon, covering a total of 165 ha, recorded a total of 66 individuals (equivalent to 0.19 individuals per ha), which is similar to the estimated 0.29 individuals per ha with a dbh of >51 cm in Madre de Dios and 0.2 individuals per ha in Ucayali; these are the two main areas of occurrence in the country and where many logging concessions are located. A higher population density of 0.71 individuals per ha was recorded in a forest conservation concession in Madre de Dios, and the species was found to be “abundant” at Cocha Cashu Biological Station (also in Madre de Dios; 1.75 individuals per ha), and at only slightly lower abundance in the surrounding region (1.25 individuals per ha). Individuals recorded in 2019 in the Madre de Dios forest conservation concession in the Las Piedras River basin, Peru, were all extremely old (the average dbh was 87.66 cm, meaning that the average age of the trees was estimated at 684.8 years), and levels of regeneration and recruitment were very low: only 0.06 juvenile trees (10–40 cm diameter) were found per ha and there was a total absence of saplings of 4–10 cm in diameter. The authors noted that the low numbers of seedlings and absence of saplings indicated that recruitment levels were insufficient for the species’ long-term survival in the Madre de Dios conservation concession.

Threats: The IUCN Red List assessment noted that selective logging poses a “major threat”, and the species’ intrinsic slow growth impedes regeneration after harvest, and considered deforestation and habitat degradation

to be major threats to the species. The Supporting Statement notes that the species was considered “high risk” species facing illegal and/or unsustainable harvest in Peru and Bolivia.

Dipteryx odorata

Biological characteristics: Reproductive maturity was reported at a minimum dbh of 39 cm. A maximum adult age of 1,200 years has been estimated for the species. Seeds are mainly dispersed by bats.

Population status: The global population was assessed as Data Deficient and in decline in the IUCN Red List in 2017. The species occurs in very low densities.

A study carried out in Brazil in 2012 in three forest inventories that practised reduced impact logging (RIL) noted that *D. odorata* displayed a very low density of adults (classed as trees of >45 cm dbh) of under 0.15 trees per ha in all three locations. Within Tapajós National Forest, Brazil, the density of large mature *D. odorata* trees (≥45 cm dbh) was also found to be low, at 0.12 individuals per ha.

The species was noted to exhibit an “inverted J-shaped” distribution throughout its range, with few adult trees in large size classes and higher numbers of juveniles. According to an inventory of *D. odorata* in Brazil where RIL was practised, all large-diameter (>90 cm dbh) trees had been extracted. The authors noted that the negative impact of RIL on reproduction and regeneration processes of the species may reduce the already relatively low density of saplings in future. Two studies noted the lack of smaller adult size classes (~15–45 cm dbh) in two localities in Brazil, attributed to the intensive seed collecting boom of the 1940s (see section on “Tonka bean trade” below) that affected recruitment. It was noted that unsustainable seed collection could negatively impact population viability.

Threats: The IUCN Red List assessment noted that selective logging poses a “major threat” and the species’ intrinsic slow growth impedes regeneration after harvest. The Supporting Statement notes that the species was considered “high risk” and facing illegal and/or unsustainable harvest in Peru and Bolivia.

Dipteryx oleifera

Biological characteristics: The genus is slow growing, with species taking an estimated average of 46–177 years to reach 30 cm in diameter, the size at which individuals of *D. oleifera* were observed to reach reproductive maturity and begin bearing fruit. A maximum adult age of 330 years has been estimated for the species. Seeds are dispersed by mammals, mainly bats, as well as agoutis and squirrels.

Population status: *Dipteryx oleifera* was assessed as globally Least Concern with an unknown population trend in the IUCN Red List in 2020. Globally, the species’ population was considered to have been reduced and fragmented by exploitation and clearance of forest for agriculture across its range, with the species considered a conservation priority. The species was nationally assessed as vulnerable in Colombia (2007), Costa Rica (2005), and Panama (2016). Approximately 40% of the Colombian population of *D. oleifera* was considered to have been “heavily exploited for timber.” In Costa Rica, the potential distribution of *D. oleifera*, based on known occurrence points, was estimated at approximately 10,180 km²; however, the species’ assessors noted that less than half of this area had remaining forest cover suitable for the species. *D. oleifera* was reported to occur at a mean density of approximately 1.08 trees per ha in central Panama.

Threats: Slash and burn agriculture was considered to threaten the survival in Nicaragua.

CITES trade summary: *Dipteryx oleifera* (according to the proposed standard reference: it was originally listed as *D. panamensis*) is currently listed in CITES Appendix III by Costa Rica (since 2003) and Nicaragua (since 2007), with a zero-export quota for Nicaragua in 2022.

Following its 2003 listing, CITES trade in *D. oleifera* from Costa Rica totalled 23 m³ of timber from artificially propagated sources exported to the USA for commercial purposes (Table 3). Trade from Nicaragua mainly comprised sawn wood and timber (364 m³) exported to Costa Rica, Cuba, and the USA (Table 3). Trade from Panama mainly comprised wild-sourced sawn wood, totalling 28,700 kg reported as imports by Germany and 22,746 kg by the USA (Table 3).

This may not be a complete picture of global trade in *D. oleifera* (*D. panamensis*), as some Parties report CITES Annual Reports based on permits issued and not all trade, therefore origin permits of trade from range States other than Costa Rica and Nicaragua in accordance with Article V of the Convention may not always be captured in the CITES Trade Database.

Table 3. All CITES trade in *Dipteryx oleifera* (reported under its synonym *D. panamensis*) since its listing in CITES Appendix III by Costa Rica (2003) and Nicaragua (2007: trade records were available from 2007–2020). All trade was direct and rounded up to the nearest whole number. All exporting Parties submitted annual reports for 2003–2020 (CITES Trade Database, 2022).

| Exporter | Importer | Term | Unit | Source | Purpose | Reported by | Total |
|------------|------------|--------------|----------------|--------|---------|-------------|--------|
| Costa Rica | USA | timber | m ³ | A | T | Exporter | 23 |
| | | | | | | Importer | |
| Nicaragua | Costa Rica | carvings | m ³ | W | T | Exporter | 19 |
| | | | | | | Importer | |
| | | sawn wood | m ³ | W | T | Exporter | 211 |
| | | | | | | Importer | 39 |
| | | timber | m ³ | W | T | Exporter | 95 |
| | | | | | | Importer | |
| | Cuba | sawn wood | m ³ | W | T | Exporter | 42 |
| | | | | | | Importer | |
| | USA | carvings | m ³ | I | T | Exporter | |
| | | | | | | Importer | 4 |
| | | sawn wood | m ³ | W | T | Exporter | 0.5 |
| | | | | | | Importer | |
| | | timber | m ³ | W | T | Exporter | 15 |
| | | | | | | Importer | |
| | | wood product | m ³ | W | T | Exporter | 2 |
| | | | | | | Importer | 2 |
| Panama | Germany | sawn wood | kg | W | T | Exporter | |
| | | | | | | Importer | 28,700 |
| | USA | sawn wood | kg | I | T | Exporter | |
| | | | | | | Importer | 22,746 |
| | | specimens | (blank) | W | S | Exporter | 50 |
| | | | | | | Importer | |

Dipteryx punctata

Biological characteristics: In Venezuela, the species reportedly takes five years to reach reproductive maturity (Santórum, in litt., 2022).

Population status: An expert estimated that there are hundreds of thousands of trees in the southern states of Venezuela (Amazonas and Bolívar; Santórum, in litt., 2022).

Dipteryx charapilla

Population status: Assessed as globally Vulnerable in the IUCN Red List in 1998, based on the species' limited distribution. The species is potentially endemic to Peru: its occurrence in Brazil is uncertain. *Pariente Mondragon* (2018) reported that in Peru the species meets the IUCN criteria for endangered.

Dipteryx ferrea

Population status: An independent assessment suggested the species meets the IUCN Red List criteria for critically endangered in Peru, based on declines as a result of logging.

Dipteryx polyphylla

Population status: Assessed as globally Near Threatened in the IUCN Red List in 2020, based on habitat loss driven by land use change. Timber exploitation was listed as a possible threat.

Table 4. Trade summary of *Dipteryx* timber reported at genus level and species level where available.

| Range State | Trade summary |
|-------------|---|
| Bolivia | <i>Dipteryx</i> spp. made up 26% of Bolivia's exports of wooden flooring by volume in 2019. Europe was considered the largest export destination with exports to the region increasing by 200% between 2016–2019. Bolivia exported over 3.5 million kg of <i>Dipteryx</i> to Europe in 2019, of which 90% was imported by France, Germany, the Netherlands, and Belgium. In 2019, approximately 35% of Bolivia's wooden flooring exports by volume did not specify species or trade names, therefore the country's recent <i>Dipteryx</i> export figures may be an underestimate. |
| Brazil | The USA and the EU imported 11 million kg and around 7 million kg of <i>Dipteryx</i> timber from Brazil respectively (reported as "Cumaru decking", "Brazilian Teak (Cumaru)", "yellow Cumaru lumber", and sometimes identified as <i>D. odorata</i>) between 2018–2021. The species exported by Brazil is likely to be <i>D. odorata</i> . <i>D. odorata</i> was the third-most exported species in Brazil between 2017 and 2022 (SERPRO, 2022). |
| Costa Rica | According to the CITES Trade Database, exports of <i>D. panamensis</i> (<i>D. oleifera</i>) as reported by Costa Rica from 2010–2019 consisted entirely of 22.9 m ³ of artificially propagated timber exported to the USA for commercial purposes. |
| Colombia | Colombia's National Environmental Licensing Authority (ANLA) approved the export of a total of 19,893 m ³ of <i>Dipteryx</i> timber from 2019–2020, of which 43% was reported to the genus level and 43% was reported as <i>D. oleifera</i> . Colombia's national reporting system indicates harvest of 98,696 m ³ of <i>Dipteryx</i> timber between 2019–2021; almost 80% of this volume was harvested in 2019. Of the total amount of timber harvested over this period, 75% was reported at the genus level, with <i>D. oleifera</i> and <i>D. odorata</i> accounting for 23,898 m ³ and 1,726 m ³ respectively. |
| Guyana | The Guyana Forestry Commission (2016) reported average prices in 2015 for Tonka Bean tree (possibly <i>D. odorata</i>) as USD162 and USD827 per m ³ of logs and dressed lumber, respectively. Export quantities (assumed to be measured in m ³ of logs, no units were given) of the species over 2009–2014 were reported to be low, ranging from 112 in 2015 to 952 in 2010. |
| Nicaragua | According to the CITES Trade Database, between 2010 and 2019 exports of <i>D. panamensis</i> (<i>D. oleifera</i>) as reported by Nicaragua principally comprised 254 m ³ wild-sourced sawn wood, exported for commercial purposes to Costa Rica and Cuba (Table 4). Smaller volumes of carvings, timber, and wood products were exported to Costa Rica, Cuba and the USA. |
| Panama | According to the CITES Trade Database, trade in of <i>D. panamensis</i> (<i>D. oleifera</i>) from Panama 2010–2019 principally comprised 27,800 kg sawn wood imported for commercial purposes in 2010, as reported by Germany only. Panama also reported exports of 50 specimens to the USA for scientific purposes (Table 4). |
| Peru | <p><i>Dipteryx</i> was reported to have undergone an extractive boom in Peru over the last decade. In 2006, <i>Dipteryx</i> spp. traded under the common name Shihuahuaco represented 50% of wood exports from Peru; by 2015 it was reported that this had risen to 80%. The majority of <i>Dipteryx</i> timber extracted from Peru is thought to be exported to China to provide raw material for the flooring market.</p> <p>According to MINAGRI-SERFOR forest yearbooks, 1,064,333 m³ of Peruvian <i>D. micrantha</i> wood was estimated to have been harvested from 2000–2020, equivalent to approximately 110,079 mature trees (>51 cm dbh), although it was noted that this may be an underestimate of harvest as the datasets used do not include <i>D. micrantha</i> felled for charcoal production and do not fully address illegal as well as legal logging volumes.</p> <p>Analysis of trade data of Peruvian exports of timber under the common names Shihuahuaco and Cumaru reveals that from 2015–2018 Peru exported over 101 million kg, with roughly 79 million kg going to China, 14 million kg to the EU and 1.3 million kg to the USA. Between 2018–2021 Peru exported over 82 million kg, with 51 million kg going to China, 19 million kg to the EU and 1.8 million kg to the USA.</p> <p>According to SERFOR, 247,395 logs of <i>Dipteryx</i> spp. were harvested as Cumaru in 2019, of which 64,698 m³ were exported as sawnwood, strips and joinery. For the same year, seven timber species accounted for more than 90% of the total harvested volume, all of them considered hardwood for the decking industry, of which 60% was reported as Cumaru (<i>Dipteryx</i> spp.).</p> |

| Range State | Trade summary |
|-------------|---|
| | <p>A 1,805 km² forest concession in Alto Ucayali, east-central Peru, which included <i>D. micrantha</i>, was managed using a 30 year cutting cycle and an average applied logging intensity of 12 m³ per ha per year.</p> <p><i>The only record of Dipteryx in the LEMIS database from 2008–2020 is one trade record of a wild-sourced scientific specimen of Dipteryx alata imported in 2014 from Peru via Mexico.</i></p> |

Tonka bean trade

Several *Dipteryx* spp. are also subject to seed collection for the national and international markets, *normally traded as dried seeds* (Santórum, in litt., 2022). Nationally they are used as a food source and internationally they are primarily used in the perfumery industry. *Dipteryx odorata* and *D. punctata* are two species whose seeds are traded internationally (De Carvalho, in litt., 2022).

Dipteryx spp. produce mature fruits supra-annually. Honorio Conrado (in litt., 2022) noted that mature fruit production occurs at intervals of 1–7 years. However, Santórum (in litt., 2022) noted that in Venezuela *D. punctata* trees bear fruit every two years. Average fruit production per tree has been reported to be 150 kg in *D. alata* and 10–20 kg in *D. odorata* (or possibly *D. punctata*). Average fruit production per tree was estimated at 20–30 kg for *D. punctata* in Venezuela, depending on the size and age of the tree (Santórum, in litt., 2022), and in Brazil one *D. odorata* tree was thought to produce fruits equivalent to 5 kg of dried tonka beans (Menu, in litt., 2022). Fallen *Dipteryx* fruits are gathered from beneath the trees, and must be gathered within five days of falling, as older fruits will rot and are not suitable for commercialisation (Guzman in litt., 2022). To obtain 1 kg of marketable tonka beans, 350–400 fruits are needed, or about 40 kg of unprocessed fruits for 2 kg of seeds (Santórum, in litt., 2022). Processed fruits (dried) yield 1 kg of tonka beans from 11 kg of fruits (Santórum, in litt., 2022). A study in the state of Pará in Brazil noted that ~80 kg of green seeds (fresh) yields ~40 kg of dried seeds (da Silva et al., 2010). Tonka beans can be stored up to five years when processed (dried) (Santórum, in litt., 2022).

Tonka bean trade boomed alongside the rubber trade as extractive industries were established in the Amazon in 1870–1960 (Torrealba and Scaramelli, 2018). In Venezuela, in the Orinoco and Cauca River basins, tonka bean trade was the principal commodity exported from 1870–1965, primarily to France, Germany, and the USA (Torrealba and Scaramelli, 2018). Collection was organised around “sarrapieras”, temporary stations in the forest from which collectors, mainly indigenous peoples and local communities, were based during the tonka bean collection season (Torrealba and Scaramelli, 2018). At its peak, Venezuela exported 784 t of tonka beans in 1942 (Santórum, in litt., 2022). From the 1940s, the demand for tonka beans declined due to the development of synthetic compounds, and in the 1950s, Venezuela centralised the tonka bean industry through the Instituto Agrario Nacional, and later in 1964 Venezuela prohibits the extraction of tonka beans through “sarrapieras” (Torrealba and Scaramelli, 2018). In 1954, the US Food and Drug Administration banned the use of tonka beans as a food additive due to their coumarin content (Guzman, in litt., 2022), and coumarin is also restricted as an additive in the EU since 2008. Later, in the 1960s, the extraction came to a halt in Venezuela as no markets for the beans could be found (Torrealba and Scaramelli, 2018). Tonka beans were important as a flavouring in the tobacco industry until the ingredient was largely removed from use between 1985 and 1997 (Guzman, in litt., 2022). Since the 80s, the level of tonka bean harvest has been at much lower levels (Torrealba and Scaramelli, 2018). Based on available data (Table 5), tonka bean trade appears to be fluctuating but not steadily increasing, with production from Brazil fluctuating between ~40 and ~100 t annually (Table 5).

Historical seed collection in combination with logging was considered to have caused major population declines in *D. alata*, according to the IUCN Red List assessment from 1998 (WCMC, 1998), possibly referring to the tonka bean boom of the early 20th century until the 1940s. Harvest of, and international trade in, seeds were not mentioned as threats in the most recent IUCN Red List assessment for *D. alata* dating 2017 (Requena Suarez, 2021). The Supporting Statement notes that harvest of *D. alata*, *D. odorata* and *D. punctata* seeds for the tonka bean trade has the potential to impact recruitment in wild populations, particularly as *Dipteryx* spp. produce mature fruits supra-annually, only a small percentage of mature trees may produce fruit each year, and there is a risk of synergistic pressure from both timber and seed harvest in some areas. Honorio Conrado (in litt., 2022) noted that in scarce years, harvesters may cover more ground to reach desired volumes of beans. De Carvalho (in litt., 2022) noted that more research is needed to evaluate the impact of seed collection.

Table 5. Summary on tonka bean trade from Brazil, Venezuela, and Bolivia.

| Range State | Trade summary |
|-------------|---|
| Brazil | <p>A study of the market for <i>D. odorata</i> seeds from Pará state, Brazil, reported that, in 2005, exports were mainly to Japan, France, Germany, and China. Pará state was noted to produce almost all of Brazil's tonka beans at the time of the study; seeds were reported to be harvested by ~2,700 rural families in the agricultural off-season from September–November, with each family harvesting ~80 kg of fresh seeds equivalent to ~40 kg of dried beans. <i>They were sold both as fresh seeds and as dried seeds (da Silva et al., 2010).</i> The trade in tonka beans in the region was observed to be somewhat dependent on the Brazil nut (<i>Bertholletia excelsa</i>) trade, as middlemen only purchased tonka beans together with Brazil nuts. Overall production in Pará for 2005 was estimated at ~108,000 kg of beans, of which 80–90% were purchased by only two companies in Belém; however, the beans are also used in Brazil's domestic perfume and cosmetics industry, making total export volumes unclear. <i>Menu (in litt., 2022) noted that the Brazilian market for tonka beans (known nationally as Brazilian vanilla) was a "very small part" of the global production.</i></p> <p><i>Between 1989 and 1996, tonka bean exports from Brazil peaked in 1994, with just under 120,000 kg exported, and declined to under 40,000 exported in 1996 (da Silva et al., 2010). Production of seeds in the state of Pará increased from 43,000 kg in 1997 to 110,000 kg in 2005 (da Silva et al., 2010). In 2020, Brazil produced a total of 117,000 kg of tonka beans, worth ~USD514 million¹¹ (Menu, in litt., 2022).</i></p> <p><i>Two programmes, Origens do Brasil and Florestas de Valor, commercialised 11,170 kg of dried tonka beans involving six different producing institutions representing 600 people living in 12 communities and 42 indigenous villages in a territory of 11,943,866 ha of Amazonian Forest (Menu, in litt., 2022). One tree can produce 5 kg of dried tonka beans, which can be sold for ~USD9.65¹² per kg, representing an income of ~USD42 per tree per year, compared to the value of a cut tree estimated at USD96 (Menu, in litt., 2022). This source of income was reported to deter people from cutting <i>Dipteryx</i> spp. (Menu, in litt., 2022). Origens do Brasil and Florestas de Valor were estimated to represent 15% of the Brazilian market for tonka beans, with the remainder sold by other communities, middlemen, as well as from privately owned land (Menu, in litt., 2022).</i></p> <p><i>According to Froes (in litt., 2022), tonka beans are traded as dried beans or in the form of an absolute, which is the raw material used in the fragrance production. Most of the absolute processors are in Europe and the USA, with few producers of absolute in Brazil, where most is exported in the form of dried beans (Froes, in litt., 2022). In Brazil, harvesters collect the fallen fruits near their home, walking no further than 5 km into the jungle for collection (Froes, in litt., 2022). The fruits are brought back to villages in baskets where they are opened manually with a hammer or a knife to release the bean, after which the beans are dried in the sun and in the shade and gathered together in big bags to be sold (Froes, in litt., 2022).</i></p> <p><i>According to Herrero-Jáuregui et al. (2012), "current intensities of <i>D. odorata</i> seed collection increases the density of saplings" however the development of saplings into young trees did not appear to occur in two plots of 25 ha in Brazil. The authors concluded that enrichment planting was needed to prevent local extinctions (Herrero-Jáuregui et al., 2012). No information on the level of extraction was provided and the authors noted that the study should be interpreted with caution due to low sample sizes (Herrero-Jáuregui et al., 2012).</i></p> |
| Venezuela | <p><i>Santórum (in litt., 2022) described a similar process as outlined above for tonka bean harvesting by indigenous communities in Venezuela. In Venezuela, <i>Dipteryx</i> mainly occurs in the Amazonas and Bolívar states (Santórum, in litt., 2022). Tonka bean collectors stop collection during the rainy season, even though trees will reportedly still have many harvestable fruits (Santórum, in litt., 2022).</i></p> |

¹¹ Based on an average exchange rate of BRL1 to USD0.1958 in 2020, for BRL2,625 million, (<https://www.exchangerates.org.uk/BRL-USD-spot-exchange-rates-history-2020.html#:~:text=Welcome%20to%20the%202020%20BRL,rate%20in%202020%3A%200.1958%20USD>).

¹² Based on an exchange rate of BRL1 to USD0.1929 on 19th August 2022 (<https://www.xe.com/currencyconverter/convert/?Amount=50&From=BRL&To=USD>).

| Range State | Trade summary |
|-------------|---|
| | <p>One company operating in Venezuela reported a minimum harvest of 40–60 tonnes of <i>D. punctata</i> annually from the Caura River basin (Santórum, in litt., 2022).</p> <p>While some experts noted that when trees do not produce a full crop of fruits harvesters go further afield, Guzman (in litt., 2022) noted that harvesters will not harvest fruits from trees that are not producing well in a given year, thus leaving some trees untouched every harvesting cycle. Additionally, Guzman (in litt., 2022) noted that only 5% of the area of occurrence is utilised for tonka bean extraction. Guzman (in litt., 2022) noted that tonka bean harvesting and trade does not threaten Venezuelan populations of <i>Dipteryx</i>, and that the harvesting promotes the conservation of the tree.</p> <p>Communities in some tonka bean-producing range States cultivate <i>Dipteryx</i> spp. stands for seed harvest, which may reduce the impact on wild populations. This was corroborated by Santórum, Guzman, and Quintero (in litt., 2022).</p> <p>Additionally, a small-scale community tonka bean harvesting and forest conservation project has been running in Venezuela since 2007.</p> |
| Bolivia | <p>In Bolivia, the seeds of <i>D. alata</i> (known as Almendra Chiquitana in Bolivia) are exported internationally, with harvest taking place in communities in the department of Santa Cruz. Demand for Almendra Chiquitana was reported to be growing. In 2018, the harvest of seeds was estimated at 9,000 kg from the Santa Cruz department for both national and international use.</p> |

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

B) Compelling other reasons to ensure that effective control of trade in currently listed species is achieved

The fruit, seeds and flowers of *Dipteryx* spp. are important food sources for numerous mammal, bird and insect species, including bats, parrots, agoutis, peccaries, deer, tapir, hummingbirds, and bees. *Dipteryx oleifera* is classified as a keystone species because its fruits support many species during a time of food scarcity for frugivorous animals. In addition, the broad buttresses of *Dipteryx* spp. make these trees ecologically important both for forest structural integrity while standing and for the large gaps created when they fall, facilitating successional forest growth.

Dipteryx alata, *D. micrantha*, *D. odorata* and *D. oleifera* provide nesting sites for threatened birds. Logging of *Dipteryx* spp. in Peru has reportedly resulted in the loss of nests and juveniles and a reduced availability of nesting sites for the nationally vulnerable and globally Near Threatened Harpy Eagle *Harpia harpyja*. Cavities in the trunks of *D. micrantha* trees are used as nest sites by the CITES Appendix I listed Scarlet Macaw *Ara macao* and other threatened macaw species such as the Red-and-Green Macaw *Ara chloroptera*. In addition, *D. micrantha* is an important nesting tree species for Peru's nationally vulnerable Crested Eagle *Morphnus guianensis*. Similarly, the decline in *D. oleifera* in Costa Rica was considered the cause of the decline in the Critically Endangered Great Green Macaw *Ara ambiguus*. In Nicaragua, *D. oleifera* was also noted to provide an estimated 80% of the diet of *A. ambiguus* and 90% of its nesting sites.

Additional information

Threats

The habitats of *Dipteryx* spp., the Amazonian rainforests of Brazil and Peru, the savannas of Brazil, Bolivia and Paraguay, and the rainforests of Colombia, Costa Rica, Nicaragua, and Panama, are increasingly threatened by deforestation and forest degradation, logging, land conversion to agriculture, and climate change.

The rainforests of Central America for example have been subject to significant clearance for fruit plantations and pastureland. Major forest and woodland habitats in Brazil, in which at least 11 of the 14 recognised *Dipteryx* spp. occur are in decline. The rate of destruction of the Amazon region had declined in Brazil between 2004–2012 but, as has been widely documented, has begun to increase sharply, particularly since 2019. In the Xingu basin in the state of Pará, for example, 196 trees were reportedly felled per minute in March and April 2021, a 40% increase over the same period in 2020. Seasonally dry forest, including the Caatinga shrub and thornforest ecoregion, is increasingly threatened by grazing, logging and fire.

The savannas of Brazil, Bolivia, and Paraguay, including the Cerrado ecoregion in which *D. alata* occurs, are also under threat. Around 67% of the Cerrado has been already either completely converted for agriculture or modified in a major way, with 19.8% remaining undisturbed as of 2017 (Strassburg et al., 2017).

Many of the range States of *Dipteryx* spp. have recently experienced substantial deforestation and forest degradation. The FAO's Global Forest Resources Assessment (FRA) for 2020 included three *Dipteryx* spp. range States (Brazil, Bolivia, and Paraguay) amongst the ten countries with the highest average annual net loss of forest area over the period 2010–2020, with an annual net loss of 0.30%, 0.43% and 1.93%, respectively. The assessment found that Brazil had the highest annual net loss of forest area of any country assessed, with an average net loss of 1,496 ha per year. Moreover, Brazil accounted for approximately 7% of global wood removals in 2018, the fourth highest percentage of any individual country, and annual deforestation rates in the Amazon increased significantly in 2016 to 3.9 million ha per year. The Cerrado biome in Brazil is experiencing similar declines, with a net loss of 9,520 km² between 2000 and 2015 and an annual net loss of 1.2% per year due to land conversion. Tropical moist forest degradation is also particularly prevalent in Nicaragua, where 65.8% of forests have been degraded between 1990 and 2019, the second highest proportion of forest loss of previously undisturbed forest (that is, forest unaffected by deforestation or degradation) in the Americas. Over the 30-year period 1990–2019, Brazil, Bolivia, Colombia, Peru, and Venezuela were found to have undergone total declines in the area of undisturbed tropical moist forests of 24.9%, 34%, 21.6%, 11.8%, and 18.6%, respectively. Additionally, disturbance rates (that is, natural and anthropogenic degradation or deforestation) in Colombia, Venezuela, Nicaragua, and Ecuador have increased significantly from 2000–2014 compared to 1990–1999, by 0.23 million ha per year, 0.17 million ha per year, 0.08 million ha per year, and 0.09 million ha per year, respectively.

Conservation, management and legislation

Table 6 summarises the available information on the national legislation in some of the range States of *Dipteryx*.

Table 6. Summary of national legislation in *Dipteryx* range States, including any MED (minimum exploitable diameter) when applicable.

| Range State | National legislation summary |
|-------------|---|
| Bolivia | Export of logs is banned under Article 8 of Supreme Decree No. 24453 of 21st December 1996, which regulates the implementation of the Forestry Act No. 1700 of 12th July 1996. |
| Brazil | Under Normative Instruction No. 15 of 5th December 2011 (amended by Normative Instruction No. 13 of 24th April 2018 to specify that exports of charcoal of native species require authorisation from IBAMA, the export of roundwood of all native species from natural forests, with the exception of <i>Minquartia guianensis</i> , is prohibited. No <i>Dipteryx</i> spp. were included in the country's 2014 list of endangered flora. The current MED for species of <i>Dipteryx</i> is 50 cm. However, a modelling study of the <i>D. odorata</i> population in Brazil suggested that the MED should be increased to 100 cm, with cutting cycles of 30 years, in order to achieve sustainable timber harvest. |
| Costa Rica | <i>D. oleifera</i> was listed in CITES Appendix III by Costa Rica in 2003 under its synonym <i>D. panamensis</i> , due to concerns about overexploitation for the timber trade. A ruling of Costa Rica's Constitutional Chamber banned the exploitation of <i>D. oleifera</i> from the wild in 2008 (including standing wood, naturally fallen wood, and residual wood), noting that the ban should remain in place until the tree species itself as well as the Great Green macaw, <i>Ara ambiguus</i> (for which <i>D. oleifera</i> provides nest sites and is a critical food source) remain on the list of threatened species. However, it is unclear whether the ban was established via legislation, or via an executive decree. Prior to this, Ministerial Decree No. 25167 of 12th June 1996 restricted the harvest of <i>D. oleifera</i> in the north of the country (between the San Carlos, San Juan, and Sarapiquí rivers) to protect <i>A. ambiguus</i> nest trees, and a compensation payment scheme was established for owners of isolated trees of <i>D. oleifera</i> and forests containing the species within the area covered by the Decree, to encourage conservation. The export of logs and squared timber was additionally prohibited by Forest Law No. 7575 of 16th April 1996; however, it is unclear whether timber from plantations is exempt from this export ban. |
| Colombia | Colombia has reportedly had an export ban on roundwood from natural forests since 1997. |
| Ecuador | Under Article 46 of the Law on Forests and Conservation of Natural Areas and Wildlife of 10th September 2004, the export of roundwood, unless authorised by the Ministry of Environment for scientific purposes, is prohibited. |
| Guyana | Guyana agreed FLEGT Voluntary Partnership Agreements (VPAs) with the EU to ensure that timber and timber products exported to the EU are legally-sourced, although the EU-Guyana VPA has yet to be signed as of 2020. Under the Forest Regulations of 1st January 1953, the MED for all tree species unless otherwise specified is 60.96 cm. |

| Range State | National legislation summary |
|-------------|---|
| Honduras | <p>Honduras agreed FLEGT Voluntary Partnership Agreements (VPAs) with the EU to ensure that timber and timber products exported to the EU are legally-sourced. The EU-Honduras VPA was signed in February 2021.</p> <p>Export of unprocessed roundwood or squared timber of broadleaved species from natural forests is prohibited under Article 102 of Decree No. 98-2007 of 26th February 2008.</p> |
| Nicaragua | <p><i>Dipteryx oleifera</i> was listed in CITES Appendix III by Nicaragua in 2007, under its synonym <i>D. panamensis</i>, due to concerns about overexploitation for the timber trade.</p> <p><i>Dipteryx oleifera</i> (under the synonym <i>D. panamensis</i>) was added to the list of species with indefinite national harvest bans by Ministerial Resolution No. 29/06 of 16th June 2006. Additionally, the export of roundwood, timber and sawn timber of “any forest species” from natural forests was prohibited by Law No. 585 of 21st June 2006.</p> |
| Panama | <p>Under Decree No. 107 of 19th January 2021, export of logs from natural forests or reservoirs that have not undergone primary processing or treatment (against insects and fungi) is prohibited; exported timber must also originate from areas with sustainable forest management plans endorsed by the Ministry of Environment. Previously, export of unprocessed timber from natural forests was restricted by Decree No. 83 of 6th August 2008, but this was repealed in January 2016 by Decree No. 7 as the restrictions were considered to have been “ineffective” at stimulating wood processing mechanisms within the national timber industry.</p> <p>Resolution No. 5 of 22nd January 1998 specifies that logging permit holders must record volumes felled and pay the costs of surveys, inspections and technical services for all trees ≥ 20 cm in diameter.</p> |
| Paraguay | Under Decree No. 24498/72 of 18th February 1972, the export of roundwood, logs and beams is prohibited. |
| Peru | <p>According to TRAFFIC’s 2014 briefing document on Peru, a ban on the export of logs from natural forests has been in place since 1972 (TRAFFIC, 2014). However, the original legislation could not be found to verify this.</p> <p>According to the World Resources Institute’s 2014 Peru country profile, forest concession agreements require reduced impact logging practices, cutting rotations of at least 20 years, and the retention of a minimum of 10% of mature adult trees (seed trees) of each harvested species to enable regeneration.</p> |
| Suriname | <p>Under the Forest Management Law of 18th September 1992, <i>D. odorata</i> and <i>D. punctata</i> are listed as Category C species, making felling these species illegal unless specifically approved by the Forestry Department. Additionally, a permit is required for the export of “raw wood, round wood, round or felled pole wood, processed wood, wood products and forest by-products”.</p> <p><i>Dipteryx odorata</i> and <i>D. punctata</i> are protected under the Forest Management Law of 18th September 1992. All other marketable or potentially marketable tree species have MEDs of 35 cm.</p> |
| Venezuela | <p><i>In Venezuela, D. punctata</i> has been protected since 1952, when it became the official tree of Bolívar state, meaning the felling of the tree became prohibited (Santórum, in litt., 2022).</p> <p>Reportedly, forest concession holders may only extract trees greater than 40 cm dbh (not specific to <i>Dipteryx</i> spp.)</p> |

Artificial propagation

Dipteryx oleifera timber plantations have been established in Panama and Costa Rica. In Costa Rica, an experimental plot planted with 49 *D. oleifera* individuals in 1985 was reported to have a 14% survival rate after 24 years, and the surviving trees were “straight and of good form”. Planting schemes for *Dipteryx* spp. in Costa Rica have reportedly successfully produced fruiting trees. In pure stands of *D. oleifera* within a plantation at La Selva Biological Station, Costa Rica, rotation periods of 25 and 32 years were estimated for thinned and unthinned stands, respectively. *Dipteryx* spp. plantations have also been established in Trinidad and Tobago, and possibly also Jamaica, for seed collection and as shade trees for cocoa. Plantations have also reportedly been established in the Experimental Annex von Humboldt in Ucayali, in Peru. *Dipteryx* spp. (*D. odorata* and/or *D. punctata*) are cultivated and wild seedlings are supported by local communities in Venezuela and Brazil (Froes, in litt., 2022), to ensure future tonka bean harvest and for conservation of the species. Communities in some

tonka bean-producing range States cultivate *Dipteryx* spp. stands for seed harvest, which may reduce the impact on wild populations. This was corroborated by Santórum, Guzman, and Quintero (in litt., 2022).

Anecdotal evidence of introduced populations or plantations in Nigeria and Kenya were found in online advertisements for tonka beans, through mentions of Nigeria as a major exporter of tonka. No evidence for these exports could be found, and the genus was not thought to occur in Kenya (Luke, in litt., 2022).

Implementation challenges (including similar species)

According to the Thünen Institute of Wood Research in Germany, it is not possible to distinguish clearly the individual species within the genus *Dipteryx* using macroscopic and microscopic identification of wood anatomy. Identification of *D. alata*, *D. ferrea*, *D. micrantha*, *D. odorata*, and *D. punctata* using genetic markers is currently possible. Samples of *Dipteryx* spp. will reportedly be included in the database of species identifiable in the field using the open source, field deployable XyloTron platform.

Several species of *Dipteryx* are traded under the common names Cumaru or Shihuahuaco and are neither distinguishable nor identified to species level in trade. For example, *D. punctata* is less known in international markets but is generally misidentified as *D. odorata*. According to Koch (in litt., 2019), *D. alata* and *D. odorata* are “commonly confused” with *Handroanthus* spp., *Tabebuia* spp., and *Roseodendron* spp. (known as “Ipê”) in trade, although it is possible to differentiate between *Dipteryx* and Ipê timber based on microscopic wood characteristics. CoP19 Prop. 44 is proposing the listing of Ipê (*Handroanthus* spp., *Tabebuia* spp., and *Roseodendron* spp.) in CITES Appendix II.

The Supporting Statement notes that the confusion regarding the taxonomy and distribution of some *Dipteryx* spp., such as *D. charapilla* and *D. odorata* in Peru, compounded by the difficulty in differentiating the timber to species level, has led to some species being mistakenly traded as others. For example, the principal species that are exported from Peru are unclear. *D. odorata* had previously been considered a major timber species in Peru and timber has been exported from Peru under numerous names including *D. odorata*, as well as synonyms *Coumarouna odorata*, *C. micrantha*, and the trade names Cumaru and Shihuahuaco. However, recent taxonomic studies indicate that *D. odorata* does not occur in Peru, and studies have concluded that the timber harvested in the south of the Peruvian Amazon described as *D. micrantha* or *D. odorata* is in fact *D. ferrea*; while timber extracted in the north is considered *D. micrantha*.

No identification manuals are available for the seeds (tonka beans).

Potential risk(s) of a listing

Experts expressed concern that with Appendix II listing of *Dipteryx* seeds, harvest and sale of tonka beans may be halted for several years while CITES listing is implemented nationally, and therefore may drive communities currently harvesting the beans to other sources of income such as logging and agriculture (Froes, in litt., 2022; Guzman, in litt., 2022; Menu, in litt., 2022; Santórum, in litt., 2022). As the impact of tonka bean harvesting remains unclear, Annotation #17 without the addition of seeds may be more appropriate until strategies to mitigate potential negative impacts on livelihoods, and knock-on effects on forest cover, are developed in accordance with Conf. 16.6 (Rev. CoP18).

Menu (in litt., 2022) noted that listing in Appendix II may stimulate illegal logging as legal avenues may have a deterrent effect and could create barriers for the development of sustainable forest management in Brazil.

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Transfer of *Paubrasilia echinata* from Appendix II to Appendix I with annotation: All parts, derivatives and finished products, including bows of musical instruments, except musical instruments and their parts, composing travelling orchestras, and solo musicians carrying musical passports in accordance with Res. Conf. 16.8.

Proponent: Brazil

Summary: *Paubrasilia echinata*, commonly known as Pau-brasil, Pernambuco or Brazilwood, is a slow-growing leguminous tree, reaching around 15 m in height with a maximum trunk diameter of around 70 cm. It is endemic to the Mata Atlântica (Atlantic Coastal Forest) in Brazil, which is ranked fourth in the list of global biodiversity hotspots. Many aspects of the biology of Pau-brasil and the composition and structure of the plant community in which it occurs are poorly known. The wood of Pau-brasil is currently used worldwide for the manufacture of high-quality bows for musical instruments, for which the species has been exploited for over 200 years. *P. echinata* was classified as Endangered on the IUCN Red List in 1998 and has been assessed as endangered on the list of Brazilian flora threatened with extinction since 1992. The species was originally listed as *Caesalpinia echinata* in Appendix II at CoP14 (2007) with Annotation #10; that name became a synonym of *Paubrasilia echinata* in 2019, following taxonomic changes adopted at CoP18. Annotation #10 includes logs, sawn wood, veneer sheets and unfinished wood articles used for the fabrication of bows for stringed musical instruments, but exempted finished bows. This is the only species subject to this annotation.

Domestic trade in *P. echinata* wood between companies and bow makers within Brazil have been regulated and controlled by the Document of Forest Origin (DOF) since 2006, but there appears to be some uncertainty about the accuracy of records of existing stocks registered in Brazil at the time the Appendix II listing came into force. DOF does not regulate domestic or international trade in bows as finished products, and therefore these do not need to be declared to the authorities and the total number of bows sold and exported per year remains unknown. According to the International Pernambuco Conservation Initiative (IPCI), a new domestic permit requirement for the export of finished bows became effective in June 2022—however reports from Brazil show that permit applications are not yet possible.

No empirical estimates of the natural populations of *P. echinata* across the Atlantic Forest are currently available. The species is fragmented between forest remnants and localised extinctions of subpopulations have been observed. The deforestation of the Mata Atlântica has been intensifying in recent years and over 21,600 ha of the territory were lost to deforestation between 2020 and 2021. The ongoing habitat loss and decline in habitat quality, coupled with exploitation for its wood, strongly suggest the population trend of the species is declining.

The species has been heavily traded for over 500 years, initially as a source of red dye (brazilein) and more recently as timber. Since the early 1800s, the species' wood, which is highly valued for its combination of durability, flexibility and resonance, has been extracted to produce bows for several musical instruments such as violins, violas, cellos, and basses. Overall, it has been estimated that over half a million mature trees were removed over the last five centuries. Trade is international; 92% of production was exported, estimated at more than 127,000 pieces of bow blanks or bows according to data collected during inspections carried out by the Brazilian Institute for the Environment and Renewable Natural Resources (IBAMA) of companies producing bows. The vast majority of exports are to the USA and Europe and to a lesser extent Asia.

According to an international survey of bow makers in July 2022, the average number of bows made out of *P. echinata* produced every year worldwide is approximately 25,000. Of the 337 bow makers who responded, approximately 91% produce fewer than 50 bows per year and nearly 44% produce fewer than ten.

Harvesting of *P. echinata* in its natural habitat and export is prohibited under Brazilian law (Federal

Law No. 11,428/2006 and Federal Decree No. 6,660/2008) and only wood from planted trees that are registered with the environmental agency or pre-Convention material can be traded. Its designation as endangered on the list of Brazilian flora threatened with extinction, means collection, harvesting, transportation, storage, handling, processing, and commercialisation from natural habitat is prohibited.

In the last five years, investigations by IBAMA and the Federal Police have shown that wood from natural forests has continued to be harvested to supply the growing international market for bows for musical instruments. Since 2018, IBAMA officers seized over 200,000 bow blanks and bows made with illegal (i.e., native) raw wood. The proponent considers that significant trade in illegally sourced wood may have taken place since the species was listed.

The proponent seeks to list the species in Appendix I with an annotation to include all parts and derivatives, including bows of musical instruments, with the exception of musical instruments and their parts, composing travelling orchestras, and solo musicians carrying musical passports in accordance with Res. Conf. 16.8 (now Res. Conf. 16.8 (Rev. CoP17)) on *Frequent cross-border non-commercial movements of musical instruments*. The stated purpose of the Proposal is to recognise the precarious conservation status of the species and to bring trade in finished bows under CITES trade control in order to reduce the opportunity for exports in contravention of Brazilian law. The justification for and intention of an exception for trade under “musical passports” is less obvious.

Analysis: *Paubrasilia echinata* has been subject to extensive historical exploitation for international trade and is affected by habitat loss due to deforestation, agricultural development, and urbanisation. Population estimates are not available, although known native populations are fragmented and small across the species’ range and some subpopulations have disappeared from areas where they used to occur. There is evidence of ongoing international demand in the USA, Europe, and Asia, and instances of illegal trade are being reported. Based on the registered annual rates of deforestation of the species’ natural habitat that contributed to an overall decline of more than 90% of the forest’s historical range, *P. echinata* appears to meet the biological criteria for inclusion in Appendix I of Annex 1 of Res. Conf. 9.24 (Rev. CoP17). Since Brazilian law does not allow the exploitation of *P. echinata* from its natural habitat, and only trade in wood from trees planted and registered with the environmental authorities or recognised as pre-Convention is allowed, the effect of the proposed transfer of this species from Appendix II to I largely relates to the cessation of the current Appendix II exemption for trade in finished products.

On this point, the proposed annotation for the species if transferred to Appendix I is to include “all parts and derivatives, including bows of musical instruments, with the exception of musical instruments and their parts, composing travelling orchestras, and solo musicians carrying musical passports in accordance with Res. 16.8” (now Res. Conf. 16.8 (Rev. CoP17)). Under that Resolution the use of “musical passports” only applies to Appendix I specimens acquired before the species was included in the Appendices, which in this case would be 2007 (the species was listed at CoP14), as well as Appendix II and III listed species. Any movement of post-2007 musical instruments and their parts, unless recognised as from artificially propagated trees, would need to be permitted on a case-by-case basis in compliance with Articles III and VII of the Convention (for example personal and household effects or pre-Convention specimens).

The inclusion of an annotation to an Appendix I listing proposal for a plant species would be unusual. If the intention of the proponent is to subject finished products to CITES trade control, while allowing for use of musical passports in accordance with Res. Conf. 16.8 (CoP17), this could alternatively be achieved by amending the Proposal to retain the species in Appendix II with a change to Annotation #10 to this effect. No other species are subject to this annotation. Brazil could also submit a zero quota for wild-sourced commercial exports to be posted on the CITES website to indicate that trade in wild harvest of the species from Brazil is not permitted.

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

Paubrasilia echinata (Gagnon, Lima and Lewis, 2016)

The species was listed in CITES Appendix II with the synonym *Caesalpinia echinata* in 2007, which became a synonym of *Paubrasilia echinata* in 2019 following taxonomic changes adopted at CoP18.

Despite being recognised as a single species, the morphological features of Pau-brasil vary considerably across its range and different morphotypes are genetically distinct. However, no subspecies have been officially recognised.

Range

Brazil

IUCN Global Category

Endangered A1acd (assessed 1998, ver. 2.3)

Biological criteria for inclusion in Appendix I**A) Small wild population**

No reliable, updated data on the size of the remaining populations are available. Natural populations are no longer found in Sergipe. A single forest fragment with a native population of Pau-brasil is situated in the state of Espírito Santo in Aracruz, and a new population has recently been discovered in another municipality of the same state. In compiling the National Floristic Inventory, the Brazilian Forest Service recorded no Pau-brasil specimens in sample plots in several areas where the species once occurred within the states of Espírito Santo, Sergipe, Paraíba, and Rio Grande do Norte. Natural populations of Pau-brasil were found in Rio de Janeiro, but no population size estimates were determined.

Extinctions of Pau-brasil subpopulations have been recorded over the years throughout the Atlantic Forest due to intense habitat degradation. Local extinctions caused the fragmentation of the species and the subsequent reduction in genetic variability. Further threats such as the decline of cocoa production areas (especially in southern Bahia) and selective logging of centennial trees in Paraíba, Rio Grande do Norte, to supply the market for musical instruments have aggravated the status of the remaining populations.

Despite a lack of concrete recent data on current population trends, the advance of deforestation, the demolition of the cocoa-cabruca agroforestry system and the increase in selective logging strongly suggest the overall trend to be declining.

B) Restricted area of distribution

Pau-brasil occurs exclusively in Brazil, specifically between Rio de Janeiro and Rio Grande do Norte, and it is restricted to the Mata Atlântica (Atlantic Coastal Forest). The Atlantic Coastal Forest currently extends for less than 100,000 km², i.e., approximately 7% of its original extent. The species inhabits lowland semi-deciduous seasonal forests, dense rainforest, dune forests and sandbanks within the coastal fragments of the Atlantic Forest biome.

The latest data on the distribution of the species was reported by Rocha and Simabukuro (2008) and Rocha (2010). The following table shows the municipalities where natural populations of Pau-brasil have been confirmed to occur.

Table 1. Areas with botanical records of natural populations of Pau-brasil.

| State | Occurrence area |
|---------------------|---|
| Rio Grande do Norte | Baía Formosa, Extremoz, Natal, Nísia Floresta, Parnamirim, and Tibau do Sul |
| Paraíba | Camaratuba, Mamanguape, and Rio Tinto |
| Pernambuco | São Lourenço da Mata until Vitória de Santo Antão, Nazaré da Mata, Tracunhaém, Pau d'Alho, Timbaúba, and Goiana |
| Alagoas | Junqueira and Coruripe |
| Bahia | Barrolândia, Camacan, Caraíva, Eunápolis, Guaratinga, Ibirapitanga, Ipiaçu, Itamaraju, Itapé, Jussari, Mascote, Pau-Brasil, Pirai do Norte, Porto Seguro Santa Luzia, Tapera, Ubaitaba, Una, and Vitória da Conquista |
| Espírito Santo | Aracruz and Vila Velha |
| Rio de Janeiro | Araruama, Arraial do Cabo, Búzios, Cabo Frio, Duque de Caxias, Iguaba Grande, Itaipuaçu, Japeri, Miguel Pereira, Niterói, Nova Iguaçu, Rio de Janeiro, São Pedro da Aldeia, and Saquarema |

The Rio de Janeiro Botanic Garden and others have identified and mapped 13 representative areas where Pau-brasil occurs in the state of Rio de Janeiro. Among these, ten are located within legally protected conservation

units. The total area of the 13 regions is 13,250 ha, with individual areas ranging from 72 ha to over 5,800 ha. Surveys from 2005 identified 1,754 individual trees on 130 different properties in the cocoa-growing region in southern Bahia. Of these, 85 had been planted and the rest were native. Moreover, isolated trees have been cultivated as ornamental plants in streets and parks throughout the country and sometimes can be found in commercial plantations too.

The historical geographical distribution of Pau-brasil has severely decreased due to deforestation, logging, exploitation for its wood, the development and expansion of urban centres, and changes in land-use for agriculture and forestry operations. At present, the largest populations of Pau-brasil are situated in forested areas within fully protected conservation units and in cocoa-cabruca agroforestry systems, where they are used as shade trees for growing crops of *Theobroma cacao*.

C) Decline in number of wild individuals

Deforestation accounted for the loss of over 14,000 ha of the Mata Atlântica between 2018 and 2019, approximately 13,000 ha between 2019 and 2020, and more than 21,600 ha between 2020 and 2021. As previously mentioned, the Atlantic Forest currently extends for less than 100,000 km² (or 10,000,000 ha), which represents about 7% of its historical range. Due to deforestation and the dismantling of the cocoa-cabruca agroforestry system, as well as intensive exploitation for the species' wood for international trade, the native population of *P. echinata* is inferred to be strongly declining.

Trade criteria for inclusion in Appendix I

The species is or may be affected by trade

Paubrasilia echinata was initially overexploited for the extraction of red dyes (brazilein) during the Portuguese occupation. *Intense harvesting continued until synthetic dyes became available in 1875, by which time extensive population declines had already taken place (IUCN, 1998).* Starting from the mid-18th century, the species' wood has been used to build stringed musical instrument bows and is still internationally sought after due to its high levels of resonance, density and durability. *According to Rocha (2010), approximately 527,000 mature specimens have been removed over the last 500 years. However, the International Pernambuco Conservation Initiative (IPCI) states that one mature tree can supply all bow makers in the USA or in France for a year, highlighting the notion that the amount of P. echinata wood used by the bow making industry (and therefore the amount of newly harvested wood the industry requires each year) is relatively small.*

When P. echinata was listed in CITES Appendix II in 2007, wood owners in Brazil were required to register their supplies with national management authorities. In the USA, bow makers apparently have approximately 60 years' worth (approximately 60 trees) of registered wood (IPCI, in litt., 2022).

The USA, Japan, Belgium, Germany, the Netherlands, Portugal, Italy, and France currently represent the largest consumers of raw *P. echinata*. IBAMA has reported that over 127,000 pieces of bow blanks and bows were sold abroad in the last 20 years. *US data report a total of 452 pieces being imported between 2009 and 2018 into Belize (70%), Brazil (20%), Mexico (5%), Austria (4%), Republic of Korea (0.7%) and Canada (0.2%). Over 95% of the pieces were labelled as wild-sourced and the rest as unknown. However, only about 24% were registered as having originated from Brazil. One instance of 316 pieces (approximately 70% of total pieces traded) being traded for commercial purposes was reported to have originated in Belize. No international trade of seedlings, seeds or bark is known. According to Cumine (2007), the term "Brazilwood" has also been used in trade to describe lower quality timber that is not P. echinata, but probably Massaranduba (Manilkara spp.).*

Following the Appendix II listing in 2007, CITES data include a total of 129 instances of P. echinata trade, totalling more than 15,000 kg and approximately 35 m³ of sawn wood reported as being exported and over 25,500 kg and more than 15 m³ reported as being imported. The top importer countries included the USA (17% of total imports), Japan (12%), China (11%), Switzerland (10%), Germany (9%), Hong Kong SAR (5%), Republic of Korea (5%), France (4%), Italy (4%), and Canada (4%). The top exporters were the USA (34% of total exports), Brazil (20%), France (9%), Germany (8%), Italy (6%), and the UK (4%).

A July 2022 survey by IPCI sent to international professional organisations and bow makers recorded the average number of bows produced per year by each bow manufacturer. Approximately 25,000 bows are manufactured every year worldwide by the 337 different bow makers who responded to the survey. Among them, approximately 91% produce fewer than 50 bows per year and nearly 44% produce fewer than ten (IPCI, in litt., 2022).

More than 200,000 bow blanks and bows made with wood that had no proof of legal origin have been seized in Brazil since 2018. Illegally harvested wood is mainly used to supply the international market for musical instrument bows in the USA, Europe and Asia. Selective logging of mature trees inside Pau-brasil National Park in Porto Seguro has also recently been curtailed.

Investigations have found that many companies and independent bow makers have been making false declarations to source illegal timber and make it appear to have a legal origin. According to the Federal Police, smugglers made more than USD46 million from international trafficking.

The proposal includes the following annotation:

All parts, derivatives and finished products, including bows of musical instruments, except musical instruments and their parts, composing travelling orchestras, and solo musicians carrying musical passports in accordance with Res. Conf 16.8 (now Resolution Conf 16.8 (Rev. CoP17)).

The inclusion of an annotation to an Appendix I listing for a plant species is unusual.

An annotation was adopted in 2007 along with the species listing in Appendix II at CoP14, which designated: "Logs, sawn wood and veneer sheets, including unfinished wood articles used for the fabrication of bows for stringed musical instruments." Since the annotation did not include finished musical instruments, these are not currently regulated under CITES. Therefore, at present, any finished bows exported from Brazil or otherwise in trade are exempted from the Convention. This is the only species currently in the Appendices with Annotation #10. The intention of the proposed annotation is not entirely clear. The use of musical passports in accordance with Res. Conf. 16.8 (Rev. CoP17) is possible only for Appendix I specimens acquired before the species was included in the Appendices, which in this case would be 2007 (the species was listed at CoP14). Any trade not covered by a "musical passport", unless recognised as from artificially propagated trees, would need to be permitted on a case-by-case basis in compliance with Articles III and VII of the Convention (for example personal and household effects or pre-Convention specimens). This burden on enforcement will still remain, whichever annotation is used.

Taking into account that the inclusion of an annotation to an Appendix I listing proposal for a plant species is unusual, if the intention of the proponent is to include finished products, while allowing for the use of musical passports in accordance with Res. Conf. 16.8 (Rev. CoP17), this could be achieved by amending the proposal to retain the species in Appendix II with a change to Annotation #10 to this effect.

Additional information

Threats

In addition to large volumes of harvest and trade, *P. echinata* is threatened by habitat loss and fragmentation due to continued deforestation, urban development, tourist development and agriculture. 90% of the historical extent of the species' native habitat, i.e. the Atlantic Forest, has been lost due to extensive urban and agricultural development (IPCI, in litt., 2022). Fragmented populations are causing a decrease in genetic variation of the species. Further pressures on the species include illegal logging activities and large volumes of timber being discarded as waste by manufacturers.

Conservation, management and legislation

The Atlantic Coastal Forest encloses at least 28 Private Natural Heritage Reserves (RPPN), as well as many protected areas (Table 2). Landowners within the Reserves guarantee protection of local animals and plants in exchange for tax reductions.

Table 2. Protected areas with recorded occurrence of natural populations of *P. echinata*.

| State | Protected Area |
|---------------------|---|
| Rio Grande do Norte | Parque Estadual das Dunas; RPPN Mata da Estrela; Parque Ecológico Água das Dunas; Parque Estadual Mata de Pipa |
| Paraíba | ESEC Pau-brasil; REBIO Guaribas |
| Pernambuco | ESEC Tapacurá |
| Alagoas | ESEC Serra do Ouro; RPPN Usina Coruripe |
| Bahia | ESEC Pau-brasil, RPPN Estação Veracruz, PARNA do Descobrimento, PARNA do Monte Pascoal, PARNA do Pau-brasil, REBIO de Una, RPPN Serra do Teimoso |
| Espírito Santo | APA Lagoa Grande |
| Rio de Janeiro | APA Serra da Capoeira Grande, REBIO Tinguá, APA Massambaba, RESEC Estadual de Jacarepiá, APA Serra de Sapiatiba, APA do Pau-brasil, Parque Estadual Serra da Tiririca; Parque Municipal da Boca da Barra; Reserva Ecológica Darcy Ribeiro |

Specific national regulations and governance measures are in force to protect *P. echinata* (Table 3).

Table 3. List of national legal instruments that govern the exploitation of native Brazilian plants (including *P. echinata*).

| Law | Description |
|--------------------------------|---|
| Federal Law No. 6.607/1978 | Declares Brazilwood as the national tree with an associated campaign about the relevance of the species to the history of Brazil together with measures to encourage the planting, across the entire national territory, of Brazilwood tree nurseries, aimed at promoting the species' conservation. |
| Federal Decree No. 750/1993 | Prohibits harvesting in Mata Atlântica |
| CONAMA Resolution No. 278/2001 | Allows IBAMA to suspend authorisations on harvesting |
| CONAMA Resolution No. 317/2002 | Establishes the necessary criteria for conservation of genetic material and sustainability of the harvest of plant species threatened with extinction found in Mata Atlântica that need to be included in State Plans for Conservation and Use. |
| Federal Law No. 11,428/2006 | Prohibits the exploitation of native species included in the Official List of Threatened Species of Brazilian Flora in the Atlantic Forest. |
| Federal Decree No. 6,600/2008 | Regulates Federal Law No. 11,428 of 2006 |
| Federal Law No. 12,651/2012 | Creates the National Program for Pau-Brasil Conservation |
| MMA Ordinance No. 320/2012 | Creates the National Program for Pau-Brasil Conservation |
| MMA Ordinance No. 443/2014 | Lists <i>P. echinata</i> as threatened with extinction in Brazil. It imposes full protection for species in the categories Extinct in the Wild (EW), Critically Endangered (CR), Endangered (EN), and Vulnerable (VU), including prohibition of collection, harvesting, transportation, storage, handling, processing, and commercialisation, among others. |

As listed in the table above, Federal Law No. 11,428 of 2006 and the Federal Decree No. 6,600 of 2008 prohibit the exploitation of native species included in the Official List of Threatened Species of Brazilian Flora in the Atlantic Forest. Therefore, *P. echinata* cannot be extracted from its natural habitat. However, enforcement of national legislation is believed to be weak in the Atlantic Forest region and protection of forest fragments varies across the country.

The Document of Forest Origin (DOF) has been used for the transaction of wood between bow makers and companies within Brazil since 2006. However, the origin of the stocks registered in the system is believed not to be reliable and bows, as finished products, are exempted from being declared to the authorities. Therefore, the transportation and commercialisation of bows does not require environmental documents and Brazilian authorities do not know the number of bows that are sold each year. Moreover, this system does not request information on by-product waste (e.g., defective bow blanks). *A new domestic permit requirement for the export of finished bows became effective in June 2022—however reports from Brazil indicate that the permit application is currently unobtainable (IPCI, in litt., 2022).*

At the international level, *P. echinata* has been listed in CITES Appendix II since 2007, with Annotation #10 designating logs, sawn wood, veneer sheets, including unfinished wood articles used for the fabrication of bows for stringed musical instruments as needing a CITES permit. Finished bows are exempted from needing a CITES permit.

*Bow makers represent the community most involved in current conservation and plantation efforts, despite most of them living outside Brazil (Lichtenberg et al., 2022). Since 2000, the IPCI has partnered with government agencies from Brazil, non-governmental organisations, several universities, local cacao farmers and regional science and conservation organisations among others to carry out activities aimed at the conservation of *P. echinata*. These include growing and planting of seedlings, developing and implementing replanting strategies in farm and forest locations, funding inventories and the study of the distribution and dynamics of the species, mapping forest fragments and investing in studies into the genetic, anatomy and growth of the species, as well as silviculture and pernambuco propagation methods (IPCI, in litt., 2022). Over 250,000 *P. echinata* seedlings have been planted in five states (Bahia, Espírito Santo, Pernambuco, Rio Grande do Norte, and Paraíba) to date (IPCI, in litt., 2022). Approximately half of all replanting occurred in natural preserves and urban areas, whereas the remainder were planted on farms and small-scale plantations for potential commercial use once the trees reach maturity at 30–40 years (IPCI, in litt., 2022).*

No monitoring programme for natural populations of *P. echinata* is currently in force. According to Santana et al. (2020), a number of plantations in the northeastern states of Pernambuco, Rio Grande do Norte, and Alagoas are being monitored by a Brazilian non-governmental organisation called Associação Plantas do Nordeste.

Artificial propagation

No large-scale commercial plantations of *P. echinata* exist. Wood from small commercial plantations and conservation initiatives is not traded. The majority of the plantations do not meet the requirements established by national laws, which determine that: a) existing plantations must be registered in a timely manner with the environmental agencies, and b) technical projects prepared by legally qualified professionals must be submitted. Most plantations are not listed in IBAMA's National System of the Control of Origin of Forest Products (SINAFLOR) either.

Little information on the success or failure of propagation efforts or forestry experiments is available. Several partnerships carry out planting of Pau-brasil seedlings in privately owned areas for future commercial exploitation. However, the ideal age for harvesting planted *P. echinata* trees for the production of bows remains largely unknown. A study by Rolim and Piotto (2018) on a 24-year-old plantation found that the species needs long cycles (40–50 years) to reach at least 30 cm dbh. *According to the IPCI (IPCI, in litt., 2022), the tree begins to produce the dark heartwood for which it is renowned at 10 years old, and the heartwood becomes dominant after 20 years.* Moreover, timber traders prefer wild-grown wood as it is considered to be superior to that grown in plantations. *A paper by Lichtemberg et al. (2022) reports that the timber quality of planted trees compared to high-quality timber from trees in natural habitats has been seriously questioned, as have regulations for their commercial use.*

There is no available information on plantations of *P. echinata* outside of Brazil.

Implementation challenges (including similar species)

Paubrasilia echinata has no similar species listed in CITES. The wood can be easily identified by its orange/reddish colouration, storied rays on the tangential face, and the presence of brazilein, which appears as a reddish dye when in contact with a basic solution. The general aspect and colour help distinguish *P. echinata* from similar species (such as *Brosimum rubescens*, *Centrolobium* spp. and *Manilkara* spp.). *Handroanthus* spp. and *Dialium guianense* are also harvested for the production of bows for musical instruments—these may be distinguished from *P. echinata* by the deposits in heartwood vessels (*Handroanthus* spp.) and by axial parenchyma in narrow bands (*Dialium guianense*).

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Inclusion of all African populations of *Pterocarpus* species in Appendix II of CITES with Annotation #17, including already listed species *P. erinaceus* (CoP17, no annotation) and *P. tinctorius* (CoP18, Annotation #6) in accordance with Article II, paragraph 2 (a) of the Convention

Proponents: Côte d'Ivoire, European Union, Liberia, Senegal, Togo

Summary: *Pterocarpus* is a genus of around 40 species native to tropical and subtropical regions worldwide of which 12 are native to Africa. In addition there are disputed reports of a South American species occurring also in Democratic Republic of the Congo (DRC) and one additional African native species is accepted by some botanists. African species are an important source of highly valued timber traded internationally, exported mainly in the form of logs and sawn timber. Commonly used trade names for the timber include “mukula”, “rosewood”, “African Padauk” or “African Padouk”. African species that produce rosewood or other precious hardwoods include *P. angolensis*, *P. erinaceus*, *P. lucens*, *P. soyauxii*, *P. tessmannii* and *P. tinctorius*. *P. erinaceus* is the only African species of the genus officially recognised in China as a “Hongmu” species (formally accepted for the production of rosewood furniture), but others are also considered desirable for furniture production. There has been a major growth in consumption of Hongmu and other “rosewoods” in China since 2010 leading to dramatically increased levels of exploitation in range States. Other genera are also traded with the name rosewood including species of *Dalbergia* (genus listing at CoP18) and *Guibourtia* (three African species listed at CoP18).

Two African species of *Pterocarpus* are already listed in Appendix II. The Endangered species *Pterocarpus erinaceus* was added to Appendix II at CoP17 with no annotation. *P. tinctorius* which is evaluated as Least Concern was added to Appendix II at CoP18 with Annotation #6 (Logs, sawn wood, veneer sheets and plywood). Most of the remaining African *Pterocarpus* spp. are widespread and may be locally common. The exception is the very rare *P. zenkeri*. The taxonomic status of this species is still debated but it is assessed as Endangered. Most other African *Pterocarpus* have been assessed as globally Least Concern since 2018, although several of these species are in significant decline in parts of their range. Species considered to be overexploited for their timber, with unsustainable harvest rates and some local stocks now exhausted include *P. angolensis*, *P. soyauxii*, and *P. tessmannii*.

Pterocarpus angolensis is one of the most valuable timber species in southern Africa harvested for local use and international trade. Intensive harvesting and the lack of natural regeneration have been causes for concern across parts of its range. Current levels of timber harvesting are thought to be unsustainable in various countries, almost certainly exceeding the rate at which harvestable-sized trees are being replenished in the population.

Pterocarpus soyauxii has a wide distribution. It has not yet been evaluated at a global level but has been evaluated as nationally threatened in DRC. The timber is harvested for international trade and it is one of the main species currently recorded in Chinese and Vietnamese markets.

Pterocarpus tessmannii occurs in DRC, Equatorial Guinea, and Gabon. It is exploited for timber and is now listed as globally Near Threatened.

Pterocarpus zenkeri was assessed as Endangered in 2015. It is endemic to Cameroon and is considered to be very rare. Although it is not known to be in trade currently, the similarity with *Pterocarpus soyauxii* may lead to it being harvested either intentionally or accidentally.

Of the other species, some are harvested for timber (*P. lucens*, *P. mildbraedii*, and *P. osun*), but this is uncertain for *P. brenanii*, *P. rotundifolius*, and *P. santalinoides*. The presence of *P. officinalis* in Africa is disputed.

In general, very little species-specific trade data are available for African *Pterocarpus*, and it is unknown how much harvest in each species is for domestic versus international markets. There is evidence of continuing increases in export of processed and unprocessed 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 expansion of demand for Hongmu and other “rosewoods”, has led to an unprecedented interest in Mukula timber in the main producer countries notably Zambia and DRC; with exponential development of logging leading to cumulative extractions estimated as several tens of thousands of m³ in countries including Zambia, DRC, Mozambique, Malawi, and Angola.

There are reports that since the listing of *P. erinaceus*, traders are redirecting attention to other, non-CITES species of *Pterocarpus*. Timber traders appear to be continually searching for substitute species for international trade, working both within and outside the law. Trade generally shifts between African *Pterocarpus* spp. depending on availability and multiple species are commonly traded under the same names. It is difficult to determine trade levels for individual species. In Customs data, most importing countries record imports of “rosewood” as tropical hardwood “not elsewhere specified”.

The timber of African *Pterocarpus* spp. is hard to distinguish. Even the most commonly logged species of African *Pterocarpus* are not easy to identify by loggers, local botanists and forest managers. There are, for example, similarities of appearance between the timber of the CITES-listed *P. erinaceus* and *P. tinctorius*. The sawn timber of *P. tinctorius* is commonly confused with that of *P. angolensis* and *P. soyauxii* and there may be confusion between *P. soyauxii* and *P. tessmannii*.

The Proposal is to list all African populations of *Pterocarpus* species in Appendix II of CITES with annotation #17, including already listed species *P. erinaceus* (CoP17, no annotation) and *P. tinctorius* (CoP18, Annotation #6) in accordance with Article II, paragraph 2 (a) of the Convention. There has been negligible trade of timber products of *P. erinaceus* and *P. tinctorius* reported as originating outside Africa and neither are known to be in plantations outside Africa. One species of Asian *Pterocarpus* would remain in the Appendices with Annotation #7.

Analysis: *Pterocarpus* is a tropical tree genus that produces valuable timber. Twelve species occur in Africa. Based on available information *P. angolensis*, *P. soyauxii*, and *P. tessmannii* appear to meet Criterion B in Annex 2a of Res. Conf. 9.24 (Rev. CoP17). The rare species, *P. zenkeri* appears to meet Criterion A in Annex 2a of Res. Conf. 9.24 (Rev. CoP17).

The most commonly logged species of African *Pterocarpus* are considered to be difficult to distinguish from one another by people involved in the trade, including loggers and forest managers, and by local botanists. Some African species can be distinguished using chemical and anatomical approaches, but it is extremely difficult, if not impossible, to distinguish African *Pterocarpus* species based on the wood anatomical features alone. As *P. erinaceus* is currently listed in Appendix II and believed to be affected by trade, all other African species therefore meet the lookalike criteria for listing in Appendix II provided in Annex 2b of Res. Conf. 9.24 (Rev. CoP17).

Previously listed species of the genus (*P. erinaceus* and *P. tinctorius*) are included in the Appendices regardless of where their populations are so that plantations outside their natural range are included. Were this proposal to be accepted, populations outside their natural range would no longer be included in the Appendices. Neither *P. erinaceus* nor *P. tinctorius* are known to be in plantations outside Africa this amendment to their listing should have no conservation impacts.

Annotation

Pterocarpus erinaceus was listed in CITES Appendix II without annotation. Almost all trade in the species since then has been reported by exporters in terms that are covered by Annotation #17. Experience with CITES listings of other rosewood species (e.g., see CoP17 Prop 53) has demonstrated that other annotations have been circumvented through minimal transformation of wood products. Inclusion of transformed wood would prevent this.

The current annotation for *P. tinctorius* is Annotation #6: “Logs, sawn wood, veneer sheets and plywood”. The change in annotation would mean that transformed wood would also come under CITES controls, again ensuring against circumvention seen with *Dalbergia cochinchinensis*.

The proposal to apply the Annotation #17 to all African populations of *Pterocarpus* spp. (including *P. erinaceus* and *P. tinctorius*) appears to be appropriate given that logs and sawn wood are the main products that are traded internationally and the inclusion of other forms of worked wood would prevent loopholes. If all African *Pterocarpus* spp. were covered by the same annotation, this would be an aid to enforcement. One species of Asian *Pterocarpus* would remain in the Appendices with Annotation #7.

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 12 African species of the genus *Pterocarpus* according to the African Plant Database. An additional species, *Pterocarpus zenkeri* Harms, is not recognised by some taxonomists. According to the African Plant Database, “*P. zenkeri* is a doubtful species very similar to *P. osun*”. *Pterocarpus zenkeri* has however been assessed for the IUCN Red List. *The presence of P. officinalis in Africa is disputed.*

IUCN Global Category, Range and Population Trend

Table 1. IUCN assessment, distribution range and population status for African *Pterocarpus* species (Sources Supporting Statement, IUCN Red List Assessments and GlobalTreePortal).

| Species | IUCN Global Category | Range | Population trend | Harvested for timber |
|--------------------------------|--|---|--|----------------------|
| <i>Pterocarpus angolensis</i> | LC (2018) | Angola, Botswana, Congo, DRC, Eswatini, Malawi, Mozambique, Namibia, South Africa, Tanzania, Zambia, Zimbabwe | Decreasing in parts of its range | Yes |
| <i>Pterocarpus brenanii</i> | LC (2018) | Malawi, Mozambique, Zimbabwe, Zambia (<i>Malawi is not included in GlobalTreePortal</i>) | Unknown | Unknown |
| <i>Pterocarpus erinaceus</i> | EN (2020) | Benin, Burkina Faso, Cameroon, Central African Republic, Chad, Côte d'Ivoire, Gambia, Ghana, Guinea, Guinea-Bissau, Liberia, Mali, Niger, Nigeria, Sierra Leone, Senegal, Togo (<i>Gabon is also included in GlobalTreePortal</i>) | Decreasing | Yes |
| <i>Pterocarpus lucens</i> | LC (2012) | Angola, Botswana, Cameroon, Chad, Congo, DRC, Ethiopia, Ghana, Guinea, Guinea-Bissau, Malawi, Mali, Mozambique, Namibia, Niger, Nigeria, Senegal, Sudan, Uganda, Zambia, Zimbabwe (<i>Congo, Guinea-Bissau, Malawi, Nigeria and Zimbabwe are not included in GlobalTreePortal whereas Burkina Faso and South Africa are included</i>) | Stable | Yes |
| <i>Pterocarpus mildbraedii</i> | LC (2022) | Benin, Cameroon, Côte d'Ivoire, Equatorial Guinea, Gabon, Ghana, Liberia, Nigeria, Sierra Leone, Tanzania. Records for DRC are based on misidentification. <i>The distribution given in GlobalTreePortal also includes the Central African Republic, Congo, Togo</i> | Unknown | Yes |
| <i>Pterocarpus officinalis</i> | <i>Presence in Africa is disputed. The published assessment for the species, native to the</i> | DRC (pantropical as Plants of the World Online suggests “Mexico to tropical America”). <i>The distribution of this species does not include Africa (GlobalTreePortal, Barstow and Klitgård, 2018)</i> | Presence in Africa is disputed. Decreasing elsewhere | Yes |

| Species | IUCN Global Category | Range | Population trend | Harvested for timber |
|----------------------------------|---|---|------------------|----------------------|
| | Caribbean, Central and South America, is NT (2018). | | | |
| <i>Pterocarpus osun</i> | LC (2020) | Cameroon, Equatorial Guinea, Nigeria. <i>The distribution of this species does not include Equatorial Guinea (Global Tree Portal, Barstow, 2020)</i> | Unknown | Yes |
| <i>Pterocarpus rotundifolius</i> | LC (2019) | Angola, DRC, Malawi, Mozambique, Tanzania, Zimbabwe, Zambia. <i>The distribution includes Botswana, Eswatini, Namibia, South Africa (Global Tree Portal, Botanic Gardens Conservation International (BGCI) & IUCN SSC Global Tree Specialist Group, 2019a)</i> | Stable | Unknown |
| <i>Pterocarpus santalinoides</i> | LC (2019) | Benin, Burkina Faso, Cameroon, Côte d'Ivoire, Ghana, Guinea, Guinea-Bissau, Liberia, Mali, Nigeria, Senegal, Sierra Leone, Togo. Also widespread in South America. <i>This species also occurs in Gambia (Global Tree Portal, Botanic Gardens Conservation International (BGCI) & IUCN SSC Global Tree Specialist Group, 2019b)</i> | Stable | Unknown |
| <i>Pterocarpus soyauxii</i> | Draft assessment LC | Angola, Cameroon, Central African Republic, Congo, DRC, Equatorial Guinea, Gabon, Nigeria | | Yes |
| <i>Pterocarpus tessmannii</i> | NT (2022) | DRC, Equatorial Guinea, Gabon. <i>Also in Guinea (GlobalTreePortal)</i> | Unknown | Yes |
| <i>Pterocarpus tinctorius</i> | LC (2018) | Angola, Burundi, DRC, Malawi, Mozambique, Tanzania, Zambia | Decreasing | Yes |
| <i>Pterocarpus zenkeri</i> | EN (2015) | Cameroon (<i>GlobalTreePortal</i>). <i>This is a Cameroon endemic known only from two localities: Yaoundé (Central Region) and Dikop near Eseka (South Region) (Cheek, 2015)</i> | Decreasing | Possibly |

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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

Africa has 12 *Pterocarpus* species; the presence of one, *P. officinalis*, is disputed. An additional species is recognised by some botanists. Various species produce rosewood or other precious hardwoods, including *P. angolensis*, *P. lucens*, *P. mildbraedii*, *P. osun*, *P. soyauxii*, *P. tessmannii*, and *P. tinctorius*. Several species are traded under the same common name "Padouk" (*P. mildbraedii*, *P. soyauxii*, *P. tessmannii*, and *P. tinctorius*). Currently two African *Pterocarpus* species (*P. erinaceus* and *P. tinctorius*) are listed in CITES Appendix II. Between 2010 and 2014, China registered a 700% increase in the import of African rosewood logs and sawn wood. Rosewood is a commercial term encompassing hardwood species harvested to produce Chinese traditional Hongmu furniture. Trade in rosewood has been characterised by a shifting supply between different countries and species. African rosewood imports comprised nearly half of these rosewood imports. High-value rosewood was traditionally produced from *Dalbergia* species, but the overwhelming demand from China and the increasing rarity of Asia's Hongmu species combined with stricter conservation measures and enforcement of logging regulations forced the trade to turn progressively towards similar alternative species and in particular the *Pterocarpus* genus.

Pterocarpus erinaceus

This species is the only African species of *Pterocarpus* on the list of 29 species officially designated as Hongmu under China's National Hongmu Standard (2017). The Standard can be legally enforced in relation to product marketing claims, and has played a key role in guiding the choice of materials in rosewood product manufacturing and consumption (Forest Trends, 2020). Viet Nam has also historically been a key market for rosewood, but imports dropped dramatically between 2015–2020, then stopped altogether in 2021. Viet Nam is the world's second-largest importer of African timber after China, with imported *P. erinaceus* largely processed into rosewood furniture for export to China. There are reports that following the listing of *P. erinaceus*, traders are redirecting their attention to other, non-CITES species (Wilmé et al., 2020). Timber traders appear to be continually searching for substitute species to exploit internationally, working both within and outside the law (UNODC, 2020). At the same time, harvesting of *P. erinaceus* continues to cause population declines. There has, for example, been a decline of about 50% in the population of *P. erinaceus* outside conservation areas between 2013–2021 (Forestry Commission of Ghana, 2021).

Other species considered to be over-exploited for their timber with many local stocks now exhausted include *P. soyauxii*, *P. tinctorius*, and *P. tessmannii*, which is now listed as Near Threatened. The very rare and Endangered *P. zenkeri*, endemic to Cameroon, is potentially harvested for its timber either intentionally or due to mistaken identity with *Pterocarpus soyauxii*.

Pterocarpus angolensis

This species is among the most important indigenous timber species in southern Africa. Its wood is known under many local names, such as umbila, muninga, mukwa, kiaat, and girassonde. It is sought after for carving, furniture and flooring because of its grain, colour, durability and stability, and is the most widely exploited wood in southern Africa (De Cauwer et al., 2017). The extensive trade in this wood on national and international markets has caused it to become depleted from some areas. Across parts of its range the largest and most mature trees are at risk from overexploitation for timber causing decline in some subpopulations outside of protected areas and sometimes inside protected sites due to both legal and illegal logging activity. In areas of extraction the size structure of subpopulations has become truncated, with smaller trees (often under 30 cm dbh) being much more common than those of larger size. This reduces the amount of seed being produced by the population causing concern about the impact this might have on regeneration at some sites (Barstow and Timberlake, 2018). Harvesting in Tanzania has been at a rate that could reach "economic extinction". There is little trade data available for this species: 5,000 m³ are reported to have been exported from Zambia. The biggest importers of the timber are Thailand and China. This tree can also be used for medicinal purposes (Barstow and Timberlake, 2018).

The species is threatened by land-use changes, overharvesting and/or frequent intense fires in many countries. In addition, climate change is expected to decrease the distribution range, especially where climate projections predict a decrease in summer rainfall. These threats affect the species' wood availability as the tree grows only in natural mixed forests. In contrast to the importance of its timber, information on *P. angolensis* is not sufficient to support forest management, especially data on population dynamics and productivity. More knowledge on the productivity of *P. angolensis* would allow improved forecasts of its growth, mortality, recruitment, and timber yield (De Cauwer et al., 2017).

Pterocarpus angolensis has been evaluated as of national concern in Angola. This dry forest species has a large proportion of its global range concentrated in Angola where it occurs in areas that are affected by high and increasing deforestation rates and relatively low tree cover. It has been proposed as one of 11 timber species of high priority for conservation attention in Angola (Romeiras et al., 2014).

While unsustainable or illegal logging only accounts for 9% of the net annual deforestation rate in Mozambique, there are also concerns over the potentially complete depletion of commercial species over the next 15 years. For example, more than half the volume of the commercial species harvested belong to just three species including *Pterocarpus angolensis* (umbila). Based on customs import and export data, the rate of harvesting for this species exceeds the higher limit of Mozambique's annual allowable cut (Macqueen, 2018). In the first quarter of 2014, Mozambique became the largest African supplier of imported timber to China. China was the destination for 93% of Mozambique's timber exports (Macqueen, 2018). Most exports of the main timber species exported are logs or low value-added products. Trees larger than the minimum permitted diameters have become difficult to find due to the pressure on logging, which incentivises cutting trees below those limits. Lack of information is also a problem since local tree fellers simply cut trees first and try to sell to those who show interest afterwards. Any wood not bought by Chinese operators is bought by local carpenters for furniture production—except curved or split wood, which is often abandoned in the forest (Macqueen, 2018).

Pterocarpus angolensis (Kiaat) was declared a protected species in Namibia in 1952. Its status as a protected species was listed in the regulations to the Forest Act 2001. Within Namibia, permit data indicate that Kiaat is used to produce planks, blocks, sawn timber and wood carvings. Its status as a protected species was listed in the

regulations to the Forest Act 2001. This species is also protected in Botswana, Malawi, Zimbabwe, Eswatini, Zambia, and South Africa (Nott et al., 2020).

Pterocarpus angolensis is considered to be vulnerable in Malawi, when considering wild populations, and vulnerable in Namibia and Zimbabwe when considering the species as an economic entity. However, in Zimbabwe the species as a whole is considered at low risk of extinction and within South Africa the species is assessed as least concern (Barstow and Timberlake, 2018).

In Tanzania, an 80% decline in population is estimated based on seed collection data between 2000–2020 with an estimated 85% decline in use and trade in 94 local timber yards surveyed (Mashimba, in litt., 2022).

Pterocarpus lucens

This species has a wide distribution in two bands across tropical Africa from Senegal to Ethiopia and Angola to Mozambique. It was evaluated as Least Concern globally in 2012 with a stable population. It is not recorded as nationally threatened for any country (GlobalTreePortal, 2022), but has been recorded as threatened at the population level in Burkina Faso, Niger, and Senegal (Winfield et al., 2016). It is apparently widely exploited both for a variety of local uses and the international timber trade.

Pterocarpus soyauxii

This species was among the first timber species to be exported from Gabon. Between 2000–2003, Gabon exported 120,000 m³ of *P. soyauxii* logs annually. Since 2010 the export of logs has been banned by Gabon.

Padouk was the second most important timber exported from Gabon by volume 2007–2017 (all commodities combined) with a volume of 1,194,407 m³ (Mahonghol et al., 2020). Much of this is presumably *P. soyauxii*, which is listed as one of the 24 species currently harvested (Timber Trade Portal, 2022). *Pterocarpus soyauxii* is one of the main timbers produced in DRC and in the Congo (Timber Trade Portal, 2022).

The timber of *P. soyauxii* is one of the main African *Pterocarpus* species currently recorded in trade within China. In Kunming it is recorded with other rosewood species such as *P. erinaceus*, *P. tinctorius* and *Guibourtia* spp. and is traded as hewn logs and squared timber having been brought into China mainly through Zhangjiagang (Zhang and Chen, 2022b). This species also comprised around 80% of African rosewood imports to Viet Nam during 2018–2022 with a value of USD62 million (Panjiva, 2022).

A draft global conservation assessment has been undertaken for *P. soyauxii* that is unlikely to raise conservation concerns (Barstow, in litt., 2022). Despite the potential threat to the species from timber use, the species has a wide range and large population and regenerates well (Doucet, in litt., 2022). Nevertheless, *P. soyauxii* has been evaluated as nationally threatened in DRC with threats including logging, shifting agriculture and other forms of land degradation (Kiyulu and Rodrigues, 2014).

Pterocarpus tessmannii

This species is exploited for timber and traded with other *Pterocarpus* species as “African padauk”. More research is needed into the extent and impact of this (Hills, 2021). There is no known information on its conservation status at a national level in DRC, Gabon, or Equatorial Guinea.

Other species

Very limited information appears to be available on levels of trade and the impact of exploitation for *Pterocarpus brenanii*, *P. lucens*, *P. osun*, *P. rotundifolius* and *P. santalanoides*. Bans on exports of *P. lucens* are however in place in several countries giving possible indications of concerns about levels of exploitation.

In the Central African Republic, only 34 “species” found in the closed forest area are typically harvested including Padouk (*Pterocarpus* spp.) but these are not amongst the most heavily traded (Timber Trade Portal, 2022).

International trade in *Pterocarpus* spp. is poorly documented by importing countries. Because there is no universal definition of “rosewood”, there are no global statistics on the rosewood market. In most national systems, imports are typically registered as tropical hardwood “not elsewhere specified”. Most of the rosewood species used for Hongmu come from the *Dalbergia* and *Pterocarpus* genera (UNODC, 2020).

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

Despite their listing in CITES Appendix II, *P. erinaceus* and possibly *P. tinctorius*, are still subject to illegal logging and trade, notably because existing exports are mislabelled as other non-CITES species of the genus.

A paper submitted by the Scientific Authorities of Germany and Belgium at CITES PC24 in 2018 noted that the expansion of Hongmu demand has led to an unprecedented interest in Mukula timber in the main producer countries notably (but not exclusively) Zambia and DRC; with exponential development of the logging marketing chain leading to cumulative extractions estimated as several tens of thousands of m³ in countries including Zambia, DRC, Mozambique, Malawi, and Angola, although no precise assessment of the volumes involved is available. In some places, there may be problems regarding the legality and sustainability of the logging operations. Although the extraction of Mukula timber affects several species in the genus *Pterocarpus*, they are not differentiated or are poorly differentiated by loggers and traders in the international markets. Empirical observations indicate that several of these species are in critical decline in numerous regions of the countries within their range (PC24 Doc. 19.2, 2018).

Even the most commonly logged species of African *Pterocarpus* are often not easily distinguishable from one another by loggers, local botanists and forest managers. There are reported confusions between *P. soyauxii*, *P. tessmannii*, and *P. castelsii*. The latter, not accepted by taxonomists but anyway recorded in the field, refers to *P. soyauxii* or *P. tessmannii*, adding even more confusion. Although some of the species have recently been distinguished from one another thanks to a combination of chemical and anatomical approaches, it is extremely difficult, if not impossible, to distinguish African *Pterocarpus* species based on the wood anatomical features alone.

As some of the *Pterocarpus* species are not clearly distinguishable, and there is evidence that two currently CITES-listed *Pterocarpus* species might be traded under the label of non-CITES listed species, it is also proposed that the African species in the genus are included in Appendix II under Criterion A of Annex 2 b of Resolution Con. 9.24 (Rev. CoP17), or the lookalike provisions.

There are similarities of appearance between the timber of *P. erinaceus* and *P. tinctorius*. Trade shifts between African *Pterocarpus* spp., depending on availability, and multiple species are commonly traded under the same names as has been seen with other rosewoods such as *Dalbergia* spp. and *Guibourtia* spp.

Although the extraction of Mukula timber affects several species in the genus *Pterocarpus*, they are not differentiated or are poorly differentiated by loggers and traders in international markets (PC24 Doc. 19.2, 2018). In Zambia more research is needed to clarify the botanical characteristics of what goes under the name Mukula. The mechanisms underpinning the market in this timber move away from targeted species and countries, targeting instead lesser known species or new geographies altogether. The recent documented cases of Mukula harvesting in DRC, Malawi and other countries illustrate this (Cerutti et al., 2018). The sawn timber of *P. tinctorius* is similar in appearance to that of *P. angolensis*, *P. soyauxii* or *P. castelsii*. All these species are referred to locally as Mukula as reflected in the Namibian Department of Forestry in-transit permit data. Many consignments of this timber are transported through Namibia from Zambia and the DRC for export to China (Nott et al., 2020).

Background to annotation amendments

P. erinaceus was listed at CoP17 with no annotation meaning that all readily recognisable parts are included in the listing. The majority of trade reported by exporters has been in logs (>204,000 m³), sawn wood (1,106,000 m³) with only 550 kg and 45 m³ of wood products. Significantly higher quantities have been reported by importers but with the same products dominating the trade. No trade has been reported of transformed wood by exporters although importers have reported around 200 m³.

P. tinctorius was listed at CoP18 with Annotation #6, including logs, sawn wood, veneer sheets and plywood. Since then, exporters have mostly reported exports of timber, sawn wood and logs. Only 2 kg of wood products have been reported by importers and no trade in transformed wood has been reported, although this would not have been covered by the current annotation.

Additional information

Threats

African *Pterocarpus* spp. also provide a range of medicinal products harvested for local use. Overharvesting may be a threat locally. The viability of existing wild *P. angolensis* populations is threatened, not only due to extensive harvesting of its bark for medicinal purposes or as fuel wood, but also owing to establishment problems resulting from seedling failure after germination. It has been estimated that only 2% of fruits produce seedlings in a given year, with just half of those seedlings surviving the first year of growth. Successful seed germination appears to depend on wildfires that remove the wings and bristles from pods, crack the seed and improve contact with the soil (Sadiki et al., 2018).

De Cauwer et al. (2016) considered the environmental drivers of change in the transition zones of woodlands in Namibia. Their study concluded that while *Kiaat* communities were better able to withstand high fire frequency than other communities, they show a higher vulnerability to climate change.

Pterocarpus mildbraedii is reported to be threatened by deforestation and urbanisation.

Conservation, management and legislation

In Cote d'Ivoire, Decree No. 2013-508 of 25 July 2013 bans the exploitation, cutting, transport, marketing and export of all Pterocarpus spp. (UNEP-WCMC, 2020).

Benin has prohibited the export of all timber species in their raw form, including Pterocarpus erinaceus, a protected species since 2005, with only finished products permitted in trade. Burkina Faso has prohibited the export of all logs and processed products of Pterocarpus erinaceus and P. lucens since 2005. Cameroon has banned log exports of rosewood species found in the country since 1999. Ghana has banned the harvesting and export of P. erinaceus and P. lucens since 2014 (Zhang and Chen, 2022a).

In view of the rapid increase in the trade of P. erinaceus, Ghana has established domestic regulatory mechanisms to manage and monitor the populations of P. erinaceus. This includes the adoption of a quota system to regulate the harvest and trade and ensure sustainable trade and conservation of the species in the wild. Enforcement of a national ban on harvesting and export of P. erinaceus and plantation development programmes have been put in place.

Artificial propagation

Attempts to establish plantations of P. angolensis have been largely unsuccessful. More knowledge on the productivity of P. angolensis would possibly assist in establishment of plantations as well as allowing improved forecasts of its growth, mortality, recruitment and timber yield for sustainable management in natural forests (De Cauwer et al., 2017).

Implementation challenges

The CITES listing of P. erinaceus has led to significant implementation challenges. In October 2018, the CITES Standing Committee decided to suspend trade in P. erinaceus from Nigeria until the Party made a scientifically based non-detriment finding (NDF) to the satisfaction of the Secretariat and PC Chair. Given the high volumes of trade, a Significant Trade Review was carried out for the species. UNEP-WCMC was commissioned to undertake reviews of P. erinaceus trade for all 17 range States for consideration by the CITES Plants Committee. All range States were consulted and asked to provide information on the scientific basis by which they had established that exports were non-detrimental and compliant with Article IV, including details on the population status and threats to the species within their country, together with information on trade, legal protection, and management actions. The study concluded that no range State of P. erinaceus demonstrated that the provisions of Article IV were being met. Further guidance and capacity building in relation to timber NDFs are therefore required across all range States to ensure that any future exports are science-based and that ongoing adaptive management is in place (UNEP-WCMC, 2020).

Subsequently at the March 2022 Standing Committee, the Secretariat was asked to open an Article XIII procedure for P. erinaceus for all range States based on the exceptional circumstances due to pervasive documented illegal trade. As a consequence of this and analysis of the responses from the range States, a suspension was put in place in June 2022 for all commercial trade in P. erinaceus from those Parties that did not reply or did not provide a satisfactory justification while other range States adopted a voluntary zero quota (See: Notification to the Parties No. 2022/045).

It is not clear to what extent range States currently have inventory data for Pterocarpus spp. but partial information is available as highlighted in Winfield et al. (2016) to guide NDF decisions. This report notes that a surprising amount of information was available for a number of Pterocarpus species in Africa, mainly the highly exploited species P. erinaceus, P. lucens, and P. angolensis. However, even these studies were restricted to selected populations, thus leaving large data gaps. Without even a basic understanding of existing standing stocks and their structure it is difficult to ascertain what a sustainable level of harvest would or could be for any of these species. Some inventory data are available: for P. angolensis in Namibia and large-scale inventories for P. soyauxii have been carried out in central African countries.

Law enforcement in the forest sector in various African countries is perceived as in need of strengthening. The forest sector in Mozambique, for example, has about 630 inspectors, which is far below the ideal minimum of 1,800 to allow adequate monitoring. Despite recent purchases of more vehicles for inspectors, agent recruitment and training, and more fixed inspection posts, only a small number of transgressors, vehicles and containers have been seized in recent years (Macqueen, 2018).

There is a risk that consignments of African Pterocarpus timbers will be mislabelled as species not included in the CITES Appendices, such as the Asian species P. macrocarpus and P. indicus, which are both listed on the IUCN Red List as Endangered with the main threat being overexploitation for timber. There is potential for mixing of different species in the process of transshipment involving various countries in Africa and Asia.

Potential risk(s) of a listing

The timber sector is of major importance to African economies in terms of employment and socio-economic development. All aspects of forestry and sustainable wood utilisation are important within the range States. Livelihood implications need to be considered. In parts of Africa, the uncontrolled expansion of logging of Pterocarpus spp. has been made possible by the extreme poverty of the rural populations, for whom the purchase prices proposed by intermediaries are an unparalleled windfall, despite being very low (PC24 Doc. 19.2, 2018).

There is a potential risk of the trade shifting to other non-CITES species that produce similar timber. Furthermore, a significant proportion of the range of Pterocarpus santalinoides is in South America where populations would not be included in the listing. Although Pterocarpus officinalis is considered within the listing proposal as an African species, the range is actually Caribbean, Central, and South America. Trade in these populations would not be included in the listing.

Potential benefit(s) of listing for trade regulation

The two African species of Pterocarpus currently listed in CITES do not have overlapping distributions. The range States for the two species include 25 countries. The inclusion of all African species of the genus would increase the number of range states (notably adding Eswatini, Ethiopia, Namibia, South Africa, and Zimbabwe) and potentially strengthen trade controls on cross-border trade within Africa. Currently illegal Mukula extraction is considered a regional issue with significant extraction in southeastern DRC, Zambia, and northeastern Angola. Trucking routes are documented towards ports on the Atlantic (Angola, Namibia, and South Africa) and Pacific (Tanzania and Mozambique).

Efforts to improve sustainable forest management and promote certification are underway at a national and regional level in African countries. CITES listing could reinforce these efforts. Furthermore, it could establish necessary controls required for more effective data gathering on trade and support regional collaboration in data sharing and transparency as recommended by Mahonghol et al. (2020).

In China, the implementation of laws, regulations and administrative measures relating to rosewood (including Pterocarpus spp.) is ultimately guided by the species listed in the CITES Appendices. Listing of all African Pterocarpus species in Appendix II will strengthen the legislation and policy framework for Pterocarpus use in China, the main importing country. The Chinese rosewood industry has already initiated some informal discussions around the listing of all Pterocarpus in the CITES Appendices.

China has adopted stricter domestic measures than CITES to strengthen the trade control of imported CITES wood species by requiring a CITES import permit certificate for Appendix II species. Exporters are obliged to obtain a CITES export permit issued by the Management Authority of the exporting country in accordance with the requirements of CITES. They must also apply to the China CITES Management Authority in advance and obtain the CITES import permit certificate. These documents are checked by China customs. Before issuing the import permit, the CITES Management Authority will contact the CITES Secretariat or the CITES Management Authority of the exporting country for verification of the authenticity and validity of any CITES permit/certificate prior to the issuance of the import permit certificate (Zhang and Chen, 2022a).

CITES listing would provide a clearer picture of the international trade in African Pterocarpus timber as part of the global rosewood market. Currently, because there is no universal definition of "rosewood", there are no global statistics on the rosewood market. In most national systems imports are typically registered as tropical hardwood "not elsewhere specified". While traditional rosewoods have many uses, today most of the trade refers to tropical hardwoods suitable for making traditional furniture in the Asian style, typically referred to as Hongmu. Most of these rosewood species come from the Dalbergia and Pterocarpus genera (UNODC, 2020).

Other comments

Pterocarpus is within the scope of certification schemes: the FSC Certificates Public Dashboard records certificates for the genus relating to Cameroon and Gabon.

Viet Nam is the second-largest importer of African timber beyond China, with imported P. erinaceus largely processed into rosewood furniture for export to China. In 2019, Viet Nam signed a bilateral trade agreement, the Forest Law, Governance, Enforcement and Trade Voluntary Partnership Agreement (FLEGT-VPA), with the EU, which contained a commitment to ensure that wood products are legally sourced. In 2020, a Decree was issued on the Viet Nameese Timber Legality Assurance System (VNTLAS), stipulating that for countries and species deemed "high-risk," importers must provide additional documentation demonstrating legal compliance and undertake due diligence (Treanor, 2022).

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Inclusion of African populations of *Khaya* spp. in Appendix II with annotation #17 “Designates logs, sawn wood, veneer sheets, plywood and transformed wood.”

Proponents: Benin, Côte d'Ivoire, European Union, Liberia, Senegal

Summary: The genus *Khaya* consists of tree species native to tropical and subtropical Africa, Madagascar, and the Comoros. Five species are currently recognised: *K. anthotheca*, *K. grandifoliola*, *K. ivorensis*, *K. madagascariensis*, and *K. senegalensis*. A sixth species, *K. comorensis*, is subject to debate. A taxonomic revision of *K. anthotheca* is underway following a recent study based on genetic markers and morphological analysis.

The proposal covers African populations of the genus *Khaya*. No species within the genus are native elsewhere. Plantations of *Khaya* have been developed in Africa and in various countries outside this region, including Australia, Brazil, Indonesia, Malaysia, and Sri Lanka. These plantations are thought to be relatively small and may not yet be important sources of timber for international trade. *Khaya* spp. are all large trees that grow up to 60 m in height for the largest species (*K. anthotheca* and *K. ivorensis*). They produce some of Africa's most valuable timber for the international market traded under various names, notably as African Mahogany or Acajou. Logs, sawn timber and veneer are amongst the products exported from a range of African countries. The wood is used for boat building, construction, carpentry, panelling, flooring, furniture, veneer and plywood. It has been imported to Europe from West Africa since the late 19th century, with Central African countries later becoming an important source. The EU, USA, and China are currently among the major importers.

The timber of four of the five currently recognised *Khaya* spp. (*K. anthotheca*, *K. ivorensis*, *K. grandifoliola*, and *K. senegalensis*) is traded internationally, placing significant pressures on wild populations. These species are widespread in Africa but considered globally threatened because of population declines resulting primarily from commercial logging. They were all classified as Vulnerable in the IUCN Red List assessments in 1998 and are currently being re-assessed. Population density is generally considered to be low for *Khaya* spp.

The fifth species, *K. madagascariensis*, has been heavily exploited in the past so that commercial stocks are no longer available; it has an estimated population of 1,400 mature individuals, in approximately 14 subpopulations. The IUCN Red List assessment of 2020 classified it as Vulnerable based on past population declines of over 30% in three generations as a result of exploitation for timber.

- ***K. anthotheca*** is widespread in various forest types. It is harvested commercially for timber, traded as African Mahogany, *Khaya* or Acajou and has a declining population.
- ***K. grandifoliola*** occurs in semi-deciduous forest in countries extending from Guinea to Uganda. It has a declining population, and it is harvested commercially for timber, traded as African Mahogany, *Khaya* or Acajou.
- ***K. ivorensis*** is widespread occurring in evergreen and semi-deciduous forest with high rainfall. It is the most exploited species of the genus because its wood is regarded as of better quality than that of other species.
- ***K. madagascariensis*** is endemic to Madagascar and the Comoros. The population has declined due to timber exploitation in the past. There are no recent recorded exports.
- ***K. senegalensis*** is widespread in West and Central African savanna areas. The population has declined at least in parts of its range. It is harvested commercially for timber traded as African Mahogany, *Khaya* or Acajou.
- ***K. comorensis*** is not generally accepted as a distinct species. It is recorded from Comoros. There is no information known on its population size, trend or utilisation and trade.

No comprehensive global trade data exist for *Khaya* spp., however, from available, partial information from exporters and importers, Cameroon, Congo, and Côte d'Ivoire, Democratic Republic of the Congo (DRC), Gabon, and Ghana appear to be the main exporters of *Khaya* timber. Trade data, reported as sawn wood or similar, are not disaggregated to species, but *K. anthotheca*, *K. grandifoliola*, *K. ivorensis*, and *K. senegalensis* occur in all or some of these countries. Other species traded as African Mahogany include *Entandrophragma* spp. which are in the same botanical family. The African species of the unrelated genus *Azela* (the subject of proposal CoP19 Prop. 46) are also sometimes traded as African Mahogany.

Khaya spp. are regarded as indistinguishable from one another based on macroscopic features of their wood and also, according to the Supporting Statement (SS), based on microscopic wood characteristics. Generally, the timber of *Khaya* spp. is mixed in international trade shipments. The wood of *Khaya* spp. is easily mistaken for that of the CITES-listed *Swietenia* spp.

Analysis: Four of the five currently recognised *Khaya* species are widespread African trees that have been heavily harvested for their timber, in some cases for long periods (the fifth is *K. madagascariensis*, endemic to Madagascar and the Comoros). There are reports of declining populations as a result of harvest in a number of different range States. As a result all four species were classified as Vulnerable by IUCN in 1998 (*K. madagascariensis* was assigned the same status in 2020). However, apart from *K. madagascariensis*, there are no known national population estimates or stock assessments for the species. Nevertheless, harvesting and export has continued which is likely to have led to further depletion and in some cases exhaustion of harvestable stocks as is the case with *K. madagascariensis* which is no longer known to be in trade. There are strong indications that timber producing specimens of four (*K. anthotheca*, *K. grandifoliola*, *K. ivorensis*, and *K. senegalensis*) species are currently harvested unsustainably in sometimes large parts of their range, increasing their vulnerability to other important threats and therefore meeting Criterion B of Annex 2a Res. Conf. 9.24 (Rev. CoP17).

Given the similarity of appearance and the mixing of timbers in trade, other members of the genus would meet the lookalike criteria in Annex 2b of the Resolution. Plantations of *Khaya* have been developed in Africa and in other countries including Australia, Brazil, Indonesia, Malaysia, and Sri Lanka. Trade from plantations outside Africa would not be included under this listing, but trade from plantations within Africa would be included. The size of the African plantations and the extent to which they supply the global timber market is not known.

Available trade data for African Mahogany indicate that exports are mainly sawn wood or similar products. Annotation #17 Logs, sawn wood, veneer sheets, plywood and transformed wood would cover the products that are traded internationally from Africa. Furthermore, the inclusion of transformed wood would ensure that loopholes exploited for other timber species by minimal processing (See CoP17 Prop. 53) are closed.

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 taxonomy of *Khaya* remains unresolved. This proposal follows the nomenclature outlined in Royal Botanic Gardens Kew's Plants of the World Online database (POWO, 2022), which recognises five species (*K. anthotheca* (Welw.) C. DC. (1878), *K. grandifoliola* C. DC. (1907), *K. ivorensis* A. Chev. (1907), *K. madagascariensis* Jum. and Perr. (1906) and *K. senegalensis* (Desr.) A. Juss. (1830)), with the addition of a sixth species, *K. comorensis*, as reported by the CITES Management Authority of Comoros.

There is uncertainty as to whether K. comorensis is validly published. It is not included in POWO, 2022, Tropicos or IPNI. It is listed here (<https://unesdoc.unesco.org/ark:/48223/pf0000058054>) as nom nud (i.e., not published with the correct description or types). (Rivers, in litt., 2022).

A recent review of *K. anthotheca* by Bouka et al. (2022) identified two new undescribed species and supported the rehabilitation to the rank of species of three taxa previously considered to be synonyms of *K. anthotheca*: *K.*

agboensis, *K. euryphylla*, and *K. nyasica*. A full taxonomic revision of the genus is currently underway (Bouka et al., 2022).

IUCN Global Category, Range and population trend for *Khaya* spp.

Table 1. IUCN assessment, distribution range and population status of *Khaya* spp. (Sources SS and IUCN Red List Assessments)

| Species | IUCN Global Category | Range as given on IUCN Red List Assessment | Range as given in SS | Population trend |
|----------------------------|--|---|---|------------------|
| <i>K. anthotheca</i> | Vulnerable A1cd (assessed 1998, ver 2.3) | Angola, Cameroon, Congo, Democratic Republic of the Congo (henceforth DRC), Côte d'Ivoire, Ghana, Liberia, Malawi, Mozambique, Nigeria, Sierra Leone, Uganda, United Republic of Tanzania (henceforth Tanzania), Zambia, Zimbabwe | Angola, Cameroon, Congo, Côte d'Ivoire, DRC, Ghana, Liberia, Malawi, Mozambique, Nigeria, Sierra Leone, Uganda, Tanzania, Zambia and Zimbabwe. | Decreasing |
| <i>K. comorensis</i> | Not Evaluated | | Comoros | |
| <i>K. grandifoliola</i> | Vulnerable A1cd (assessed 1998, ver 2.3) | Benin, Congo, DRC, Côte d'Ivoire, Ghana, Guinea, Nigeria, South Sudan, Sudan, Togo, Uganda | Benin, Burkina Faso, Congo, Côte d'Ivoire, DRC, Ghana, Guinea, Nigeria, South Sudan, Sudan, Togo and Uganda. | Decreasing |
| <i>K. ivorensis</i> | Vulnerable A1cd (assessed 1998, ver 2.3) | Angola, Cameroon, Côte d'Ivoire, Gabon, Ghana, Liberia, Nigeria | Angola, Cameroon, Côte d'Ivoire, Gabon, Ghana, Liberia, Nigeria | |
| <i>K. madagascariensis</i> | Vulnerable A2c (assessed 2020). | Madagascar, Comoros | Madagascar, Comoros | Decreasing |
| <i>K. senegalensis</i> | Vulnerable A1cd (assessed 1998, ver 2.3) | Benin, Burkina Faso, Cameroon, Central African Republic, Chad, Côte d'Ivoire, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Mali, Niger, Nigeria, Senegal, Sierra Leone, South Sudan, Sudan, Togo, Uganda | Benin, Burkina Faso, Cameroon, Central African Republic, Chad, Côte d'Ivoire, Gabon, Gambia, Ghana, Guinea, Guinea Bissau, Mali, Niger, Nigeria, Senegal, Sierra Leone, South Sudan, Sudan, Togo, Uganda. | |

All *Khaya* spp. with the exception of *K. comorensis* were assessed as Vulnerable on the IUCN Red List in 1998 on the basis of unsustainable harvest for timber.

The IUCN Red List assessments are currently being updated. If the taxonomic revision of up to six species now treated as the *K. anthotheca* complex is published (see Bouka et al., 2022) they will be assessed as separate species with smaller geographical ranges.

Khaya grandifoliola, *K. ivorensis*, and *K. senegalensis* are currently being re-assessed and are anticipated to remain in the same global threat category or move to a higher threat category (Bouka, in litt., 2022).

The recent reassessment of *K. madagascariensis* categorised the species as Vulnerable based on past population declines of over 30% in three generations as a result of exploitation for timber.

Khaya madagascariensis is endemic to Madagascar and the Comoros (islands of Grande Comore, Mohéli, and Anjouan). It occurs naturally in the northern, central highland and eastern regions of Madagascar including Ambilobe, Analamerana protected area, Antanimihavotra, and Bekolosy. As of 2020, the species had an estimated extent of occurrence of 262,803 km² but an estimated area of occupancy of only 56 km². *K. madagascariensis* was historically more widespread with subpopulations previously recorded in the Malagasy provinces of Antsiranana, Fianarantsoa, Mahajanga, Toamasina, and Toliara, and Nioumachoua Island in the Comoros; it is unclear whether the species has been completely lost from these locations.

Biological and trade criteria for inclusion in Appendix II (Res. Conf. 9.24 (Rev. CoP 17) 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

Commercial exploitation of wild populations of *Khaya* spp. for the international timber market is considered an ongoing, primary threat to the genus. All *Khaya* spp. with the exception of *K. comorensis* were assessed as globally Vulnerable in the IUCN Red List in 1998 on the basis of unsustainable harvest for timber.

As well as being considered threatened at a global level, national assessments note the decline of Khaya spp. as shown below. For Khaya spp., even though there hasn't been an inventory in Ghana (where K. anthotheca, K. grandifoliola, K. ivorensis and K. senegalensis occur) across all ecological zones, it is also the general notion that the populations of the species have declined due to overexploitation such that they meet the CITES listing criteria (Ofori, in litt., 2022).

Khaya anthotheca

Khaya anthotheca has declined greatly over the past 100 years in Uganda and is considered to be vulnerable at a national level. Intense harvesting has led to at least a 50% loss of the population in forests of western Uganda in the past 80 years. *K. anthotheca* is also considered to be threatened in DRC (GlobalTreePortal, 2022). In Tanzania a 45% decline in population is estimated based on seed collection data between 2000–2020, with an estimated 85% decline in use and trade in 94 local timber yards surveyed (Mashimba, in litt., 2022). The Fifth National Report to the Convention on Biodiversity (CBD) for Zambia considers *Khaya nyasica* (currently included in *K. anthotheca*) as locally threatened due to exploitation and as a result, mature trees of these species are rare (Nott et al., 2020). *K. anthotheca* is one of 11 timber species considered of high priority for conservation attention in Angola. Within the country, *K. anthotheca* is concentrated in Cabinda's Maiombe forest, with a very small extent of occurrence (EOO) in Angola, though widely distributed elsewhere in Guineo-Congolian rainforests. There is a need to take urgent action to protect the Cabinda's Maiombe forest, where there is a significant concentration of threatened timber species of high conservation priority. Cabinda's forest may be under increasing legal and illegal logging pressure and most of the historically known Angolan populations of important timbers are located outside the boundaries of the Maiombe National Park which is a transfrontier conservation area (Romeiras et al., 2014). The main timber species harvested and traded in Angola include *K. anthotheca* (Nott et al., 2020). This species has been recorded in the past as nationally threatened in Côte d'Ivoire, and Liberia (WCMC, 1991).

Khaya grandifoliola

Khaya grandifoliola is considered to be endangered in Uganda with intense harvesting leading to at least a 50% loss of the population in forests in western Uganda in the past 80 years. *K. grandifoliola* is also considered to be threatened in Burkina Faso, and DRC (GlobalTreePortal, 2022). It has also been recorded in the past as nationally threatened in Benin (WCMC, 1991).

Khaya ivorensis

Khaya ivorensis is considered to be nationally threatened in Cameroon (GlobalTreePortal, 2022). It is one of 11 timber species considered of high priority for conservation attention in Angola (Romeiras et al., 2014). The main timber species harvested and traded in Angola include *K. ivorensis* (Nott et al., 2020). It has been recorded in the past as nationally threatened in Côte d'Ivoire, Ghana, and Liberia (WCMC, 1991).

Khaya madagascariensis

The reddish-brown wood of *K. madagascariensis* has been highly valued for carpentry, implements, carvings and traditional canoes (Andriamanohera and Rakotoarisoa, 2020). It has been used for columns in traditional palaces in Antananarivo (Maroyi, 2008). There are no recorded exports (legal or illegal). The use is limited to the national level, mainly to produce charcoal. The barks and leaves are used for medicinal plants, and to date, there are no commercial plantations of this species (Ratsimbazafy, in litt., 2022).

Khaya senegalensis

Khaya senegalensis has declined greatly over the past 100 years in Uganda. Intense harvesting has led to at least a 50% loss of the population in forests in West Nile, Acholi, and East Madi in the past 80 years (GlobalTreePortal, 2022). It has been recorded in the past as nationally threatened in Benin (WCMC, 1991). Illegal exploitation is one of the threats to *K. senegalensis* in Sudan.

Trade

Species of the genus *Khaya* produce some of Africa's most valuable timber. For two centuries, these species have all been traded as "African Mahogany" (Bouka Dipelet et al., 2019).

Commercial exploitation of *Khaya* and *Entandrophragma* spp. has "sustained much of the timber industry in Central Africa for several decades", with Meliaceae (including *Khaya* spp.) being the focus of "low levels of high-grade logging" in the region. *K. anthotheca* is also reported as one of several timber species that "dominate the domestic markets" in eastern and southern Africa. The USA, the EU, and China are three key international importers of the genus. Imports of *Khaya* spp. from Cameroon and Ghana have also been reported to enter the USA via Europe.

The major *Khaya* spp. exporting range States are Cameroon, Congo, Côte d'Ivoire, DRC, and Gabon. The volumes exported from Cameroon, Congo and Côte d'Ivoire are shown in Table 2 below.

Table 2. Volumes of *Khaya* spp. exported from Cameroon, Congo, and Côte d'Ivoire 2009–2019. Source: SS (Tables 3, 4 & 5 of Annex 2). Exports given by weight (kg) using conversion figures for exports from Côte d'Ivoire of 1 m³ of sawn timber of *Khaya* = 600 kg (timberpolis.uk).

| Range State | Exports per year (kg) | | |
|----------------------------------|---|--|---|
| | Cameroon | Congo | Côte d'Ivoire |
| Species occurring in range State | <i>K. anthotheca</i> , <i>K. ivorensis</i> , <i>K. senegalensis</i> . | <i>K. anthotheca</i> , <i>K. grandifolia</i> | <i>K. anthotheca</i> , <i>K. grandifolia</i> , <i>K. ivorensis</i> , <i>K. senegalensis</i> . |
| 2007 | 112,476,340 | | |
| 2008 | 716,813 | 6,433,518 | |
| 2009 | 345,016 | 191,187 | 8,611,224,600 |
| 2010 | 2,782,435 | 325,642 | 11,280,204,000 |
| 2011 | 4,242,703 | 574,237 | 4,059,006,600 |
| 2012 | 3,186,445 | 242,551 | 8,177,960,400 |
| 2013 | | 3,321,992 | 6,215,648,400 |
| 2014 | 6,970,473 | 2,153,972 | 4,883,626,200 |
| 2015 | | | 4,023,636,600 |
| 2016 | | | 2,852,368,200 |
| 2017 | | 714,733 | 2,015,328,600 |
| 2018 | | | 1,423,965,600 |
| 2019 | | | 1,526,990,400 |

The volume of *Khaya* exported from DRC 2007–2014 (all commodities combined) amounted to over 86,000 m³. (Mahonghol et al., 2020). This compares with major species exported from the country during the same period, *Wenge* *Millettia laurentii* (~320,000 m³) and *Sapelli* *Entandrophragma cylindricum* (~210,000 m³). The amount of *Khaya* exported appears to have increased significantly in recent years (Table 3).

Khaya ivorensis and *Khaya anthotheca*—included as "Acajou" in the list of the 24 timbers currently harvested in Gabon and *Khaya grandifoliola* is one of the main timbers produced in DRC (timbertradeportal.com). Over the period 2015–2019, China imported Acajou products (HS code 44072920) from Angola, Benin, Cameroon, Central Africa Republic, Congo, Cote d'Ivoire, DRC, Gabon, Gambia, Ghana, Nigeria, and Tanzania (Table 3). The quantities given are by weight (kg).

Table 3. *Khaya* spp. exported to China from African countries by weight (kg). Source: SS.

| Exporter | 2015 | 2016 | 2017 | 2018 | 2019 | Total |
|---------------|-----------|-----------|-----------|-----------|-----------|------------|
| Gabon | 4,207,240 | 3,243,592 | 3,907,526 | 2,278,681 | 3,390,703 | 17,027,742 |
| Cameroon | 1,448,232 | 191,076 | 240,791 | 787,813 | 875,878 | 3,543,790 |
| Congo | | | 20,412 | 921,189 | 817,759 | 1,759,360 |
| Côte d'Ivoire | 108,058 | 70,798 | 124,561 | 7,395 | 94,153 | 404,965 |
| DRC | 4,280 | 14,026 | 83,149 | 149,290 | 92,110 | 342,855 |
| Angola | | 140,050 | | | | 140,050 |
| Gambia | | | 107,676 | | | 107,676 |
| Benin | 46,000 | 58,000 | | | | 104,000 |
| CAR | | | | 74,145 | 25,065 | 99,210 |
| Nigeria | | 24,000 | 23,000 | | | 47,000 |

| Exporter | 2015 | 2016 | 2017 | 2018 | 2019 | Total |
|----------|--------|------|------|------|------|--------|
| Tanzania | 35,570 | | | | | 35,570 |
| Ghana | 34,315 | | | | | 34,315 |

In ITTO's 2017–2018 biennial review of the world timber situation, European consumer countries reporting imports of *Khaya* spp. included Cyprus, Czech Republic, Estonia, France, Latvia, and Malta.

There is a highly significant trade of Khaya spp. between African countries, much of which is poorly documented. African Mahogany species, K. anthotheca together with Entandrophragma spp, are among the key timber resources that are exported from eastern DRC to markets in the East African region. K. anthotheca is one of the species imported into Tanzania from neighbouring countries; K. anthotheca and K. senegalensis are preferred species imported to Kenya from DRC for use in construction, joinery and furniture making; and timber from Khaya spp. particularly K. senegalensis from DRC also transits through Uganda (with small quantities from South Sudan) (Lukumbuzya and Sianga, 2017).

Illegal exploitation of *Khaya* spp. has been documented in several countries, both for domestic use and for export.

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

Khaya spp. are considered to be indistinguishable from one another based on macroscopic features of their wood and also based on microscopic wood characteristics. The wood of *Khaya* spp. is considered to be easily mistaken for that of the CITES-listed *Swietenia* spp.

Within West and Central Africa due to similarities in appearance of the four species, K. anthotheca, K. ivorensis, K. grandifoliola, and K. senegalensis, are usually mixed and traded on the global market (Pentsil et al., 2016).

Imports of *Khaya* spp. into the USA were noted to have increased as the species became a substitute for declining American Mahogany, *Swietenia* spp., which had become more expensive.

Additional information

Threats

High rates of deforestation due to factors such as agricultural and rural development, uncontrolled fires and urbanisation threaten natural *Khaya* spp. Desertification is noted as a threat to *K. senegalensis*. Harvest for traditional medicine was also noted to pose a "serious threat" to natural *K. senegalensis* populations across the species' global range.

Conservation, management and legislation

A range of specific conservation measures and forest management requirements are in place for *Khaya* spp. in different African countries. Minimum exploitable diameters are specified in Angola, Cameroon, Central African Republic, Côte d'Ivoire, DRC, Gabon, Ghana, Mozambique, Sierra Leone, and Tanzania.

Artificial propagation

Although *Khaya* spp. are considered to be "largely undomesticated throughout their natural range", and exploitation of African Mahoganies has largely focussed on natural populations rather than plantations, some plantations have been reported from Benin, Burkina Faso, Côte d'Ivoire, Ghana, Mali, Nigeria, and Togo.

Plantations of Khaya were established in parts of Africa around 100 years ago. In Côte d'Ivoire, the first plantation of Khaya ivorensis was established in 1927. Enrichment planting in degraded forests subsequently showed better growth rates. In tropical Africa, Khaya ivorensis has been planted with success in mixed plantations. In Togo, Khaya senegalensis plantations began in 1918 and in Benin in 1935. Plantations have also been developed in Côte d'Ivoire and Mali. In Côte d'Ivoire, Khaya anthotheca has been planted in degraded or secondary forests. Pure plantations have also been established (Pinheiro et al., 2011).

Khaya plantations have also been established in parts of Australia, Sri Lanka, South-East Asia, and South and Central America.

Khaya senegalensis was introduced into Australia in the 1950s and was planted spasmodically on a small scale until the mid-1970s. Farm forestry plantings and further trials recommenced in the late 1990s, but industrial plantations did not begin until the mid-2000s, based on managed investment schemes. The area established (to 2008) exceeded 4,000 ha and was increasing at a rate of some 2,000 ha per year, almost wholly in the Northern Territory. (Underwood and Nikles, 2009). There are small seed orchards in Australia and a small breeding programme in Queensland (Lott and Read, 2021). In Australia, K. senegalensis was thought to have potential for

musical instrument manufacture but has predominantly been used for pulp logs and sawn logs (Wrigley, in litt., 2022).

Khaya was planted around Darwin in urban settings where it grows to a large size with huge surface roots that can lift roads and kerbing. It retains its leaves giving high resistance to wind and causing significant damage in cyclones. It is used locally to make furniture—but many of the trees that fall down are chipped and used as garden mulch (Leach, in litt., 2022).

K. anthotheca and *K. senegalensis* were introduced to Fiji in the 1900s for experimental cultivation (Wrigley, in litt., 2022).

The forest department of Sri Lanka maintains Khaya plantations. Here, the planted species is *Khaya senegalensis*. All plantations are in the dry zone area of the country. However, annual establishment rates in Sri Lanka are limited by a dearth of domestic seed production. Timbers are supplied to the local market and there is no export. The natural forests of Sri Lanka are conservation forests and there is a huge demand for plantation timber in the country. Nevertheless, there is not much interest in Khaya (Ediriweera, in litt., 2022).

Underwood and Nikles (2009) note that Khaya species are highly susceptible to *Hypsipyla* in Africa but plantings of African Mahogany in Australia and the Asian region have remained relatively free of shoot borer attack.

In Brazil, the introduction of the species is relatively recent. *Khaya ivorensis* is used in reforestation due to its resistance to *Hypsipyla grandella* Zeller, which is the main pest of Brazilian Mahogany and other *Meliaceae*, such as Cedar and *Andiroba*. *Khaya ivorensis* is cultivated in the Brazilian states of Para, Minas Gerais, Goiás, and Mato Grosso, due to the importance of its wood, its price on the international market and, because when established in organised plantations, it shows satisfactory vegetative development (Pineiro et al., 2011). In 2019, the plantations in Brazil were confirmed as *K. grandifoliola*, and not *K. ivorensis* as previously thought. Most of the research that was published citing *K. ivorensis* from Brazilian plantations prior to 2019 related to *K. grandifoliola*. The current plantation area of African Mahogany in Brazil is estimated to be 50,000 ha, predominantly composed of the species *K. grandifoliola* (66%) and *K. senegalensis* (33%). Other African Mahogany species are also planted (*K. anthotheca* and *K. ivorensis*), but on a much smaller scale (Ferraz Filho et al., 2021). In Brazil, the African mahogany plantation area is expected to increase, independent of the species chosen, with possible help from foreign capital investment and with greater interest from rural producers in the diversification of production. The trading of planted African Mahogany timber of Brazilian origin in the national and international market is perceived as a successful goal (Ferraz Filho et al., 2021).

Implementation challenges (including similar species)

The implementation of CITES provisions for valuable African timbers has proved challenging, for example in relation to the development of non-detriment findings (NDFs). It has been suggested that substantial funds may be required to produce NDFs (ATIBT Flash News, 2022).

The volume of re-exports of worked products of Khaya produced in non-African countries could produce a significant implementation burden.

Potential risk(s) of a listing

Commercial exploitation of Khaya has frequently been linked with exploitation of other species that produce similar timber, particularly *Entandrophragma* spp., which has also been traditionally traded as African Mahogany. Both Khaya and *Entandrophragma* are in the *Meliaceae* family as is the genus of “true mahoganies”, *Swietenia*. Listing of Khaya could potentially shift the focus of trade to *Entandrophragma* together with species of *Guarea* and *Lovoa*. Various species of *Entandrophragma* are included on the IUCN Red List as Vulnerable or Near Threatened.

There is a risk that permitting requirements resulting from the listing might favour plantation grown timber from non-African countries rather than the continued development of sustainable forest management for timber production within Africa.

Potential benefit(s) of listing for trade regulation

Efforts to improve sustainable forest management and promote certification are underway at national and regional levels. CITES listing could reinforce these efforts. Furthermore, it could establish necessary controls required for more effective data gathering on trade and support regional collaboration in data sharing and transparency as recommended by Mahonghol et al., 2020.

Other comments

Khaya is within the scope of certification schemes as the FSC Certificates Public Dashboard records certificates for the genus relating to Cameroon, Congo, Gabon, Ghana, Mozambique, and South Africa.

Some rural forest communities depend extensively on resources from Khaya for subsistence and income generation. Many natural products are obtained from the trees. Bark extracts have been used as astringents for wounds. The bark of *K. senegalensis* has been traditionally used in local leather industries by the people of the savannah zones of Africa for the tanning of leather, because of its rich red colour. The bark of most species of African Mahoganies contains low-quality gum Arabic, and the bark of African Mahoganies contains many medicinal properties that are used in the treatment of certain tropical diseases. In Ghana, the bark of *K. ivorensis* is a key ingredient in the production of an alcoholic beverage called bitters. The demand for bitters is very high and attracts premium prices in the alcoholic beverage market. Traditionally, mahogany seed oil has been used for a wide range of purposes, most notably as a disinfectant against infection (Danquah et al., 2020.)

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Amend the Annotation (#4) to the listing of Orchidaceae included in Appendix II with the addition of new paragraph g), to read: “(g) finished products packaged and ready for retail trade of cosmetics containing parts and derivatives of *Bletilla striata*, *Cycnoches cooperi*, *Gastrodia elata*, *Phalaenopsis amabilis* or *Phalaenopsis lobbii*”

Proponent: Switzerland

Summary: The proposal concerns the exemption of finished products packaged and ready for retail trade of cosmetics containing parts and derivatives of five orchid species (*Bletilla striata*, *Cycnoches cooperi*, *Gastrodia elata*, *Phalaenopsis amabilis* and *Phalaenopsis lobbii*) by adding a new paragraph (g) to existing annotation #4, applying to the Appendix II listing for Orchidaceae.

Studies of various orchid species by Switzerland and China concluded that all five species to which the annotation would apply are artificially propagated in large numbers to supply the cosmetic and personal care industry. There was no evidence that wild harvested plants were used in the manufacture of such products. In consequence wild populations would not be detrimentally affected by the exemption proposed.

The proposal states that “it is highly unlikely that any wild-harvested specimens of these species are used by this industry, which relies heavily on a regular and consistent supply of specimens of uniform quality, and this can only be achieved with large-scale artificial propagation.” A rapid review of the CITES trade data confirms that most of the trade in cosmetics in the five species has been reported as of artificially propagated source. Records also show that cosmetics are not the first specimens to appear in international trade as exports from range States. From 2009 to 2020, there were only a few records of exports by range States of cosmetics containing *Bletilla striata* (Republic of Korea) and *Gastrodia elata* (Republic of Korea and the Democratic People’s Republic of Korea) amounting to less than 15 kg each, all reported to be produced from artificially propagated specimens.

At CoP18 the Parties adopted a definition of cosmetics. However, the Secretariat raised concerns that there was still some ambiguity between the potential overlap between cosmetics and externally applied medicines, which could lead to ambiguities in the categories of products that are included in, or excluded from, the proposed definition. None of the five orchids involved in this proposal appear to be traded as externally applied medicines, so this ambiguity would not affect the implementation of the proposed amendment.

At its 25th meeting the Plants Committee noted Switzerland’s intention to submit a proposal on this matter. Initially, the intention was to specify that the exemption would apply to source code A or Y. The 74th Standing Committee suggested that the packaging should clearly state the full scientific name and a declaration of “artificially propagation”, however, outreach to the cosmetic and personal care industry concluded that this would be impracticable and challenging to implement.

Analysis: Considering that cosmetics containing parts and derivatives of the five orchid species proposed to be exempted from CITES regulations are all sourced from artificially propagated specimens, and that only cosmetics containing two of the species (*Bletilla striata* and *Gastrodia elata*) have been exported by range States in low quantities, the amendment proposed to Annotation #4 would appear not to pose a threat to the conservation of wild populations of these species. It is in line with Res. Conf. 11.21 (Rev. CoP18) and Res. Conf. 9.24 (Rev. CoP17), which recommend Parties ensure that annotations to listings in the Appendices include those specimens that first appear in international trade as exports from range States and that dominate the trade and the demand from the wild.

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

Bletilla striata (Thunb. ex A.Murray) Rchb.f.

Phalaenopsis amabilis (L.) Blume.

Phalaenopsis lobbii (Rchb.f.) Sweet

Reference: Roberts *et al.* (2001).

Gastrodia elata (Blume, 1856)

Cycnoches cooperi (Rolfe, 1913)

Discussion

The proposal concerns the exemption of finished products packaged and ready for retail trade of cosmetics containing parts and derivatives of five orchid species (*Bletilla striata*, *Cycnoches cooperi*, *Gastrodia elata*, *Phalaenopsis amabilis* or *Phalaenopsis lobbii*) by amending the annotation to the Orchidaceae Appendix II listing #4 with an additional paragraph g) to that effect.

Based on studies of various orchid species by Switzerland and China, it was concluded that all the species to which the annotation would apply are artificially propagated in large numbers to supply the cosmetic and personal care industry and there was no evidence that wild harvested plants were used in the manufacture of such products; wild populations would not be detrimentally affected by the exemption proposed. *The Plants Committee noted Switzerland's intention to submit a proposal at its 25th meeting (PC25 Doc. 37).* Initially, the intention was to specify that the exemption would apply to source code A or Y. The 74th Standing Committee suggested that the packaging should clearly state the full scientific name and a declaration of "artificially propagation". However, outreach to the cosmetic and personal care industry concluded that this would be impracticable and challenging to implement, in particular with regard to the labelling requirements needed to adhere to existing UK Cosmetics Regulation and the EU Cosmetic Products Regulation.

The proposal stated that "it is highly unlikely that any wild-harvested specimens of these species are used by this industry, which relies heavily on a regular and consistent supply of specimens of uniform quality, and this can only be achieved with large-scale artificial propagation." A rapid review of the CITES trade data confirms that the majority of trade in cosmetics in the five species has been reported as of artificially propagated source.

At CoP18 the Parties adopted a definition of cosmetics as "Any product or mixture of products which is applied to an external part of the body only (e.g., skin, hair, nails, genitals, lips or teeth or the mucous membranes of the oral cavity) with the intent to clean, odourise, change the appearance or protect. Cosmetics may include the following: make-up, perfume, skin cream, nail polish, hair colourants, soap, shampoo, shaving cream, deodorant, sunscreens, toothpaste." (see Notification No. 2021/044 Annex I). However, the Secretariat raised concerns that there was still some ambiguity between the potential overlap between cosmetics and externally applied medicines, which might lead to ambiguities in the categories of products that are included in, or excluded from, the proposed definition (CoP18 Doc. 102).

Taking into account Res. Conf. 11.21 (Rev. CoP18) and Res. Conf. 9.24 (Rev. CoP17), which recommend Parties ensure that the annotations to listings in the Appendices include those specimens that dominate the trade and the demand from the wild, but also that first appear in international trade as exports from range States, the trade recorded in the CITES Trade Database for Bletilla striata, Cycnoches cooperi, Gastrodia elata, Phalaenopsis amabilis and Phalaenopsis lobbii from 2010 to 2020 and the role of the species range States in the trade was reviewed:

Bletilla striata

Distribution: China, Democratic People's Republic of Korea, Hong Kong SAR, Japan, Myanmar, and Republic of Korea.

Range exporting countries of cosmetics containing the sp: Republic of Korea has registered direct exports (16 transactions), all from artificially propagated specimens for commercial purposes. Republic of Korea has not reported trade records to the CITES Secretariat, it has been the importers, mainly Japan, that have recorded imports of around 14 kg of cosmetics containing the species.

The vast majority of trade in the species and all reported imports of cosmetics are reported as of artificially propagated source. Small amounts of medicines have been reported in trade mainly from artificially propagated

sources, and the only range State with direct export records is Myanmar, with only two transactions recorded in this period totalling 15 kg.

Cycnoches cooperi

Distribution: Brazil, and Peru.

Range exporting countries of cosmetics containing the species: No exports of cosmetics containing this species from Brazil and Peru have been recorded. They only export live individuals and all are reported as artificially propagated.

All trade reported in the period has been in artificially propagated specimens, with the main derivatives in trade being extract, oil and, to a lesser extent, cosmetics. Live plants are also seen in trade.

Gastrodia elata

Distribution: Bhutan, China, Democratic People's Republic of Korea, India, Japan, Nepal, Republic of Korea, Russian Federation, and Taiwan POC.

Range exporting countries of cosmetics containing the species: The Republic of Korea and the Democratic People's Republic of Korea have exported cosmetics containing this species and originating in their countries—only one transaction each (12.6 kg and 2 g respectively) from 2010 to 2020 and registered as artificially propagated. Japan is a re-exporter of cosmetics from specimens of *G. elata* imported from these two Parties, but also in very small amounts. Medicines are often exported by China, all recorded as artificially propagated.

All trade in cosmetics from any other Party has been reported as of artificially propagated source. CITES trade data show not insignificant amounts of trade in medicinal products reported by importers as seized or confiscated, with China as the main exporter. However, medicines containing this species are not to be externally applied and are traded as capsules, so no confusion would arise from the proposed exemption.

Phalaenopsis amabilis

Distribution: Australia, Brunei Darussalam, Indonesia, Malaysia, Papua New Guinea, Philippines, and Viet Nam.

Range exporting countries of cosmetics containing the species: No cosmetics containing this species have been reported as exported from range States. Exports of this species from range States (Malaysia and the Philippines, only one transaction from Australia) consist of live specimens recorded as artificially propagated.

Apart from 80 live wild plants and 0.4 g of cosmetics with an unspecified source, all other trade in *P. amabilis* was artificially propagated. Extracts and cosmetics dominated the reported trade in derivatives, with large numbers of live specimens too (over 1.9 million).

Phalaenopsis lobbii

Distribution: China, India, Myanmar, and Viet Nam.

Range exporting countries of cosmetics containing the species: No cosmetics containing this species were reported as exported from range States. Exports of this species from range States (Myanmar and India) have consisted of live specimens recorded as artificially propagated.

Between 2011 and 2020 no trade in cosmetics was reported from any Party. Live trade made up the majority of reported imports, although this only amounted to 1,415 in total. Extracts were reported by exporters only amounting to 11 kg. Apart from five live plants reported as seized/confiscated, all trade was reported as of artificially propagated source.

Other comments

At CoP19 Parties are invited to consider the adoption of draft decisions which direct the Secretariat to, among others, undertake an assessment of the conservation impacts of exempting derivatives and/or finished products of certain Appendix II listed orchid taxa from CITES regulations through amendments to annotation #4 (see CoP19 Doc. 86). However, this document also notes that the Plant Committee recommended to SC74 that "Switzerland to be encouraged to consider submitting its proposal to the 19th meeting of the Conference of the Parties".

References

- CoP18 Doc. 102. (2019). Annotations for Appendix II Orchids.
- PC25 Doc. 37. (2020). Products containing specimens of Appendix II Orchids.
- CoP19 Doc. 86. (2022). Products containing specimens of Appendix II Orchids.
- Notification No. 2021/044 Annex I: Guidelines for the preparation and submission of CITES annual reports (May 2021) – Contents.

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