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Mixed protection of threatened species traded under CITES

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Summary

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) regulates international legal trade to prevent detrimental harvest of wildlife. We assess volumes of threatened and non-threatened bird, mammal, amphibian, and reptile species in CITES-managed trade and how this trade responded to species changing IUCN Red List categories between 2000 and 2018. In this time, over a thousand wild-sourced vertebrate species were commercially traded. Species of least conservation concern had the highest yearly volumes (excluding birds), while species in most Red List categories showed an overall decrease in reoccurrence and volume through time, with most species unlikely to reoccur in recent trade. Charismatic species with populations split-listed between Appendices I and II were traded in substantially lower yearly volumes when sourced from more-threatened Appendix I populations. Species trade volumes did not systematically respond to changes in Red List category, with 31.0% of species disappearing from trade before changing category and the majority of species revealing no difference in trade volume pre- to post-change. Just 2.7% (12/432) of species volumes declined and 2.1% (9/432) of volumes increased after a category change. Our findings highlight non-threatened species dominate trade, but reveal small numbers of highly threatened species in trade and a disconnect between species trade volumes and changing extinction risk. We highlight potential drawbacks in the current regulation of trade in listed species and urgently call for open and accessible assessments—non-detriment findings—robustly evidencing the sustainable use of threatened and non-threatened species alike.

Introduction

International wildlife trade spans the tree of life, involving thousands of species and millions of individuals per year [1,2]. Effective management of wildlife trade is a necessity for human health, livelihoods, and species persistence. This management requires multifaceted processes, including population assessments, global economic investment, law enforcement, and livelihood considerations along the supply chain [3].

For over 40-years, the legal international trade in many species has been regulated by the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). As a binding international agreement, CITES mandates the protection of "*wild fauna and flora against over-exploitation through international trade*" [4]. Since 1994, CITES Parties have applied the precautionary principle [5], advocating the prohibition of trade that threatens any negative impacts on species, even where there is scientific uncertainty regarding the severity of impacts [6]. CITES lists species in two Appendices with differing constraints on trade (plus Appendix III where Parties seek cooperation to prevent unsustainable trade). Appendix I prohibits commercial trade in species threatened with extinction that are or may be affected by trade (except in special circumstances, e.g., captive breeding), while Appendix II covers species that may become threatened if trade is not appropriately managed (plus look-alikes that could be misidentified as a listed species). Where different populations of a single species face varied levels of threat from trade, they can be split-listed between Appendix I and II, aiming to prevent detriment to at-risk populations while allowing sustainable use of others.

Parties to the Convention are required to only allow the export of Appendix I and II species (or populations) after a positive Non-Detriment Finding (NDF), and only then in the volumes evidenced to be non-detrimental. There is currently no central repository nor peer-reviewed assessment of NDFs outside of Parties own scientific authorities (excluding species-specific quotas set directly by the Conference of Parties or Scientific Committees). Given that NDFs are the basis for legally trading CITES-listed species their accuracy is critical, especially since CITES trade should be sustainable and this rests predominantly on NDF's. However, some self-regulated NDFs have been criticized for lacking evidence, incorrectly affirming sustainability, and facilitating detrimental trade [7]. Given CITES' central role in the legal international trade, appropriate processes to prevent harmful trade are paramount.

The presence of threatened species (Vulnerable [VU], Endangered [EN], or Critically Endangered [CR], as defined by the International Union for Conservation of Nature-IUCN) in trade does not inherently equate to trade-induced threat nor unsustainable trade. However, trade in threatened species can directly drive population losses and inflate extinction risk [8]. Trading threatened species can also compound concurrent non-trade threats. For instance, avifaunal species primarily threatened by deforestation suffer exacerbated population declines when exploited for the cage bird trade [9], while threatened species with inherently small populations have increased risk of stochastic extinction potentially exacerbated by harvesting for trade [10]. A non-detriment finding in such cases must indicate that the removal of individuals from an already threatened population will neither further threaten that population nor exacerbate synergistic threats [11]. Anecdotal evidence of trends or abundances cannot accurately forecast the impact of compounded threats, instead requiring complex consideration and offtake modelling [12].

Where high levels of potentially unsustainable trade have already occurred, the Parties have previously overlooked the externalities and implications of decisions. Asian pangolin species were historically threatened by high levels of both legal and illegal trade, thus triggering their inclusion in the Review of Significant Trade (RST) process in 1988 [13]. Consequently, zero-export quotas were established for all wild-sourced Asian species in 2000 at CoP11. This reduced wild-sourced legal trade, but was ineffective at tackling the illegal trade threat. It was not until 2010 this was further addressed, and until 2016 that the Parties again paid concerted attention to both legal and illegal trade by issuing a reporting mandate for all Parties to submit data on illegal pangolin trade [13]. Similarly, when species face a multitude of threats an understanding of these is essential. The Appendix II-listed *Arapaima gigas* is concurrently threatened by habitat degradation, by-catch, and overfishing for local subsistence and aquaculture, with current populations and trends unknown [7]. This paucity of baseline data and the magnitude of threats led to scepticism that positive NDFs for the species were evidence based, despite its presence in international trade [7], and local extirpations have occurred outside of management areas [14]. Considering the negative externalities and interactions between trade and non-trade threats is crucial when determining offtake and policy [15].

Understanding and effectively managing legal wildlife trade is a conservation priority and global necessity to achieve wider sustainable-use and development goals. We apply a multi-level Bayesian modelling framework to provide a data-driven assessment of patterns of threat (as defined by IUCN Red List categories) in the wild-sourced, commercial trade in CITES-listed vertebrate species between 2000 and 2018. We first hypothesise that trade volume under CITES should be dominated by non-threatened species as extinction-threatened species are less likely to demonstrate the requisite positive NDF and, where they do, it would likely be for smaller numbers of individuals. Where species populations are split-listed, we hypothesise more threatened Appendix I populations to be less likely to appear in trade and when they are, it would be in smaller volumes. Lastly, we hypothesise that proactive, precautionary trade management under CITES would be responsive to species becoming more threatened (as assessed by IUCN Red list category changes). We hypothesise this regardless of whether a Red List change was due to trade threat, since species becoming rarer due to any driver are less likely to endure the previous levels of exploitation and thus the NDF recommendation would likely be for smaller volumes than it would be absent other threats.

Results and Discussion

Threatened species in trade

Birds, mammals, amphibians, and reptiles in trade are dominated by least concern (LC) species, with ten or fewer EN or CR species from each taxa present annually since 2000 (Figure 1A-D). Most species (47.6%, 488/1025) were traded for the first time as LC, with 13.7% (140/1025) classed as threatened (VU, EN, CR) when first traded (Figure 1A-D).

On average, IUCN Red List categories showed either decreasing or uncertain trends through time for probability of occurrence in trade (*hu*) and volumes when traded (*mu*) (Figure 1E – H). For birds, trade occurrence and volume of LC and NA (Not evaluated + DD categories) decreased over time (Figure 1E, Table S2), reflected in steep declines in their joint estimates (Figure 1I), whereas trade occurrence and volumes for NT, VU, EN, and CR remained stable (Figure 1E). Here, volumes were comparatively low over time (Figure 1I). For mammals, LC and VU had decreasing presence in trade through time (Figure 1F and J, Table S2), while trends for all other categories remained stable (Figure 1F) at similar volumes (Figure 1J). For amphibians, trade volumes of threatened and NA groups (Figure 1G and K, Table S2), and trade presence of EN species decreased through time (Table S2), whereas LC and NT had increasing and stable volumes in trade, respectively (Figure 1K). Similarly, CR, EN, and NA reptiles showed decreasing volume trends (Figure 1H and L). LC reptiles had increasing, and NT and VU reptiles had stable volume trends over time (Figure 1L).

These results support our hypothesis that trade under CITES is dominated by non-threatened species. Nevertheless, the presence of threatened species in trade since 2000 necessitates rigour in evidencing non-detriment, especially for those at highest risk of extinction [8]. In specific instances, trade has proved an effective conservation management tool, especially where local collectors and stakeholders are incorporated as species managers [16]. Underpinned by federal regulation designating '*Threatened'* status under the US Endangered Species Act in 1987, persistent trade, monitoring, and management of American alligator (*Alligator mississippiensis*) led to increasing wild populations (currently non-threatened; LC) and large economic returns for stakeholders [17]. Developing sustainable use thus has the potential to protect wild populations and incentivise conservation, but this must be evidenced and enforced.

Overall, 54.2% (504/930) of species commercially traded from a wild source that were still listed in the Appendices in 2018 had median estimated volumes below 1 in 2018, suggesting that the majority of species across taxa and IUCN categories are no longer traded. Despite the richness of CITES-traded birds since 2000, the majority of these species (76.3%, 305/401) had estimated median volumes less than 1 in 2018 (Figure 2A). Only 6.7% (27/401) of bird species were estimated to still occur in volumes >100, and only Orange-winged Amazon (*Amazona amazonica*), Red-fronted Parrot (*Poicephalus gulielmi*), and Senegal Parrot (*Poicephalus senegalus*) (0.7%, 3/401) occurred in volumes >1000, each popular in pet trade [18,19]. Similarly, 42.1% of mammal species (85/202) had estimated median volumes less than 1 in 2018 (Figure 2B), but a larger proportion of species traded in higher volumes, with 13.4% (27/202) estimated in volumes >100 and 5.0% (10/202) in volumes >1000 (Figure 2B). This includes VU White-lipped Peccary (*Tayassu pecari*), whose populations are declining and threatened by a combination of subsistence and commercial hunting, deforestation, and fragmentation [20].

Despite the relatively low number of amphibian species in trade, 34.1% (15/44) are estimated in volumes >100 and three LC Malagasy *Mantella* (6.8%, 3/44) in volumes >1000 (Figure 2C). Reptiles have 42.0% of species (119/283) estimated at volumes >100 and 18.4%

(52/283) in volumes >1000 (Figure 2B). Of these, 67.3% (35/52) were LC species. However, the VU Southeast Asian Box Turtle (*Cuora amboinensis*) was traded in volumes >17,000 WOE's annually since 2000, and in 2020 was reassessed as EN due to "*widespread intensive exploitation*" for pets, food, and traditional Chinese medicine [21].

Only 19.7% of species (183/930) retained a high probability of reoccurring in recent trade (in 2018, *hu* > 0.9) and 62.3% (114/183) of these species were classed as LC. Why species appear and disappear from trade remains unclear for the majority of cases. Attempts to predict which species may be traded in the future have used phylogenetic and trait-based interpolation (e.g.,[2]), but linking this to real-world drivers remains a research frontier. Predicting trade volumes is an even greater challenge, particularly considering future unknowns – including zero quotas, sudden novel demand, economic development, and societal change, all occurring across regional to international scales [22].

Across taxa, on average LC species had the highest median volumes (except in birds). LC mammal and amphibian species were traded in substantially higher volumes than CR, EN, or VU species (Table S3, Figure 2F and G), while LC reptile species were traded in substantially higher volumes than CR and EN species (Table S3, Figure 2H). Volumes traded for birds remained low across all Red-list categories (Table S3). These results (Figure 1 and 2) suggest that non-threatened species dominate CITES trade in richness, reoccurrence, and volume.

The low reoccurrence and volume of most species in trade could result from at least three starkly contrasting drivers. First, altered supply of species owing to overexploitation and reduced accessibility; for instance, in southern Sumatra, extensive field surveys revealed several threatened, sought-after species for the cagebird trade were depleted across a remoteness gradient [23]. Second, changing demand, where preferences drive changes in demand; for example, songbird ownership in Java has seen a decadal shift to non-native species [24]. Third, effective national or international legislative protection can remove or limit trade, such as the EU wild-caught bird import ban [25] – although such approaches often do not stop trade entirely and may shift global trade patterns (both spatially and to illicit forms) [26]. Trade will be further influenced by other interconnected regional to international factors, such as supply and demand infrastructure, economic development, and social change.

Managing differentially threatened populations

Split-listed species represent some of the most charismatic megafauna traded, including Southern white rhinoceros (*Ceratotherium s. simum*), African lion (*Panthera leo*), African elephant (*Loxodonta africana*), and Nile crocodile (*Crocodylus niloticus*). For all nine split-listed species evaluated, estimated median traded volumes for the Appendix I populations were lower than for Appendix II populations in 2018 (Figure 5A, *median difference* = -16.11, 90% HDCI: -63.77 to -0.91, *pd* = 99.90%). For crocodilian species, this difference in volume was at least three orders of magnitude (Figure 3A). Overall, Appendix I and Appendix II populations show stable trends through time in probability of trade occurrence (Figure 3A and C) and volume when traded (Figure 3B and C). However, Appendix I populations retained median probabilities of reoccurrence less than 0.16, whereas Appendix II populations always had a probability greater than 0.99. In 2018, Appendix I populations are estimated to be 92% (90% HDCI -1.00 to -0.79, *pd* = 99.83%) less likely to be present in trade.

This indicates threatened populations of split-listed species are less likely to be traded plus likely to be traded in lower volumes. Split-listing has clear potential to achieve synergistic benefits, protecting at-risk populations while providing livelihood benefits and legal supply [27]. Robust mechanisms are needed to differentiate between populations of a species in trade, and traceability is complex to guarantee and enforce [28,29]. Therefore, while split-listing suggests CITES policy can provide population-specific protection and management, the tools and infrastructure to identify individuals to specific populations are absent for many taxa. Species can have populations that are better or worse suited to utilisation (including split-listed species), but for the vast majority of species spatial variation in suitability for harvesting between populations is not considered. Thus, relatively common species could experience local extirpations if smaller declining populations are overexploited, even if their global population trends are stable [30]. Considerations of the resilience of individual populations of species through space and time remains a research and policy frontier. By including only species that were wild-sourced and commercially traded at least once in both Appendices during our timeframe, we exclude certain split-listed species (e.g. *Vicugna vicugna*) only traded under Appendix II (thus the more threatened Appendix I populations were not traded at all) [27], suggesting that the effectiveness of split listing may be even greater.

CITES response to changing extinction risk

The final key step in examining CITES-listed trade considers whether trade responds to changes in IUCN Red List categories. Species change Red List category to reflect updated knowledge of populations, threats, or previous errors. Between 2000 and 2018, 395 wild-sourced species commercially traded under CITES changed or were given their first Red List assessment, equating to 432 species-level category changes (35 species changed Red List category more than once). There was substantial variation in volumes traded pre- to post-change in Red List category (Figure 4A – H). However, contrary to our hypothesis that species would be less likely to reoccur or occur in smaller volumes after becoming threatened, changes in volume were not broadly associated with changes in Red List category, irrespective of change type, and individual species responses varied greatly (Figure 4A - D).

On average, only birds and reptiles revealed any pre- to post-changes in volume, with birds that became non-threatened slightly increasing in volume (median difference = 0.3, 90% HDCI: 0.0 to 0.7, *pd* = 99.84%) and reptiles that became threatened decreasing in volume (median difference = -8.0, 90% HDCI: -32.6 to -0.11, *pd* = 99.76%) (Figure S1A-D and Table S4). Similarly, there was limited evidence that changing category relative to species that did not change category led to a difference in volume for the average species (Figure S1E-H and Table S4). Birds that became non-threatened were estimated to reappear in higher volumes than those that did not change (Table S4). Mammals and reptiles that stayed threatened and amphibians that stayed non-threatened or became threatened were estimated to be traded in lower volumes in 2018 than those that did not change category (Table S4).

Of individual Red List category changes, 45.8% (198/432) showed minimal change in traded volume pre- to post-change ($-1 < \text{median difference} < 1$, clustered on the dashed zero lines in Figure 4). This can largely be attributed to 31.0% of changes (134/432) having a median pre- and post-change volume of < 1, suggesting the species presence in trade had already declined to near zero before an Red List category change (stopped being traded) or while CITES-listed had not yet been traded. Also contributing to the apparent lack of change in volumes are species that remained traded at similar volumes pre- to post-change. For instance, Madagascar Big-headed Turtle (*Erymnochelys madagascariensis*) had no identifiable change in volume after a reassessment from EN to CR in 2008 (median difference = -0.23 90% HDCI: -23.7 to 22.6). Similarly, Saker falcon (*Falco cherrug*) remained traded in the hundreds (median difference = 273.8, 90% HDCI: -261.4 to 936.6) following reassessment from LC to EN in 2004 with "*inadequately controlled capture for the falconry trade*" given as explanation for the reassessment [31]. As hypothesised for many species that became or stayed non-threatened, there was no change in volumes post-change. For example, after Northern red-shouldered macaw (*Diopsittaca nobilis*) was first assessed in 2014 (as LC) it remained traded in the hundreds (median difference = 225.3, 90% HDCI: -207.7 to 749.3) with trade not considered to be a threat [32,33].

In 2.7% (12/432) of species-level changes, volumes fell pre- to post-change (negative lower and upper 90% HDCI bounds). For example, Grey parrot (*Psittacus erithacus)* median volumes decreased by 3588.3 (90% HDCI: -8419.9 to -437.7) after reassessment from VU to EN in 2016. Conversely, only 2.1% (9/432) of species-level changes were associated with increased volumes, the majority of which had stayed or became non-threatened. For instance, Common long-tailed macaque (*Macaca fascicularis*) volumes increased sharply immediately after reassessment from NT to LC in 2008 (difference = 2833.4, 90% HDCI: 430.0 to 6696.7). Volumes have since decreased, with the species reassessed in 2020 as VU owing to declines from hunting (local consumption) and extraction for international trade (taken for breeding or directly exported) [34].

It is important to consider both these static changes in volumes pre- to post-change, with the associated changes in volume and occurrence trends through time [35]. Such an approach is necessary as volumes may remain constant just after a change, but there may be longer-term changes in volumes through time, e.g., post-change the volumes may gradually decrease. We considered these trend differences in both presence (*hu*) and volume (*mu*) using species-level trend coefficients. Only five species category changes (1.2%) displayed negative occurrence-trend (*hu*) differences (90% HDCI below zero), i.e., a species is decreasing in occurrence probability more rapidly post- than pre-change (Figure 4E – H). Thirty-eight species (8.8%) had substantial positive difference in occurrence trends (HDCI above zero), suggesting species presence trends were more positive after a change than before. For example, although Golden mantella (*Mantella aurantiaca*) decreased substantially in traded volume when reassessed from VU to CR, it shifted from a declining occurrence trend pre-change to an increasing trend post-change. Care must be taken with interpretation where species-level reoccurrences asymptotically approach either 0 or 1 e.g. a pre-change trend towards zero (negative trend), and a post-change trend asymptotic with zero (flat trend), would also have a positive trend difference (post-change trend – pre-change trend), hence trend changes must be cross-referenced with the absolute values. Only 0.4% (2/432) of species saw negative volume trend-differences, i.e., volumes falling faster post- than pre-change. For Southern lechwe (*Kobus lechwe*), volumes were low (<100 WOE's year^{-1}) and stable prior to reassessment from NT to LC in 2008, but immediately post-change volumes peaked in the hundreds then rapidly fell. Conversely, 2.7% of species (12/432) had positive volume-trend differences suggesting volumes increased at a faster rate post-change; in all but one case (VU to NT), the move was from not assessed to LC or NT.

Recently, numerous species have been reassessed by the IUCN into a higher threat categories with trade given as justification following rigorous assessment of species populations or threats and open-access, peer-review [36,37]. Despite IUCN assessments reflecting changing trade impacts, we find an unclear response from CITES. NDFs are not publically available, in part owing to a lack of central data-basing (excluding 36 NDFs and 29 NDF Guidelines), making it impossible to scrutinise the evidence or methods used in creating an NDF. Updating NDFs in light of changing threats or population trends is a key step for proactive trade management. Crucially, this could reduce the risk that species highly threatened by anthropogenic stressors are additionally traded and suffer subsequent Allee effects or stochastic extinction. For example, vulture species critically endangered by poisoning are still being traded [38]. Well-managed trade in threatened species is crucial to long-term conservation goals, making sharing and building on successful NDF approaches of utmost importance [12,39].

Our analyses are limited to the legal wild-sourced commercial trade, which is regulated, quantified, and aims to promote sustainable use. However, this represents only a fraction of trade, overlooking all illicit international trading and all legal or illegal within-country trade. Patterns of threat in illegal trade could plausibly run opposite to the patterns we find in legal trade [40]. The same could be true for captive trade, as the general volume and presence decline across species in the wild-sourced trade (Figure 1) could be indicative of a shift to captive sources, as found in previous studies [1].

Implications for CITES-regulated trade

Our study highlights that less-threatened species, including split-listed Appendix II populations, dominate CITES trade in richness, occurrence, and volume. However, we find limited evidence that when individual species became or remained threatened they were less likely to appear in future trade or be traded at lower volumes. Legal trade in threatened species places considerable onus on the accuracy and robustness of the CITES NDF procedure. A process that has been effectively used to bolster conservation efforts and species recovery (e.g. for Southern white rhino [41] and American alligator [17]), but has also been plagued with controversy concerning its rigour and transparency [7,42,43]. Since 2008, Thailand has been subject to a CITES Review of Significant Trade (RST), where the NDF's have been queried for four heavily exported seahorse species to assess whether such export was evidenced as non-detrimental. Ultimately, Thailand was unable to produce positive NDF's for the species [39] and their trade was classed as "*urgent concern*" by the CITES Animal Committee [44]. Compounding the validity of NDF's is whether they exist at all. Work examining African rosewood (*Pterocarpus erinaceus*) trade from Ghana found no up-to-date scientific NDF, despite this species' presence in trade – a non-compliance issue in clear contradiction of the Convention [45].

A 2020 CITES Report of the Secretariat on Non-detriment Findings examined the 36 publically available NDFs, concluding standards vary greatly [46]. Only 44% (16/36) fully considered non-trade threats and the overall threats to species, 42% (15/36) considered species-specific biology or life-history factors influencing their vulnerability, 36% (13/36) clearly considered the precautionary principle, and just 17% (6/36) fully considered historical and current patterns of harvest and mortality. A single NDF considered the role of the species in the ecosystem, and no NDF's reached three or more robustness targets ("*good*" data, multiple indicators, triangulation, or peer-review/stakeholder consultation). Given this and our results, we urgently call for greater transparency and gradual transition to publishing all NDFs.

Processes such as the RST exist to identify and respond to species/populations at risk of unsustainable CITES trade, but this makes two problematic assumptions: 1) that unsustainable trade can be recognized by other Parties; and 2) acting *after* trade has occurred is an appropriate response. Discerning unsustainable or ecologically harmful trade from trade data alone (i.e., independent of population-level data) is almost impossible, yet that is the main data source used to identify species for the RST [47]. We posit that unsustainable offtake for trade should be recognized *prior* to its occurrence. This could be achieved through a review of NDF documents confirming the analyses include robust evidence, demonstrate that relevant ecological information was used, and ultimately justify offtakes appropriate to ensure species survival. Open-access NDF's would be a step closer to this. There are clear logistical challenges, primarily that under the current Convention there is no provision for making NDF data, methods, or results available and any change would require a considerable amendment to the Convention text. Similarly, there are risks to sharing species data openly, but geographic data can be anonymized [48]. Additionally, there are major challenges to sourcing the necessary expertise and finances to perform NDF reviews. Reviewers could potentially be found within the research community or Scientific Authorities of other Parties, but acquiring the funding and standardising the process would be non-trivial. While ideally the process would be managed within individual Parties, the initialisation and oversight would need to come via the Animal and Plant Committees. Given the challenges, we suggest this would be developed gradually, starting initially with sharing and reviewing methods, and culminating in results being open access. As individual Parties have autonomy to implement the Convention and make NDFs as they see appropriate, the review would represent a judgement of the validity of an NDF, flagging where it is inaccurate. Exporting Parties could appeal with evidence if they believed the review was in error, and in such cases decisions to sanction or not could come from a panel from the Animal and Plants Committees. Any trade undertaken by a Party that was justified by an inaccurate NDF could then be viewed in breach of the Convention (lack of a valid NDF) and subject to follow-up action including sanctions. However, initially these reviews would be used to build capacity and develop consistent methods; trade could be allowed under inaccurate NDFs for a set number of years while processes and methods were fully developed.

Making space for controversy and debate

Wildlife trade science is diverse. It spans those focusing on protecting species from overexploitation to those working to ensure continued livelihoods. From researchers utilising large quantitative datasets to those using qualitative evidence. And from independently supported researchers to those at least in part supported by the trade industry (e.g., the luxury fashion industry [49]), with associated risks of the "science for profit" model [50], or by animal welfare groups and their potentially anti-trade stance. Such a diversity of researchers offers great potential to overcome one of the greatest challenges faced by biodiversity and humanity – how to deliver sustainable offtake that protects species whilst delivering on societal needs. At present, this diversity has resulted in increasingly entrenched and polarised viewpoints about how to assess and manage wildlife trade.

Some of the approaches, recommendations, and discussion points we highlight contradict the opinions of others, in particular our integration and interpretation of the IUCN Red List with CITES trade data, and the suggestion of reforms to CITES NDF policies. We have emphasised that threatened species (VU, EN, or CR) can appear in wild-sourced trade and not be threatened as a result. However, we need greater consideration of how concurrent threats to species are considered. It is robustly evidenced that habitat loss and extraction (for trade or consumption) [9,51,52], and climate change and extraction [53] can drive synergistic declines in target species. There is a need to evidence, not assume, that exploiting a species for which trade is not the primary driver of loss will not *further* contribute to declines. This is embodied in the Text of the Convention *"Trade in specimens … must be subject to particularly strict regulation in order not to endanger further their survival"* [4].

It has become common to see examples of species benefiting from legal trade (predominantly of large, commercialised reptiles) [54,55] held up as counterpoints to the risk of unsustainability and thus the need for regulatory reforms. Sustainability must be evidenced; an example from a different species (or indeed Class) merely highlights that for most species there is no available evidence of benefits or declines [8]. The precautionary principle mandates caution in the absence of evidence.

Reforms to CITES are not a new phenomenon (for examples see [13,56,57]), but reforms or amendments take time to disseminate and enforce. The Bonn and Gaborone amendments were proposed in 1979 and 1983, respectively, but currently are only accepted by 149 and 102 of the 183 Parties, respectively. An amendment requiring Parties to submit the methods and results of all NDF's would be complicated and controversial. But the mere fact that the Review of Significant Trade (RST) process has uncovered instances of detrimental trade and missing NDFs highlights that assuming these documents are robust and up to date is insufficient [12,45]. The logistical and political difficulties of implementing change should not censure criticism of regulatory processes nor debate of the status quo. Rather, it should offer space for constructive collaboration across the diversity of wildlife trade scientists to ensure that Parties to CITES deliver on its mission.

Conclusions

The dynamic nature of international wildlife trade and the huge diversity of species involved necessitates a nuanced consideration of trade. While we apply novel analytical methods and indicators to find that CITES trade is dominated by non-threatened species, with unclear responses in trade to changing species threat category, this is no substitute for transparent, accurate, and up-to-date NDF procedures evidencing the population-level effects of trade for all species. Indeed, trade can promote species recovery [17], but this cannot be assumed *a priori* for all species without data-driven justification – conservation outcomes must be evidenced to avoid compounding species extinction risk.

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Author contributions

O.M, B.R.S, T.H and D.P.E. conceived the study idea; O.M collated the data; O.M. analyzed the data and produced the figures with input from B.R.S, T.H and D.P.E; and O.M. wrote the first draft of the manuscript, with all co-authors substantially contributing to revisions.

Declaration of Interests

The authors declare no competing interests.

Main text figure legends

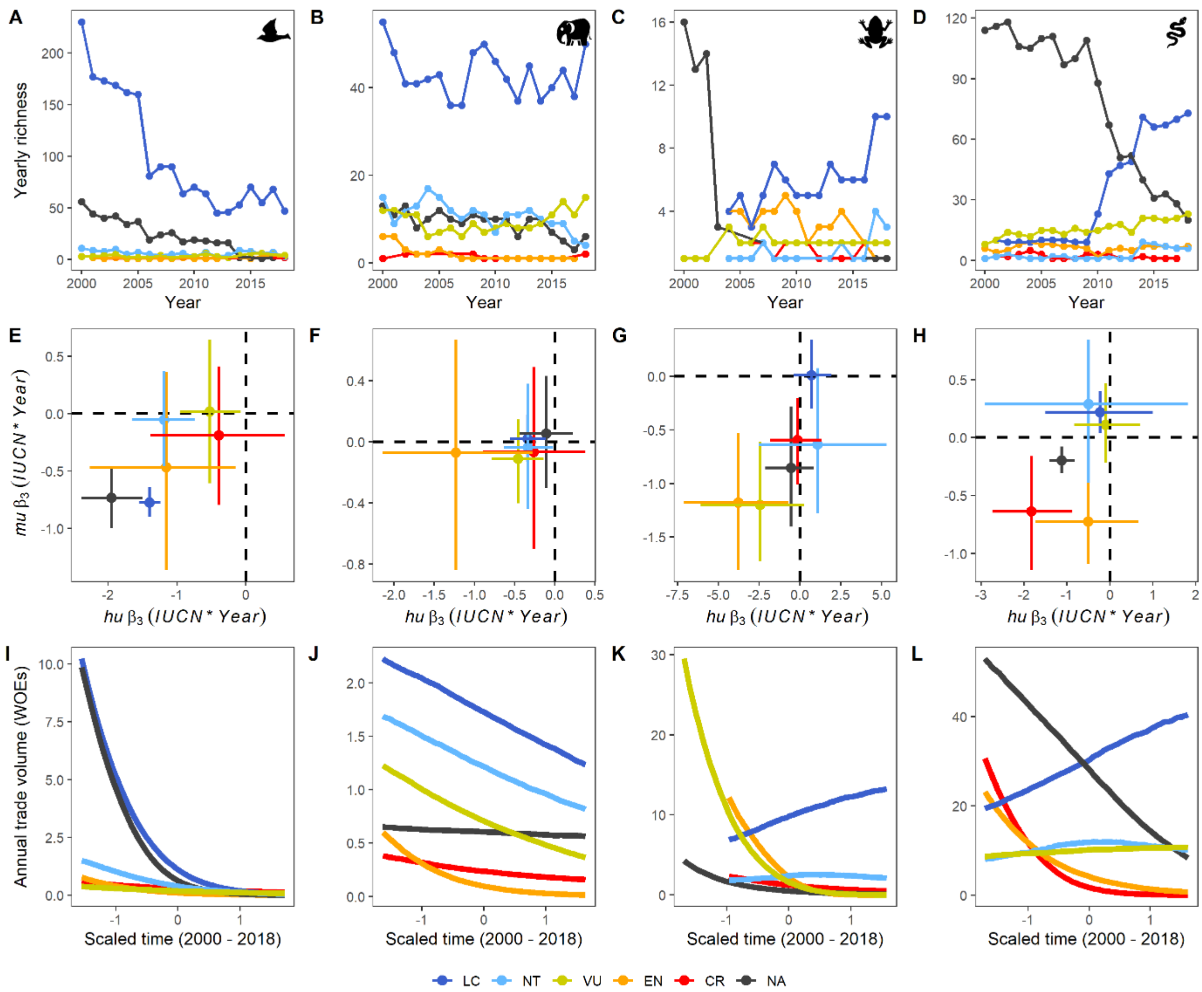

Figure 1. Summary of CITES trends through time. A- D, summary plot of traded species richness per year. E – H, slope coefficients for IUCN Red List categories through time (IUCN*Year) for *hu* (probability of occurrence in trade) and *mu* (volume when traded in WOE's – whole organism equivalent) distributional parameters. Points are medians, solid lines the 90% highest density continuous interval (HDCI), and dashed lines at 0. I – L, joint hurdle-distribution estimates of traded volume through time for the average species in whole organism equivalents (WOE's), lines show the median values. The lagged volume term was fixed at the Red List category mean per class. IUCN categories are respectively coloured dark grey (Not assessed or Data deficient - NA), red (Critically endangered - CR), orange (Endangered-EN), yellow (Vulnerable-VU), pale blue (Near-threatened-NT), and dark blue (Least concern-LC). See also Table S2.

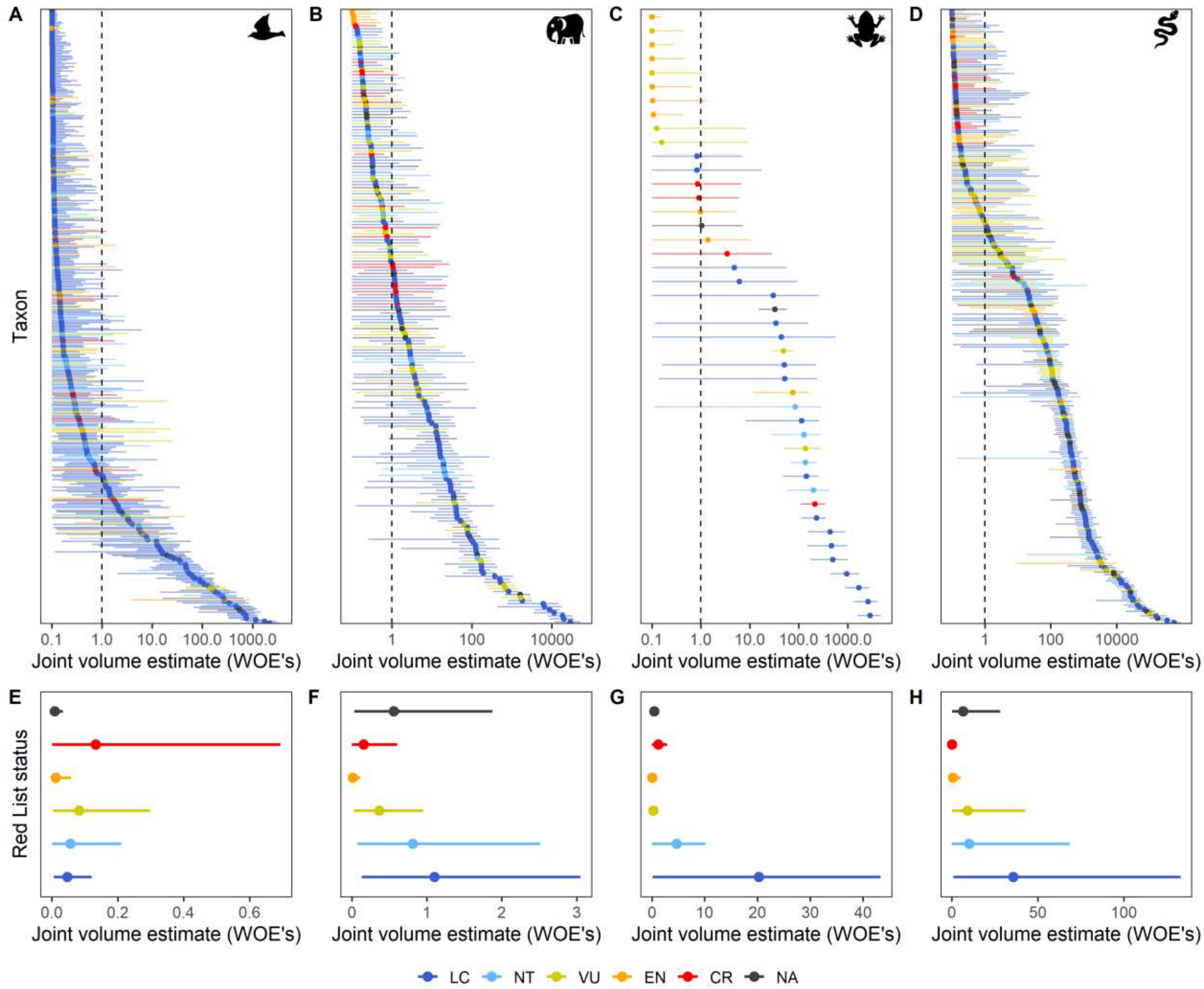

Figure 2. Estimated volumes in trade across both species and Red-list categories. A – D provide species-level joint distributional estimates for all CITES-traded species in 2018 for bird (A, n = 401), mammal (B, n = 202), amphibian (C, n = 44), and reptile (D, n = 283) species, respectively. An adjustment of 0.1 was added to the entire posterior to aid visualisation of species estimated at volumes approaching 0. E-H show joint distributional volume estimates in WOE's for each Red List category in 2018, excluding species-level variability, for birds, mammals, amphibians and reptiles respectively. Red List category is coloured dark grey (Not assessed or Data Deficient - NA), red (Critically endangered - CR), orange (Endangered-EN), yellow (Vulnerable-VU), light blue (Near threatened-NT), and dark blue (Least concern-LC), respectively. Points denote median volumes and lines the 90% HDCI. See also Table S2.

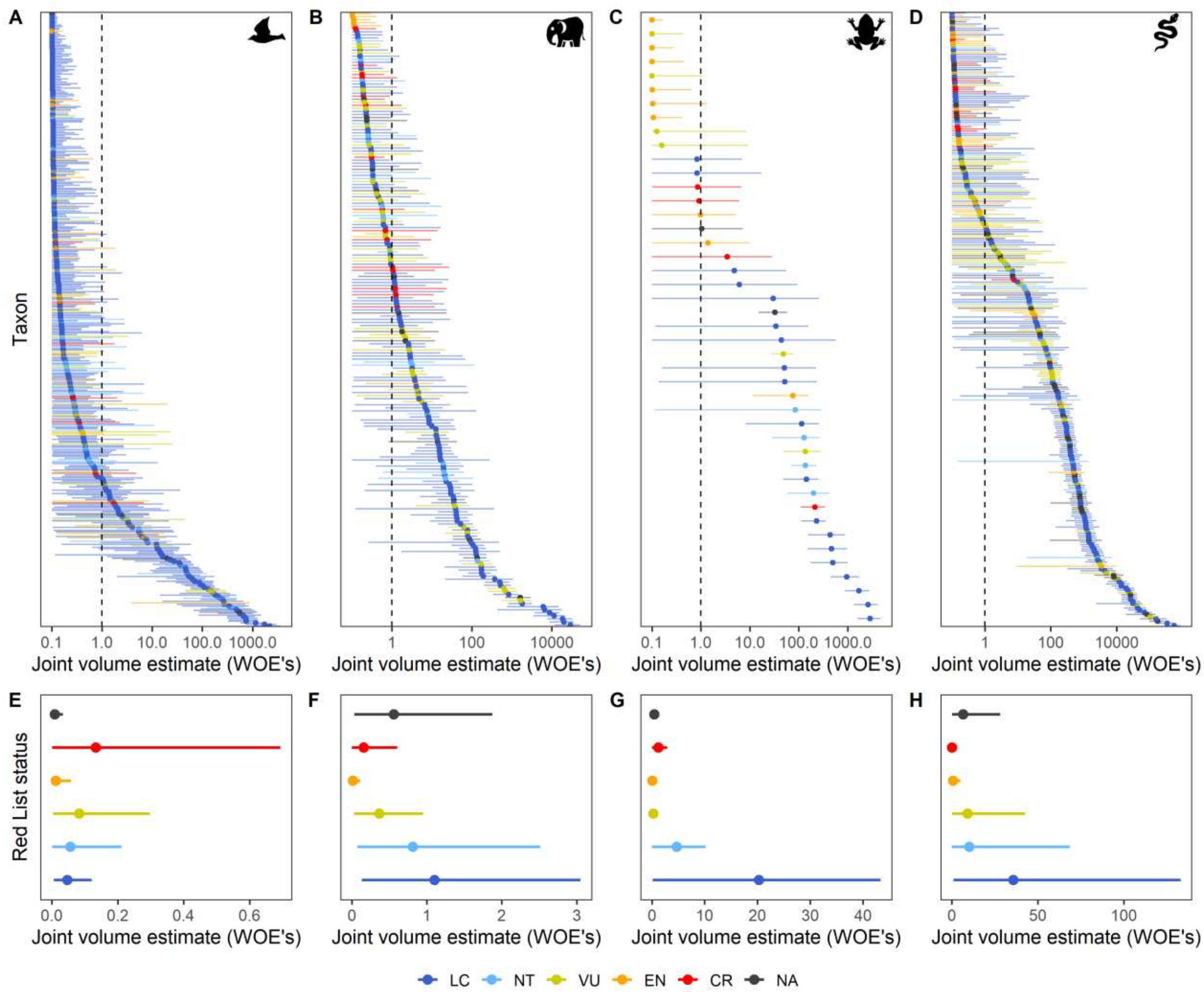

Figure 3. Effect of split listing species populations. A. Joint distributional volume estimates per listing for each split listed species in 2018. Grey lower panel shows population-level estimates for Appendix I (red) and II (blue) groups, excluding species-level variability. Points are medians and the interval is the 90% HDCI. The x-axis on a $\log_{10}$ scale for clarity (an adjustment of 0.1 was added to the entire posterior to aid visualisation of species estimated at volumes approaching 0), the dashed line shows a yearly volume of 1 WOE. B. Estimated probabilities of occurring in trade (*hu*) through time for Appendix I and II listed populations. C. Estimated volumes when traded (*mu*) through time for Appendix I and II listed populations. D. Population-level slope coefficients for populations listed in Appendix I and II through time (Appendix*Year) for both *hu* (probability of being traded) and *mu* (volume when traded) distributional parameters.

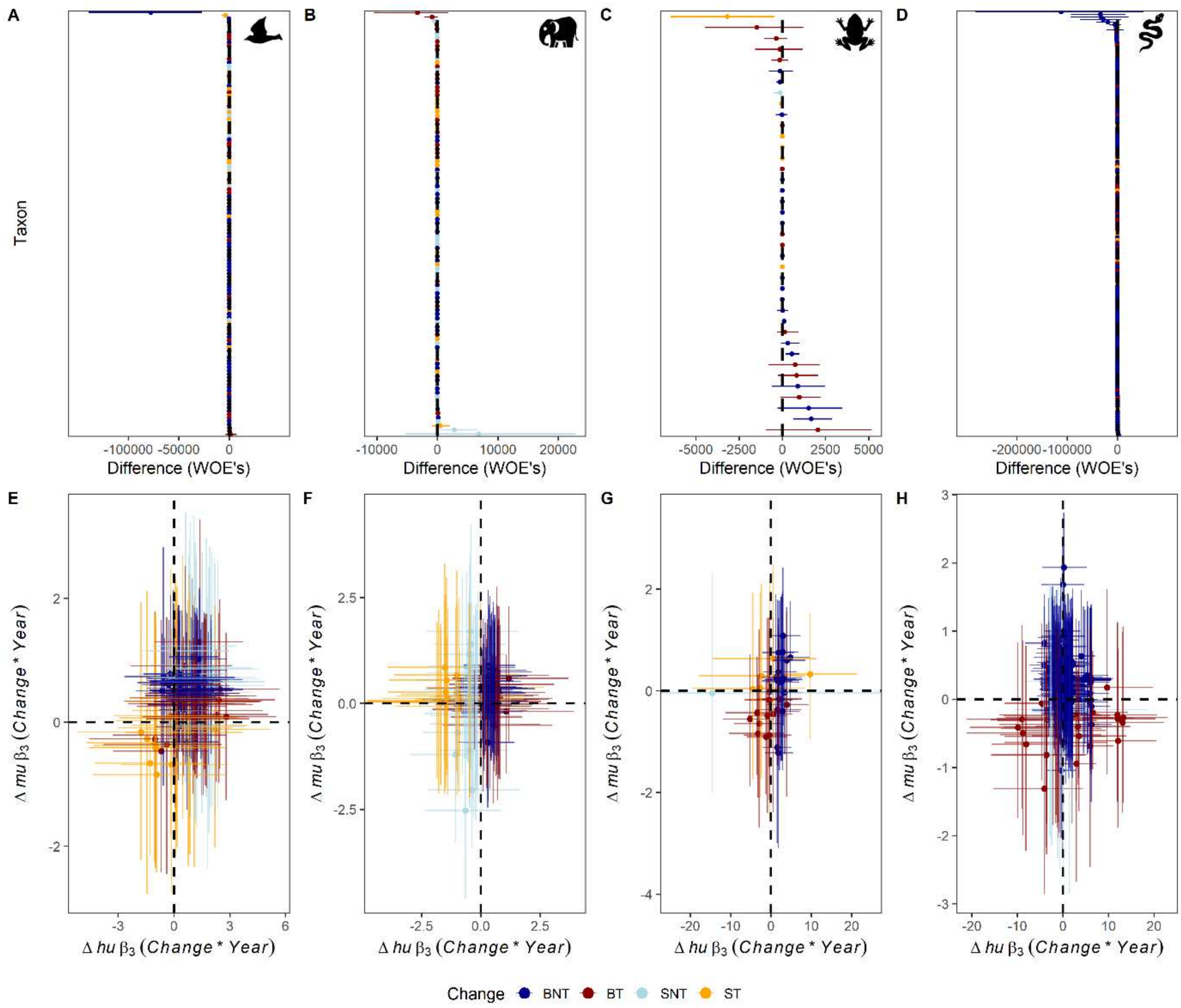

Figure 4. Differences in trends and volumes pre- to post Red List category change. A – D. Species-level estimated volume differences 1-year pre- to 1-year post-change. Each point represents a species-level change for birds (n = 127), mammals (n = 103), amphibians (n = 39), and reptiles (n = 163) respectively. Negative values denote an estimated decrease in traded volume pre- to post-change, vice versa for positive values. E – H. Difference in species-level slope coefficients (Change*Year) pre- to post-change per change for both *hu* and *mu* distributional parameters. Negative values denote the change is associated with a decreasing trend (for *mu* – decreasing volume, for *hu* – decreasing presence) through time relative to the species prior category trend, vice versa for positive values. Points are posterior medians, error bars the 90% HDCI per point and legend acronyms are as follows: BNT - became non-threatened, SNT - stayed non-threatened, BT- became threatened, and ST - stayed threatened. Points along the dashed lines denote no difference in *hu* or *mu* trends, or joint distributional volume estimates. See also Table S4

STAR Methods

Resource availability

Lead contact

Further information and requests for resources should be directed to and will be fulfilled by the Lead Contact, Oscar Morton (omorton1@sheffield.ac.uk).

Materials availability

This study did not generate new unique reagents.

Data and code availability

All data used in this analysis are from publically available sources and no new datasets are generated. Code to reproduce the analysis have been deposited in a freely available institutional repository (10.15131/shef.data.17151449) available upon publication. Any additional information is available from the lead contact upon request.

Experimental model and subject details

CITES data extraction

The CITES Trade database stores all reported wildlife trade (exports and imports) by CITES Parties. These reports are compiled in official annual reports and deposited in the CITES Trade Database. All deposited records were downloaded in bulk (version 2020.1, https://trade.cites.org/), which resulted in a database with 21,635,430 unidirectional trade records. Comprehensive detail of the data structure can be found at the point of access. We follow established protocols for cleaning and preparing the data [1,58]. For a full summary of the data curation pipeline see Table S1. In summary, all re-exports were removed to avoid double counting (keeping only original exports, where the exporter matches the recorded origin), because where trades ultimately pass through multiple countries they may be reported multiple times artificially inflating their presence in the data. Similarly, we focused only on exporter-reported values as it is known that import permits are not required for Appendix II species and as such can lead to underreported figures for these species if trade is not reported [58]. However, there is not one ('correct') standardised approach to analysing CITES trade data, and using only exports could be viewed as an overestimation if some records reflect granted permits, but not realised trade. Therefore, we include a complete re-analysis of all hypotheses using import data (processed identically to the export data). In the supplementary methods we present all main text figures replicated using importer-based values and all supplementary results tables are marked (*, **, etc.) where values differ to those from the exporter-based data. We find no systematic differences between the datasets that affect our overall conclusions.

We also removed all records where species were not traded under any specific Appendix (I, II or III), coded "N". All trades were classed as either wild-sourced or not, using the reported "Source" codes. We follow established criteria and only assign records as wild-sourced where the source code is W, X or R (this respectively includes "Specimens taken from the wild", "Specimens taken in the marine environment not under the jurisdiction of any State", and "Ranched specimens: specimens of animals reared in a controlled environment, taken as eggs or juveniles from the wild, where they would otherwise have had a very low probability of surviving to adulthood") [59]. Records listed as "U" (Unknown) or [Blank] could also refer to wild sourced records. However, they may also refer to records sourced from non-wild sources but lacking documentation. Retrospectively, we cannot know with certainty the reasons enforcement officers around the globe recorded these thus they are also excluded. All subsequent analyses focus solely on these wild-sourced trade records. Records with a source code of C, D, A, F, I, O, U or [Blank], were all excluded at this point. Similarly, as species are traded through CITES for a range of reasons including scientific research and reintroduction, we focus only on trade reported as being for a commercial or personal purpose (purpose codes 'T' and 'P'), which we subsequently term commercial. We include personal following previous studies to potentially capture wild-sourced pet trades [60]. As a result, we exclude the codes B, E, G, H, L, M, N, Q, S, Z or [Blank]. Some commercial movements may potentially be excluded under the medical code (M) or the circus trade (Q), but equally these codes can represent non-commercial trades. Due to this uncertainty, M and Q are excluded. Therefore, all subsequent reference to the data or trade data is in reference to only the wild-sourced and commercially traded records. We limit our time frame to 2000 – 2018 to best understand recent trade. Despite data being present in the CITES data for up to 2021 we conservatively only include records up to and including 2018.

Trade quantities are reported in many "Terms" (teeth, skulls, skin fragments, carvings etc.), which make comparisons of "Quantity" misleading. For example, four skulls represent four individuals, but four small leather pieces or four teeth could represent anything from one to four individuals. Therefore, all records were standardised to whole organism equivalents (WOE's) following the methodologies outlined by [1]. This allows a more robust comparison across trade records as one WOE represents one individual, regardless of taxa or original term. Building on a published WOE conversion protocol [1], we use their published vertebrate conversion factors and add five additional terms which each denote 1 WOE (gall bladder, eggs, eggs (live), specimen and trunk). We applied this conversion protocol to records where the “Unit” term was specified as *NA* denoting “number of specimens”. Records are reported in various other “Unit” terms including bags, bottles, flasks, kilograms, cubic feet, sets, etc. but reconciling this unit diversity remains a research frontier. In total 19.67% of vertebrate records could not be converted to WOE’s. These unconvertable records were removed. We then further focused on bird, mammal, amphibian and reptilian trade data from 2000 – 2018 and removed all records where species were reported as clearly unknown such as *Falco* spp. or *Felis* spp.

Species presence in trade is highly variable with some species being traded consistently each year (2000 – 2018) and others only being traded certain years. This can be attributed to two distinct processes: 1) the species may not have been (reported) in trade that year; or 2) the species was not formally CITES listed prior to (or after) a particular date and as such its trade was not recorded. We cross-referenced the historical CITES listings, which record the year individual species, genera, families, or orders are listed, and matched this information to the processed CITES trade data. Species were marked as absent from trade (a traded volume of 0) if they were not recorded traded but were CITES listed in that year, while species that were added to CITES, deleted from CITES, or added, deleted, and added again to the Appendices have shorter time series. For example, if a species was recorded in trade from 2010, but was listed in 2003, we record that species’ time series as beginning in 2003 (not 2000), its traded volume being 0 for the years 2003 – 2009, and then the reported trade volume from 2010 onwards.

Method details

IUCN data

We obtained IUCN assessments (including all historical assessments) for all wild-sourced commercially trade terrestrial vertebrates (2000 – 2018) using the “*rredlist*” package. We converted pre-2000 codes (lr/cd/nt) and removed all other older notations (such as “*rare*” or “*CT*”, commercially threatened) as more recent assessments before 2000 had been done. The pre-2000 codes were converted were converted thusly “lr” (least concern), “cd” (near-threatened) and “nt” (near-threatened). All species that were returned as not assessed were checked manually for spelling conventions, synonym use or older classification style. Species that had genuinely not been assessed or had been taxonomically split were included as not assessed. As the IUCN assessment data includes the year the assessment was published, species that were in trade preceding a full IUCN assessment were coded as Not evaluated until the year their assessment was published. We also grouped assessments that concluded a species was Data deficient (DD) with the Not evaluated species as a DD finding infers that there was inadequate information to make an assessment and subsequently refer to this group as "Not assessed". All species assessments were read in full as part of this process.

We removed one Extinct species (*Chelonoidis niger*) as a likely misidentification, as assessments of wild and captive populations show all individuals have <80% of the *Chelonoidis niger* genome. We removed one Extinct in the Wild (EW) species (*Oryx dammah* – only 4 records). The records may have been listed as not bred in captivity if the captive breeding did not meet the stringent requirements for CITES classification as bred in captivity. We also removed all instances where species were identified as hybrids such as "*Felis* hybrid" or "*Bison* hybrid" (8 different hybrid types were removed). The reviewed database of species assessments through time were then incorporated into our database of wild-sourced commercial CITES trade, giving a database of species traded volumes (WOEs) and presence in trade through time with up to date IUCN assessment (LC, NT, VU, EN, CR and Not assessed) data for each year. Of the 1053 taxa present in the data, 491 were first traded as LC, 71 as NT, 83 as VU, 36 as EN, 26 as CR and 346 were either not evaluated/recognized or assessed as DD (1025/1053 could be included in the final models, species were lost where they could be resolved for inclusion in the phylogenetic matrices, see Table S1). All references to threat categories made in the main text are solely based on the IUCN Red List, i.e., Endangered refers to the Red List category not species classed under the US Endangered Species Act or other authority. Similarly, we explicitly use the terms threatened to describe species assessed as Vulnerable, Endangered or Critically Endangered by the IUCN Red List, and non-threatened to include species assessed as Least Concern or Near-threatened.

To examine whether trade presence is responsive to perceived changes in species threatened categories, we assessed the difference between species preceding presence and subsequent presence. IUCN changes were modelled with 6 levels, no-change i.e. species that did not change categories at all, pre-change i.e. for species that do change the time period preceding the first change, (changed but) stayed threatened i.e. EN to CR, (changed but) stayed non-threatened i.e. NT to LC, (changed and) became threatened i.e. LC to VU and (changed and) became non-threatened i.e. EN to NT. We considered that when species not assessed or assessed as DD by the IUCN were in trade and then changed or were assessed for the first time this change could either be became threatened (i.e. DD to VU or not assessed to VU) or became non-threatened (i.e. DD to LC or not assessed to NT). We classed a species transition from Not evaluated to DD as 'No-change' as this still infers that there was inadequate information to make a full assessment. However, we removed the three species that transitioned from an assessed state (LC, NT, VU, EN or CR) to DD from this analysis, as this cannot be classed as a change in perceived threat. In total 113 birds (127 unique changes), 87 mammals (103 unique changes), 33 amphibians (39 unique changes) and 162 reptiles (163 unique changes) changed or was assessed for the first time between 2000 and 2018 (totalling 395 species and 432 changes). Of the 127 changes in birds, 62 became non-threatened, 26 became threatened, 18 changed but stayed non-threatened and 21 changed but stayed threatened. Of the 103 changes in mammals, 33 became non-threatened, 24 became threatened, 29 changed but stayed non-threatened and 17 changed but stayed threatened. Of the 39 changes in amphibians, 19 became non-threatened, 13 became threatened, 1 changed but stayed non-threatened and 6 changed but stayed threatened. Of the 163 changes in reptiles, 119 became non-threatened, 33 became threatened, and 11 changed but stayed threatened. In total, 1000 species (including those classed as "No change") could be included in the final models, species were lost where they could be resolved for inclusion in the phylogenetic matrices and where species changed status to DD or only had 1 year of trade data pre- or post-change, see Table S1. In the supplementary information, we present the methods and results of a simplified analysis considering simply where species 'Increase' or 'Decrease' in extinction risk, crucially these results do not contradict our main text analysis.

We hypothesise that becoming threatened would lead to a decrease in trade presence in some cases and more often a reduction in volume relative to the preceding state and vice versa for becoming non-threatened. We hypothesised there to be a weak or null effect of staying threatened or non-threatened relative to a species previous state. This approach allowed us to infer multiple changes in a single species relative to that species preceding state. For example, a species could be pre-change (2000 – 2009), stay non-threatened (2010 – 2014) and become threatened (2015 – 2018). Here we would assess the two changes relative to the preceding (pre-change – to staying non-threatened and then staying non-threatened to becoming threatened). We did not centre each species time-series change year to the year zero, as numerous species changed category multiple times.

CITES split-listing

All species reported in trade in >1 CITES Appendix in the processed wild-sourced commercially traded CITES database were subsetted, as potentially being split listed. Each species in this subset was then manually checked to confirm its split listed status via the historic CITES listings data portal (https://checklist.cites.org/#/en). Species appearing in two Appendices because they had reservations taken out by member parties were excluded, this occurs when a party declares it will not be bound by the Convention for trade concerning a given species. Although parties with active reservations are treated as non-member states with regard to that species, such species could appear in multiple Appendices if they were an Appendix I species and the party with a reservation agreed to report trade as if the species was listed in Appendix II. A small number of species were also listed in multiple Appendices with no explanation or reason found in the historic listings and such species were also excluded. This checking process resulted in the inclusion of time series for explicitly split-listed species traded at least once in both Appendices at least once since 2000 (9 species). We summed WOE's, per species, per Appendix for the timeframe each species was both split-listed and CITES listed.

Limitations of the CITES trade data

All analyses using CITES data could be subject to unknown reporting errors, unfulfilled permits or trades reported in the subsequent year [58]. Here we attempt to standardize the data and our approach to get as wider picture of trade while ensuring accuracy. By converting the data to WOE's, following established methods [1], we standardise a wide variety of the terms used by CITES Parties and the final data values represent number of individual animals. However, a great many terms and units cannot be converted unavoidably meaning we do fail to capture some reported trade.

Similarly, data-handling choices have the potential to unintentionally bias the interpretation of trade data. Rather than attempt to reconcile importer and exporter reported values as a single "true" value, we present the exporter reported analysis results in the main text and in the supplement we provide the importer reported analysis results (Table S2-4, Figure S2 and 3).

Quantification and statistical analysis

Data analyses

All trade data was modelled in a Bayesian framework. This approach was selected due to the high number of individual species, the need to incorporate the known phylogenetic signal of species threatened category with multiple observations [61,62] and to allow derived difference calculations of the posterior. We included species phylogenetic relatedness in our models due to the sheer number of species traded, as these are not truly independent units as they come from the same phylogenetic tree. Thus, the dependency between species should be considered. Conventionally applied phylogenetic least squares (PGLS) analyses of the type implemented in the "*caper*" package [63] do not handle repeated measurements per species (i.e. trade presence for a given species across a number of years) or the additional inclusion of taxon as an independent group effect. Multiple Bayesian packages have since been developed to accommodate this [64,65]. Accordingly, all phylogenetic multilevel models were implemented using the "*brms*" package [65].

The amphibian, avian (Ericson) and mammalian phylogenies of species in our database were generated from 250 sampled trees which were then used to generate a consensus tree (phylogenies available from http://vertlife.org/phylosubsets/ see also [66–68]). A reptile phylogeny of the species traded was sourced from http://timetree.org/ [69]. The phylogenetic correlation matrices (where diagonal elements are equal to 1 [70]) for each class were computed using the *"ape"* package [71]. A small number of reptilian species names could not be resolved and could not be included in subsequent analyses (detailed in Table S1). The taxonomic species names listed in the Appendices were conserved throughout the data pipeline. Where the CITES Appendices records a number of taxa separately, that are resolved to a single species in the phylogenies, we maintained the yearly volume structure for individual CITES taxa and incorporated their variation dependant on phylogeny under the phylogenetically recognized species. For example, CITES lists the Marco polo argali, Tianshan argali, and the Gobi argali subspecies separately (so we track these yearly records separately for each species), thus their variation independent of phylogeny is modelled separately (taxa-level group effect). However, as their exact relatedness is not quantified in the phylogenies available, the variation dependant on phylogeny for the three subspecies was included via the recognized species *Ovis ammon* (Argali).

All data, across hypotheses, were modelled using the hurdle negative binomial (HNB) distribution. This is parametrised by $n \sim HNB(hu, mu, shape)$, where *n* is the outcome, *hu* is the probability of a non-zero value (presence), *mu* is the mean or location parameter of a negative binomial distribution and *shape* (or *phi*) is the over dispersion. The processed CITES trade data contains a high proportion of zeros (years where a species is listed but did not appear in the data). The HNB models the absence of trade and the volume of trade as two distinct processes. A Bernoulli regression (parametrised by *hu*) estimates the probability of being in trade ($n \geq 1$). A truncated negative binomial regression then estimates the volumes when trade occurs (i.e. $n > 0$). The distributional parameters *hu* and *mu* are distinctly estimated as a unique function of predictors (details below), and shape we only constrain to be positive. The joint estimates of the response (HNB) distribution therefore incorporate two key features: 1) whether a species is likely to be traded at all, and 2) if traded what volume this would be in.

Weakly informative priors were specified for each model parameter (see equation details below). All models were visually assessed to ensure chains were mixing and had achieved stable convergence. All Rhat (potential scale reduction factor) values were checked to be <1.05, indicating between and within chain estimates had converged. Post predictive checks were also completed using the predictive distribution, such checks were only used to assess individual model adequacy and check for systemic discrepancies between features of the real and simulated data [72].

Traded presence/volume across Red list categories – probabilities of $n > 0$ (*hu* -$\hat{P}$) and volumes when $n > 0$ (*mu*$-$ μ) are modelled as functions of the standardised lagged-volume traded the previous year, yearly Red List category (IUCN) and Year (2000 – 2018, reduced to 0 – 18 and standardised), and the interaction of IUCN category and Year. Taxon-level variance independent of phylogeny was included as a distinct group effect (indexed by *j* for $\hat{P}$ and *k* for μ). IUCN, Year, and their interaction were incorporated as phylogenetically independent group effects (Equation 1). We incorporated variation dependant on phylogeny via phylogenetic correlation matrices as a separate group effect for both (*hu* and *mu*) distributional parameters (matrices omitted from Eq. 1 for clarity). Weakly informative priors were specified for model slope (β), intercept (α) and standard deviation (σ) (a default *lkj(1)* prior was used for the correlations between grouping factors – not shown here). This model was run for a total of 4000 iterations, including 2000 warm-up iterations, for 4 chains with no thinning per taxonomic class. We note when checking the reptile model, we identified a single species the model overestimated traded volumes, *Podocnemis unifilis*, exponentially increased in traded volumes starting from 8 in 2002 (0 for 2000 and 2001), to 623444 in 2017, with volumes regularly more than doubling between years. Therefore, in 2018, our model does predict this continued growth, but the volumes actually decline to 363363. For clarity, we removed this species from the Figure 2D plot.

Trade volumes were contrasted for the average species in each IUCN threat category by using only the population-level effects and excluding the species-level variability. For contrasts the year was set at 2018 to most closely represent recent trade, and the lagged volume was held at the threat category average per taxonomic class (Table S3). Slope coefficients through time in both occurrence and volume for the average species of each class (β_3) where extracted for the whole posterior and then summarised.

$$
\begin{aligned}
\text{n} &\sim \text{Hurdle-NB}(\hat{P}, \mu, \phi) \\
\text{Logit}(\hat{P}) &= \alpha_j^1 + \beta_j^1(\text{Year}) + \beta_{2j}^1(\text{IUCN}) + \beta_{3j}^1(\text{Year} \times IUCN) + \ \beta_4^1(\text{lag}) \\
\begin{pmatrix} \alpha_j \\ \beta_{1j} \\ \beta_{2j} \\ \beta_{3j} \end{pmatrix} &\sim N\left(\begin{pmatrix} \nu_{\alpha_j} \\ \nu_{\beta_{1j}} \\ \nu_{\beta_{2j}} \\ \nu_{\beta_{3j}} \end{pmatrix}, \begin{pmatrix} \sigma_{\alpha_j}^2 & \rho_{\alpha_j\beta_{1j}} & \rho_{\alpha_j\beta_{2j}} & \rho_{\alpha_j\beta_{3j}} \\ \rho_{\beta_{1j}\alpha_j} & \sigma_{\beta_{1j}}^2 & \rho_{\beta_{1j}\beta_{2j}} & \rho_{\beta_{1j}\beta_{3j}} \\ \rho_{\beta_{2j}\alpha_j} & \rho_{\beta_{2j}\beta_{1j}} & \sigma_{\beta_{2j}}^2 & \rho_{\beta_{2j}\beta_{3j}} \\ \rho_{\beta_{3j}\alpha_j} & \rho_{\beta_{3j}\beta_{1j}} & \rho_{\beta_{3j}\beta_{2j}} & \sigma_{\beta_{3j}}^2 \end{pmatrix}\right), \text{for Taxon j} = 1, \ldots, \text{J} \\
\text{Log}(\mu) &= \alpha_k^2 + \beta_k^2(\text{Year}) + \beta_{2k}^2(\text{IUCN}) + \beta_{3k}^2(\text{Year} \times IUCN) + \ \beta_4^2(\text{lag}) \\
\begin{pmatrix} \alpha_k \\ \beta_{1k} \\ \beta_{2k} \\ \beta_{3k} \end{pmatrix} &\sim N\left(\begin{pmatrix} \nu_{\alpha_k} \\ \nu_{\beta_{1k}} \\ \nu_{\beta_{2k}} \\ \nu_{\beta_{3k}} \end{pmatrix}, \begin{pmatrix} \sigma_{\alpha_k}^2 & \rho_{\alpha_j\beta_{1k}} & \rho_{\alpha_j\beta_{2k}} & \rho_{\alpha_k\beta_{3k}} \\ \rho_{\beta_{1k}\alpha_k} & \sigma_{\beta_{1k}}^2 & \rho_{\beta_{1j}\beta_{2k}} & \rho_{\beta_{1k}\beta_{3k}} \\ \rho_{\beta_{2k}\alpha_k} & \rho_{\beta_{2k}\beta_{1k}} & \sigma_{\beta_{2k}}^2 & \rho_{\beta_{2k}\beta_{3k}} \\ \rho_{\beta_{3k}\alpha_k} & \rho_{\beta_{3k}\beta_{1k}} & \rho_{\beta_{3k}\beta_{2k}} & \sigma_{\beta_{3k}}^2 \end{pmatrix}\right), \text{for Taxon k} = 1, \ldots, \text{k} \\
\text{Log}(\phi) &= \alpha \\
\alpha &\sim \text{Normal}(0,1) \\
\beta &\sim \text{Normal}(0,1) \\
\phi &\sim \text{Gamma}(0.01,0.01) \\
\alpha_{j,k} &\sim \text{Normal}(0, \sigma_{j,k}) \\
\sigma_{j,k} &\sim \text{Normal}(0,1)
\end{aligned}
$$

Equation 1

Trade presence/volume per appendix for split listed species - probabilities of $n > 0$ (hu -$\hat{P}$) and volumes when $n > 0$ ($mu - \mu$) are modelled as functions of the standardised lagged-volume traded the previous year, population Appendix and Year (2000 – 2018, reduced to 0 – 18 and standardised), and the interaction of Appendix and Year. Taxon-level variance independent of phylogeny was included as a distinct group effect (indexed by *j* for $\hat{P}$ and *k* for μ). IUCN, Year, and their interaction were incorporated as phylogenetically independent group effects (Equation 2). As split-listed species in trade are few in number and range across classes, we analysed all classes in one model without incorporating phylogeny. Weakly informative priors were specified for model slope (β), intercept (α) and standard deviation (α) (a default *lkj(1)* prior was used for the correlations between grouping factors – not shown here). These models were run for a total of 2000 iterations, including 1000 warm-up iterations, for 4 chains with no thinning.

$$
\begin{aligned}
\text{n} &\sim \text{Hurdle-NB}(\hat{P}, \mu, \phi) \\
\text{Logit}(\hat{P}) &= \alpha_j^1 + \beta_j^1(\text{Year}) + \beta_{2j}^1(\text{Appendix}) + \beta_{3j}^1(\text{Year} \times \text{Appendix}) + \beta_4^1(\text{lag}) \\
\begin{pmatrix} \alpha_j \\ \beta_{1j} \\ \beta_{2j} \\ \beta_{3j} \end{pmatrix} &\sim N\left(\begin{pmatrix} \nu_{\alpha_j} \\ \nu_{\beta_{1j}} \\ \nu_{\beta_{2j}} \\ \nu_{\beta_{3j}} \end{pmatrix}, \begin{pmatrix} \sigma_{\alpha_j}^2 & \rho_{\alpha_j\beta_{1j}} & \rho_{\alpha_j\beta_{2j}} & \rho_{\alpha_j\beta_{3j}} \\ \rho_{\beta_{1j}\alpha_j} & \sigma_{\beta_{1j}}^2 & \rho_{\beta_{1j}\beta_{2j}} & \rho_{\beta_{1j}\beta_{3j}} \\ \rho_{\beta_{2j}\alpha_j} & \rho_{\beta_{2j}\beta_{1j}} & \sigma_{\beta_{2j}}^2 & \rho_{\beta_{2j}\beta_{3j}} \\ \rho_{\beta_{3j}\alpha_j} & \rho_{\beta_{3j}\beta_{1j}} & \rho_{\beta_{3j}\beta_{2j}} & \sigma_{\beta_{3j}}^2 \end{pmatrix}\right), \text{for Taxon j} = 1, \ldots, \text{J} \\
\text{Log}(\mu) &= \alpha_k^2 + \beta_k^2(\text{Year}) + \beta_{2k}^2(\text{Appendix}) + \beta_{3k}^2(\text{Year} \times \text{Appendix}) + \beta_4^2(\text{lag}) \\
\begin{pmatrix} \alpha_k \\ \beta_{1k} \\ \beta_{2k} \\ \beta_{3k} \end{pmatrix} &\sim N\left(\begin{pmatrix} \nu_{\alpha_k} \\ \nu_{\beta_{1k}} \\ \nu_{\beta_{2k}} \\ \nu_{\beta_{3k}} \end{pmatrix}, \begin{pmatrix} \sigma_{\alpha_k}^2 & \rho_{\alpha_j\beta_{1k}} & \rho_{\alpha_j\beta_{2k}} & \rho_{\alpha_k\beta_{3k}} \\ \rho_{\beta_{1k}\alpha_k} & \sigma_{\beta_{1k}}^2 & \rho_{\beta_{1j}\beta_{2k}} & \rho_{\beta_{1k}\beta_{3k}} \\ \rho_{\beta_{2k}\alpha_k} & \rho_{\beta_{2k}\beta_{1k}} & \sigma_{\beta_{2k}}^2 & \rho_{\beta_{2k}\beta_{3k}} \\ \rho_{\beta_{3k}\alpha_k} & \rho_{\beta_{3k}\beta_{1k}} & \rho_{\beta_{3k}\beta_{2k}} & \sigma_{\beta_{3k}}^2 \end{pmatrix}\right), \text{for Taxon k} = 1, \ldots, \text{k} \\
\text{Log}(\phi) &= \alpha \\
\alpha &\sim \text{Normal}(0,1) \\
\beta &\sim \text{Normal}(0,1) \\
\phi &\sim \text{Gamma}(0.01,0.01) \\
\alpha_{j,k} &\sim \text{Normal}(0, \sigma_{j,k}) \\
\sigma_{j,k} &\sim \text{Normal}(0,1)
\end{aligned}
$$

Equation 2

Trade presence after species change Red list categories – probabilities of *n* > 0 (*hu* -$\hat{P}$) and volumes when *n* > 0 (*mu*– μ) are modelled as functions of the standardised lagged-volume traded the previous year, species change category (Change) and Year (2000 – 2018, reduced to 0 – 18 and standardised), and the interaction of Change and Year. Taxon-level variance independent of phylogeny was included as a distinct group effect (indexed by *j* for $\hat{P}$ and *k* for μ). Change, Year, and their interaction were incorporated as phylogenetically independent group effects (Equation 1). We incorporated variation dependant on phylogeny via phylogenetic correlation matrices as a separate group effect for both (*hu* and *mu*) distributional parameters (matrices omitted from Eq. 1 for clarity). Weakly informative priors were specified for model slope (β), intercept (α) and standard deviation (σ) (a default *lkj(1)* prior was used for the correlations between grouping factors – not shown here). This model was run for a total of 4000 iterations, including 2000 warm-up iterations, for 4 chains with no thinning per taxonomic class.

In the Supplementary Methods we also present a simplified precautionary re-analysis considering only whether species "increased" or "decreased" in threat category. Full details of this approach and the results are detailed there (Figure S4), crucially these are in line with the method we present here.

$$
\begin{aligned}
\text{n} &\sim \text{Hurdle-NB}(\hat{P}, \mu, \phi) \\
\text{Logit}(\hat{P}) &= \alpha_j^1 + \beta_j^1(\text{Year}) + \beta_{2j}^1(\text{Change}) + \beta_{3j}^1(\text{Year} \times \text{Change}) + \beta_4^1(\text{lag}) \\
\begin{pmatrix} \alpha_j \\ \beta_{1j} \\ \beta_{2j} \\ \beta_{3j} \end{pmatrix} &\sim N\left(\begin{pmatrix} \nu_{\alpha_j} \\ \nu_{\beta_{1j}} \\ \nu_{\beta_{2j}} \\ \nu_{\beta_{3j}} \end{pmatrix}, \begin{pmatrix} \sigma_{\alpha_j}^2 & \rho_{\alpha_j\beta_{1j}} & \rho_{\alpha_j\beta_{2j}} & \rho_{\alpha_j\beta_{3j}} \\ \rho_{\beta_{1j}\alpha_j} & \sigma_{\beta_{1j}}^2 & \rho_{\beta_{1j}\beta_{2j}} & \rho_{\beta_{1j}\beta_{3j}} \\ \rho_{\beta_{2j}\alpha_j} & \rho_{\beta_{2j}\beta_{1j}} & \sigma_{\beta_{2j}}^2 & \rho_{\beta_{2j}\beta_{3j}} \\ \rho_{\beta_{3j}\alpha_j} & \rho_{\beta_{3j}\beta_{1j}} & \rho_{\beta_{3j}\beta_{2j}} & \sigma_{\beta_{3j}}^2 \end{pmatrix}\right), \text{for Taxon j} = 1, \ldots, \text{J} \\
\text{Log}(\mu) &= \alpha_k^2 + \beta_k^2(\text{Year}) + \beta_{2k}^2(\text{Change}) + \beta_{3k}^2(\text{Year} \times \text{Change}) + \beta_4^2(\text{lag}) \\
\begin{pmatrix} \alpha_k \\ \beta_{1k} \\ \beta_{2k} \\ \beta_{3k} \end{pmatrix} &\sim N\left(\begin{pmatrix} \nu_{\alpha_k} \\ \nu_{\beta_{1k}} \\ \nu_{\beta_{2k}} \\ \nu_{\beta_{3k}} \end{pmatrix}, \begin{pmatrix} \sigma_{\alpha_k}^2 & \rho_{\alpha_j\beta_{1k}} & \rho_{\alpha_j\beta_{2k}} & \rho_{\alpha_k\beta_{3k}} \\ \rho_{\beta_{1k}\alpha_k} & \sigma_{\beta_{1k}}^2 & \rho_{\beta_{1j}\beta_{2k}} & \rho_{\beta_{1k}\beta_{3k}} \\ \rho_{\beta_{2k}\alpha_k} & \rho_{\beta_{2k}\beta_{1k}} & \sigma_{\beta_{2k}}^2 & \rho_{\beta_{2k}\beta_{3k}} \\ \rho_{\beta_{3k}\alpha_k} & \rho_{\beta_{3k}\beta_{1k}} & \rho_{\beta_{3k}\beta_{2k}} & \sigma_{\beta_{3k}}^2 \end{pmatrix}\right), \text{for Taxon k} = 1, \ldots, \text{k} \\
\text{Log}(\phi) &= \alpha \\
\alpha &\sim \text{Normal}(0,1) \\
\beta &\sim \text{Normal}(0,1) \\
\phi &\sim \text{Gamma}(0.01, 0.01) \\
\alpha_{j,k} &\sim \text{Normal}(0, \sigma_{j,k}) \\
\sigma_{j,k} &\sim \text{Normal}(0,1)
\end{aligned}
$$

Equation 3

We contrasted the absolute difference in expected posterior volumes between 2-years pre and 1-year post-change at the species level (if a species was reassessed in 2010 we contrast 2008 with 2011). We specify these periods pre- and post-change rather than the whole pre and post-change series per species as we are specifically assessing the impact of change.

Therefore, each species category-change has its own comparison timeframe. Comparing between the whole pre- and post-change time series' could lead to erroneous conclusions. For example, take a species that was present in trade for 5-years, then absent for 5-years before increasing in perceived threat category and then remaining absent for the remainder of the series. Comparing the entire pre- and post-change posterior at all year values would reveal overall the species was less present post-change when actually the change was irrelevant as the species was already absent from trade prior to the change.

We further estimated the difference in trend or slope through time between pre- and post-change. This approach aims to detect changes in trend before and after a change e.g. whether a species was increasing in traded volumes through time and then post-change volumes decreased through time. We extracted both species-level distributional coefficients (*hu* and *mu*, $\beta_{3j,k}$) for each change type. The difference was then calculated between the species pre-change slope coefficient through time and the species post-change coefficient through time (Δ Change *Year$_{\text{Post - Pre}}$). All differences were calculated from the full posterior.

We additionally contrasted population-level estimates assessing the impact of change on the average species. This took two forms. Firstly, we contrasted whether for the average species if changes associated with any change in volume pre- to post-change. Each change was contrasted at the class average year of change for each change type. Secondly, whether species that changed category were traded in different volumes to those that remained unchanged in 2018. Thus, assessing whether volumes traded after a change was different to the baseline across species that did not change. This final comparison examines whether species that changed category were systematically present in different volumes to those species that did not change category (Figure S1 and Table S4). Both comparisons here were using the population-level effects only to consider a category change for the average species.

We assess directional differences between Red List categories, before and after a change, and between the CITES Appendices for split-listed species, using the direct probability of direction (*pd*) [73,74]. The *pd* provides evidence of directional effect existence (or the certainty that effect goes in a particular direction, i.e. if endangered species are more likely to reoccur in trade than least concern species in a given year). We term substantial to denote a *pd* >97.5%, a value highly correlated with a two-sided *p*-value of 0.05 [73,74]. The *pd* is calculated from the difference of the full posterior, not a sample or summary. For example, the difference between the population-level posterior volume of the average least concern and vulnerable reptile in a given year. For the presence and split listing analysis we set the year at 2018 – the most recent year in CITES records. We present 90% HDCI's (highest density continuous intervals) to reflect this uncertainty not 95% intervals, as 90% has been deemed more stable [75].

All statistical analyses were carried out using R version 4.0.2 [76]. Data curation and processing were carried out using "*dplyr*" 1.0.2 [77], plotting using "*ggplot2*" 3.3.2 [78], figure arrangement using "*egg*" 0.4.5 [79] and "*png*" 0.1.7 [80]. .All phylogenies were handled using "*ape*" 5.4.1 [71]. Model fitting, checking and post-processing was done using "*brms*" 2.15.0 [65], "*bayestestr*" 0.8.0 [74] and "*tidybayes*" 2.3.1 [81].

Precautionary re-analyses

The method presented in Eq. 3 and Figure 4, using the post-change categories, "Becomes threatened", Becomes non-threatened", "Stayed threatened" and "Stayed non-threatened", picks up important nuance on directional change and whether the change moves the species to a threatened or non-threatened category. The key result of this analysis is that a change does not systematically change species reoccurrence. To confirm this we ran a simpler model solely considering a directional change. Here we modelled the following categories, "Pre-change", "Increase" and "Decrease". Pre-change here denotes the same as in the main methods. Here "Increase" refers to an increase in extinction risk (i.e. LC to VU, NT to CR, etc.). Conversely, "Decrease" is any decrease in extinction risk (i.e. VU to LC, CR to EN etc.). The simplicity of these models required a number of species (changes) to be removed. All species that changed category more than once were removed, all species changing to or from Data Deficient (DD) or changing from Not evaluated (NE) were also removed as changes to or from DD or NE should not be considered an increase or decrease in extinction risk. Therefore, this re-analysis focused only on the most well-understood species that were reassessed into a different category, with full assessments pre and post-change. As before the same criteria as applied in the main text models applied here mainly all species must have at least 2-years data pre- or post-change. The number of species modelled was therefore severely reduced (36 birds, 42 mammals, 3 amphibians and 16 reptiles, totalling 97 species). The basic structure of the models remained the same as that in the main text. Due to the reduced number of species, this model does not account for relatedness between species we do however account for species variation and class level differences by nesting species within taxonomic classes in the models group-level effects (see Equation 5). The smaller number of species meeting the prerequisites for this reanalysis prevent a more nuanced analysis fully separating classes and accounting for species non-independence.

$$
\begin{aligned}
n &\sim \text{Hurdle-NB}(\hat{P}, \mu, \phi) \\
Logit(\hat{P}) &= \alpha^{1}_{j,k} + \beta^{1}_{j,k}(\text{Year}) + \beta^{1}_{2j,k}(\text{Change}) + \beta^{1}_{3j,k}(\text{Year} \times \text{Change}) + \ \beta^{1}_{4}(\text{lag}) \\
\begin{pmatrix} \alpha_j \\ \beta_{1j} \\ \beta_{2j} \\ \beta_{3j} \end{pmatrix} &\sim N\left(\begin{pmatrix} \mu_{\alpha_j} \\ \mu_{\beta_{1j}} \\ \mu_{\beta_{2j}} \\ \mu_{\beta_{3j}} \end{pmatrix}, \begin{pmatrix} \sigma^2_{\alpha_j} & \rho_{\alpha_j\beta_{1j}} & \rho_{\alpha_j\beta_{2j}} & \rho_{\alpha_j\beta_{3j}} \\ \rho_{\beta_{1j}\alpha_j} & \sigma^2_{\beta_{1j}} & \rho_{\beta_{1j}\beta_{2j}} & \rho_{\beta_{1j}\beta_{3j}} \\ \rho_{\beta_{2j}\alpha_j} & \rho_{\beta_{2j}\beta_{1j}} & \sigma^2_{\beta_{2j}} & \rho_{\beta_{2j}\beta_{3j}} \\ \rho_{\beta_{3j}\alpha_j} & \rho_{\beta_{3j}\beta_{1j}} & \rho_{\beta_{3j}\beta_{2j}} & \sigma^2_{\beta_{3j}} \end{pmatrix}\right), for\ Species\!:Class\ j \ = \ 1, \dots, \\
\begin{pmatrix} \alpha_k \\ \beta_{1k} \\ \beta_{2k} \\ \beta_{3k} \end{pmatrix} &\sim N\left(\begin{pmatrix} \mu_{\alpha_k} \\ \mu_{\beta_{1k}} \\ \mu_{\beta_{2k}} \\ \mu_{\beta_{3k}} \end{pmatrix}, \begin{pmatrix} \sigma^2_{\alpha_k} & \rho_{\alpha_k\beta_{1k}} & \rho_{\alpha_k\beta_{2k}} & \rho_{\alpha_k\beta_{3k}} \\ \rho_{\beta_{1k}\alpha_k} & \sigma^2_{\beta_{1k}} & \rho_{\beta_{1k}\beta_{2k}} & \rho_{\beta_{1k}\beta_{3k}} \\ \rho_{\beta_{2k}\alpha_k} & \rho_{\beta_{2k}\beta_{1k}} & \sigma^2_{\beta_{2k}} & \rho_{\beta_{2k}\beta_{3k}} \\ \rho_{\beta_{3k}\alpha_k} & \rho_{\beta_{3k}\beta_{1k}} & \rho_{\beta_{3k}\beta_{2k}} & \sigma^2_{\beta_{3k}} \end{pmatrix}\right), for\ Class\ k \ = \ 1, \dots, K \\
\text{Log}(\mu) &= \alpha^{2}_{l,m} + \beta^{2}_{l,m}(\text{Year}) + \beta^{2}_{2l,m}(\text{Change}) + \beta^{2}_{3l,m}(\text{Year} \times \text{Change}) + \ \beta^{2}_{4}(\text{lag}) \\
\begin{pmatrix} \alpha_l \\ \beta_{1l} \\ \beta_{2l} \\ \beta_{3l} \end{pmatrix} &\sim N\left(\begin{pmatrix} \mu_{\alpha_l} \\ \mu_{\beta_{1l}} \\ \mu_{\beta_{2l}} \\ \mu_{\beta_{3l}} \end{pmatrix}, \begin{pmatrix} \sigma^2_{\alpha_l} & \rho_{\alpha_l\beta_{1l}} & \rho_{\alpha_l\beta_{2l}} & \rho_{\alpha_l\beta_{3l}} \\ \rho_{\beta_{1l}\alpha_l} & \sigma^2_{\beta_{1l}} & \rho_{\beta_{1l}\beta_{2l}} & \rho_{\beta_{1l}\beta_{3l}} \\ \rho_{\beta_{2l}\alpha_l} & \rho_{\beta_{2l}\beta_{1l}} & \sigma^2_{\beta_{2l}} & \rho_{\beta_{2l}\beta_{3l}} \\ \rho_{\beta_{3l}\alpha_l} & \rho_{\beta_{3l}\beta_{1l}} & \rho_{\beta_{3l}\beta_{2l}} & \sigma^2_{\beta_{3l}} \end{pmatrix}\right), for\ Species\!:Class\ l \ = \ 1, \dots, L \\
\begin{pmatrix} \alpha_m \\ \beta_{1m} \\ \beta_{2m} \\ \beta_{3m} \end{pmatrix} &\sim N\left(\begin{pmatrix} \mu_{\alpha_m} \\ \mu_{\beta_{1m}} \\ \mu_{\beta_{2m}} \\ \mu_{\beta_{3m}} \end{pmatrix}, \begin{pmatrix} \sigma^2_{\alpha_m} & \rho_{\alpha_m\beta_{1m}} & \rho_{\alpha_m\beta_{2m}} & \rho_{\alpha_m\beta_{3m}} \\ \rho_{\beta_{1m}\alpha_m} & \sigma^2_{\beta_{1m}} & \rho_{\beta_{1m}\beta_{2m}} & \rho_{\beta_{1m}\beta_{3m}} \\ \rho_{\beta_{2m}\alpha_m} & \rho_{\beta_{2m}\beta_{1m}} & \sigma^2_{\beta_{2m}} & \rho_{\beta_{2m}\beta_{3m}} \\ \rho_{\beta_{3m}\alpha_m} & \rho_{\beta_{3m}\beta_{1m}} & \rho_{\beta_{3m}\beta_{2m}} & \sigma^2_{\beta_{3m}} \end{pmatrix}\right), for\ Class\ m \ = \ 1, \dots, M \\
\text{Log}(\phi) &= \alpha \\
\beta &\sim Normal(0,1) \\
\alpha &\sim Normal(0,1) \\
\phi &\sim \text{Gamma}(0.01,0.01) \\
\alpha_{j,k,l,m} &\sim\sim \text{Normal}(0, \sigma_{j,k,l,m}) \\
\sigma_{j,k,l,m} &\sim \text{Normal}(0,1)
\end{aligned}
$$

Equation 5

This simplified approach has merit but also severe limitations as LC to NT, NT to VU, or EN to CR are all classed equally as "Increases" extinction risk, a factually correct, but very limited interpretation as it is dubious all changes are equally likely to prompt policy or management measures. Crucially, these results mirror our main text findings. There is no systematic change in species traded volumes after pre- to post-IUCN change. This was true across species and for the average species changing to a more-threatened (Increase) or less threatened (Decrease) status, there was no substantial directional effect on traded volumes (Figure S4). There are a number of reasons why species may show no response to a change, namely that species presence is ephemeral and that species may have ceased to be traded (but remain listed) years before the IUCN reassessment and status change.

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