Article:
Meeks, D., Morton, O. and Edwards, D.P. (2024) Wildlife farming: balancing economic and conservation interests in the face of illegal wildlife trade. People and Nature, 6 (2). pp. 446-457. ISSN: 2575-8314

https://doi.org/10.1002/pan3.10588

Received: 26 July 2023 | Accepted: 5 December 2023

DOI: 10.1002/pan3.10588

PEOPLE AND NATURE
BRITISH ECOLOGICAL SOCIETY

REVIEW AND SYNTHESIS

Wildlife farming: Balancing economic and conservation interests in the face of illegal wildlife trade

Dominic Meeks[1] | Oscar Morton[2] | David P. Edwards[2]

[1]Ecology and Evolutionary Biology, School of Biosciences, University of Sheffield, Sheffield, UK

[2]Department of Plant Sciences, University of Cambridge, Cambridge, UK

Correspondence
Dominic Meeks
Email: dom.meeks@gmail.com

Handling Editor: Antoni Margalida

Abstract

1. Demand for wildlife and their products continues to grow, often despite increasingly militarised regulation and consumer awareness campaigns. We review the sustainability, legality and feasibility of wildlife farming of animals, as a potential conservation tool to ensure the development of an equitable and sustainable trade model.
2. While there are some positive examples of well-managed wildlife farming in trade, we identify common themes of misuse including the intentional mislabelling of wild-caught specimens in global trade and the use of wild-caught individuals to supplement captive stocks.
3. We also highlight the frequent failure to incorporate biological data into management strategies, resulting in the widespread use of species with potentially unfavourable life history traits, which constrain the economic and biological sustainability of wildlife farming programmes.
4. We develop a structured decision framework to aid the examination of when wildlife farming may most benefit or hinder species conservation.
5. *Synthesis and applications*. Key opportunities include developing species suitability assessments and removing barriers to legitimate participation with wildlife farming among poor, rural communities. In the absence of management strategies that address the issues of species suitability and accessibility, wildlife farming will continue to place significant strain on wild populations while failing to provide conservation value and sustainable economic returns.

KEYWORDS
biological feasibility, captive breeding, conservation management strategies, economic livelihoods, illegal wildlife trade, supply-side conservation, wildlife farming, wildlife trade

1 | INTRODUCTION

The global wildlife trade supplies a growing demand for animal and plant derivatives, for clothes, decorations, pets, food and traditional medicines (TRAFFIC, 2008). Around 24% of terrestrial species are traded (Scheffers et al., 2019), associating with population declines and extirpations for some populations of terrestrial vertebrates (Morton et al., 2021). Wildlife farming (used interchangeably with captive-breeding)—the rearing of non-domesticated animals for captive-breeding—has existed in various capacities for hundreds of years and recently proliferated, potentially in response to historic trade bans of wild-caught specimens and growing global demand for wildlife commodities (Harfoot et al., 2018). National governments, including China and Vietnam, have recognised the economic potential of wildlife farming and have actively encouraged participation among rural communities (Lyons & Natusch, 2014; White & Yifan, 2022). The greatest growth in wildlife farming has been concentrated among high-volume commercialised species, including Siamese crocodile (*Crocodylus siamensis*), with a captive population of ~1.39 million individuals that vastly exceeds wild populations (Daltry et al., 2016).

Wildlife farming is a potential conservation tool for exploited species (Nogueira & Nogueira-Filho, 2011). It is proposed that by flooding the market with captive-farmed products, market prices will be depressed, rendering poaching unprofitable and alleviating harvesting pressure on wild populations (Wang et al., 2019). However, there is concern that the assumptions underlying the economic theory of wildlife farming's conservation value do not reflect the realities of wildlife trade (Damania & Bulte, 2007; Tensen, 2016). Similarly, while wildlife farming is well-established, including for many CITES-listed species with permitting incentives offered for verified operation, in some instances, it can facilitate laundering and illegal wildlife trade (IWT), or unsustainable exploitation. But this has been poorly evaluated at scale.

Whether wildlife farming of animals has the widespread potential to be a conservation solution for overexploited wild populations, while still supporting the socio-economic welfare of communities, is a key emergent question that has received substantial recent attention in the literature. In this review, we have three objectives: (1) assess the economic sustainability of wildlife farming; (2) evaluate the potential for wildlife farming to facilitate IWT and its implications for sustainability; and (3) develop a decision framework for evaluating the conditions under which wildlife farming can provide both economic and conservation value.

2 | ECONOMICS OF WILDLIFE FARMING

To be sustainable, wildlife farming must both supply individuals to the market without endangering wild populations, and provide consistent and equitable income to local communities relative to exploiting wild populations (Bulte & Damania, 2005). Sustainability is thus underpinned by supply-side considerations related to species-specific biological feasibility, ease of participation and livelihood benefits, and demand-side considerations regarding consumer preferences and fluctuating demand.

2.1 | Supply-side considerations

2.1.1 | Wildlife farming viability and species-specific feasibility

The benefits associated with wildlife farming are inextricably tied to the biological traits of the focal species. Body mass, metabolism, growth rates and diet determine input costs such as feeding and spatial requirements, whilst clutch size and breeding frequency determine output costs and profitability (Naretto et al., 2016). Dietary generalists with limited spatial requirements and high reproductive output have small input costs and high-profit potential. For example, the sale of reticulated python (*Malayopython reticulatus*) skins, meat and gall bladder generated annual profits between US$23,080 and $240,040 at scales ranging from small to industrial farms in China and Vietnam (Lyons & Natusch, 2014). Wildlife farming of taxa with favourable biological traits has few barriers to participation and is economically viable at multiple scales.

For valuable, threatened species with large spatial and dietary requirements and low breeding frequencies, such as the tiger (*Panthera tigris*), profits are only obtained by farms with sophisticated, large-scale breeding infrastructure (Kirkpatrick & Emerton, 2010). Profits from wildlife farming involving such taxa are concentrated in a small number of industrial-scale farms (EIA, 2013). Species with slow reproductive rates or limited reproductive success in captive facilities, such as short-beaked echidna (*Tachyglossus aculeatus*), may not produce offspring at the rate required to generate profits, match consumer demand and displace wild-sourced products, limiting the conservation value of wildlife farming (Summerell et al., 2019). Despite the constraints of slow life history traits, there can be economic successes with, for instance, the sale of captive-bred crocodile products in Southeast Asia estimated to generate US$21.4 million per year (Daltry et al., 2016). Substantial Asian captive crocodile populations are sustained by large reproductive outputs owing to high rates of juvenile survival in captivity and moderate fecundity. However, in some instances, the supplementation of captive stocks with wild-caught specimens sourced from neighbouring countries' has driven precipitous population declines, contravening CITES Resolution Conf. 10.16 (Rev.), which prohibits regular supplementation of captive stock with wild specimens in a manner detrimental to the species' wild population (CITES, 2000; Daltry et al., 2016). Wildlife farming programmes promoted without consideration of species-specific biological feasibility, which heavily influence conservation value and economic opportunities, are likely to incur pervasive sustainability outcomes (Challender et al., 2019).

The importance of species' biological traits in determining the sustainability and economic viability of wildlife farming is not reflected in the literature. A rapid literature review returned 103 studies that met our search criteria (see Appendix S1 for methods) and assessed the economic sustainability, legality, and feasibility of wildlife farming. These studies included species-specific trade assessments, reviews of governance and international trade, and documentation of consumer attitudes towards captive-bred products. Biological feasibility was assessed in just six of the 103 studies (Figure 1a). This dearth of research potentially reflects our poor understanding of the commercial breeding feasibility of many species and our inability to predict economic and conservation outcomes for many species (Lyons & Natusch, 2011). Cross-taxa research of biological feasibility should be prioritised to ensure that wildlife farming management is accurately informed.

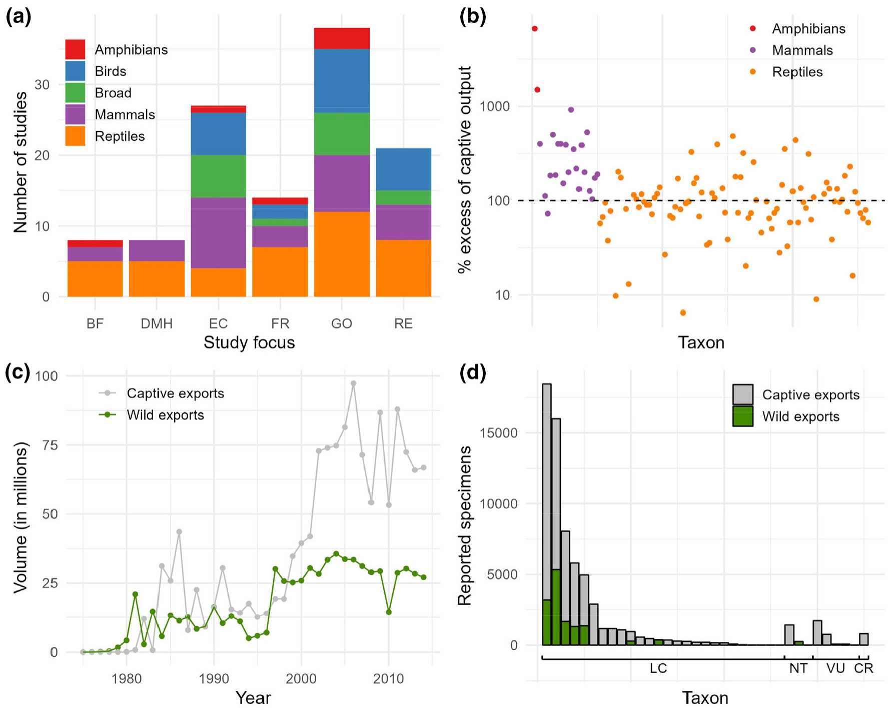

FIGURE 1 The magnitude and risks of wildlife farming. (a) Number of published studies covering distinct research areas identified by our literature search ($n=103$, with some studies spanning multiple research areas and taxonomic groups) (see Appendix S1 for methods). BF, biological feasibility; DMH, disease, morality, and hybridisation; EC, economic considerations; FR, fraud; GO, governance; RE, review. (b) Quotas for wildlife farming set by the Indonesian government as a percentage of the breeding facilities' reproductive output for specified taxon. Data from Janssen and Chng (2018). (c) Total mammal, bird, reptile, amphibian, fish, invertebrate, and plant specimens internationally traded under CITES from captive and wild sources between 1975 and 2014. Data from Harfoot et al. (2018). (d) Amount of trade in specimens from the Solomon Islands from captive and wild sources. Data from Shepherd et al. (2012).

2.1.2 | Livelihood benefits and economic barriers

Wildlife farming offers rural communities an alternative avenue for engagement with wildlife trade and provides supplementary food security where local conditions limit traditional agricultural production (Lyons & Natusch, 2014). Python breeding provided an alternative source of income and food security to communities in the Mekong Delta when typical livelihoods were disrupted by extreme weather events and disease (Aust, 2022). In a 10-year study, Nogueira and Nogueira-Filho (2011) calculated that semi-intensive wildlife farming of collared peccary (*Pecari tajacu*), fed by common fruits and agricultural by-products, was 19,000 times more productive (kg of meat/hectare/year) than the sustainable harvest of peccary from the Peruvian Amazon forest. The financial and subsistence benefits afforded by wildlife farming can prevent communities from converting biodiverse habitats to agriculture or pasture, representing an additional mechanism by which wildlife farming can provide conservation value (Nogueira & Nogueira-Filho, 2004).

Significant establishment costs represent a key economic barrier to legitimate and widespread participation with wildlife farming (WCS, 2017). Specimens are regularly sourced from wild populations to reduce establishment costs and increase early profits for programmes breeding species with low reproductive rates, such as Asian black bear (*Ursus thibetanus*) (Crudge et al., 2020). Farmers of Siamese crocodile preferentially restock captive populations with wild caught specimens due to lower costs and a belief that wild individuals are more fecund, decimating wild populations in Laos (which lacks captive-breeding farms) to supply captive populations throughout Asia (Daltry et al., 2016). Complex global supply chains saturate the economic benefits of trade for small-scale breeders, encouraging the supplementation of captive stock with wild-caught specimens (Robinson et al., 2015a). Connecting suppliers directly to foreign markets would increase the profitability of wildlife farming, while encouraging legitimate participation among local stakeholders.

2.2 | Demand-side considerations

2.2.1 | Consumer attitudes

Attitudes towards captive-bred products vary between consumer groups and product types, although a general shift to captive-sourced individuals has been documented (e.g. CITES-listed international trade, Figure 1c). For consumptive goods, such as food and traditional medicine (TM), it is commonly believed that wild-sourced products are favoured, because of perceived prestige or potency (Liu et al., 2016; Shairp et al., 2016). However, recent research contests this binary portrayal of consumer attitudes. Hinsley and Sas-Rolfes (2020) argue that methodological limitations of consumer studies oversimplify consumer behaviour and fail to incorporate the diverse array of factors that influence real-world consumer decisions. Consumers readily switch between wild-caught, captive-bred and synthetic bear bile products depending on the price, production method, perceived effectiveness, safety and legality of the product (Hinsley et al., 2022), with consumer elasticity also being observed regarding the consumption of wild-caught and farmed turtle meat (Nuno et al., 2018). Additionally, Chinese consumers demonstrated a greater willingness to accept substitute medicines for bear bile and tiger bones when made aware of the species' conservation status, building on earlier findings that revealed consumers who felt a duty towards conservation and were more likely to select farm-sourced products (Coals et al., 2020; Liu et al., 2016). A general preference for captive-bred pets and clothing products is attributed to the absence of hunting, which could damage the product, and more extensive hygiene regulations, which can reduce the incidence of zoonotic diseases and parasites (Harfoot et al., 2018). These studies underscore consumer elasticity and reveal a market opportunity for wildlife farming as a 'conservation-positive' alternative to wild-sourced products.

2.2.2 | Demand uncertainties

There is concern that increasing the supply of captive-bred products will lower prices and thus stimulate novel wildlife commodities, potentially undermining the conservation value of wildlife farming (Tensen, 2016). For instance, an increased supply of tiger bone from captive-bred sources reduced market prices for traditional tiger bone products, stimulating the emergence of novel products such as tiger bone wine and glue (Dang Vu et al., 2022; EIA, 2013). Additionally, increased supply and diversity of products increased acceptability of bear bile and tiger consumption for medicinal products (Rizzolo, 2021). Whilst consumer engagement with captive-bred commodities is encouraging, increased aggregate demand will drive intensified exploitation of wild populations if captive-bred products fail to displace wild-sourced products or cannot meet demand. Conversely, the legacy impacts of COVID-19 may push consumers towards certified captive sources following temporary restrictions on the consumption of wildlife products and elevated societal awareness of hygiene at wildlife markets (Veríssimo et al., 2020).

3 | CRIME WITHIN WILDLIFE FARMING

Any conservation value prescribed to wildlife farming is contingent on the assumption that exploitation of wild populations has been minimised. Illicit practices that violate this assumption include intentionally labelling wild-caught individuals as captive-bred ('laundering') enabling poachers to bypass regulation and exploit vulnerable wild populations under the guise of legitimate captive trade (Nijman & Shepherd, 2009). We assess the scope of wildlife laundering at local, national, and international scales, and evaluate the responsibilities and inadequacies of regulating authorities and verification strategies tasked with maintaining the sustainability and legality of wildlife farming.

3.1 | Local to national-scale laundering

National-level regulation of wildlife farming facilities is assigned to government departments such as the Forestry Administration of Cambodia (Thomson, 2008). Due to financial constraints within departments, inspections of breeding facilities can be infrequent, providing ample opportunity for laundering (van Uhm, 2018). During an investigation of wildlife farming facilities in Indonesia, indicators of laundering were regularly identified including: unfeasible differences between the number of F1 and F2 individuals; non-existent or limited breeding stock; and incomplete or replicated monthly records (Nijman & Shepherd, 2009). Efforts to govern facilities are undermined by insufficient governance structures, corruption, and limited prosecutorial force (Nijman et al., 2018). In Vietnam, 89% of porcupine farmers reported paying the government's Forest Protection Department a fee for farm registration, which is a free process, strongly indicating corruption whilst many farmers freely admit to having used wild specimens to replenish their breeding stock (Brooks et al., 2010). In Indonesia, 92% of python farmers admitted to circumventing laws and regulations by paying off officials (Lyons & Natusch, 2011). Considering low prosecution rates for wildlife crimes, current governance provides minimal deterrence to those profiting from illegitimate wildlife farming.

Our literature search identified 14 papers documenting the laundering of 157 species, including the illegal collection of 4227 green pythons (*Morelia viridis*) from wild populations in Indonesia to supply breeding farms in Jakarta (Lyons & Natusch, 2011). Additionally, a combined 3606 captive-bred specimens from five reptile species, including endangered spiny turtle (*Heosemys spinosa*), were deemed to be illegally sourced by breeding facilities in Indonesia (Nijman & Shepherd, 2009). Nijman and Shepard (2015b) concluded that captive-breeding permits for ~3 million tokay gecko (*Gekko gecko*), issued by the Indonesian Directorate General of Forest Protection and Nature Conservation, were used to launder an equivalent number of wild-caught gecko for illegal export as dried specimens. Due to the clandestine nature of illegal operations, the scope and scale of laundering is very likely greater than currently documented (van Uhm, 2018). The prevalence of laundering compromises the credibility of claims that wildlife farming is sustainable and in the absence of additional governance or enforcement, will continue to threaten the population health of commercially valuable species.

3.2 | International-scale laundering

While CITES has strict regulations requiring proof that specimens are captive bred (at least second-generation (F2) products of captive-breeding, and regulating F1 generations as '*born in captivity*' rather than '*bred*')—and breeding stock is maintained without detriment to wild populations, the lack of resources for enforcement means these restrictions can be subverted. Administrative errors within the reporting of wildlife trade are common and contribute to mismatches between the reported and actual number of imports and exports of captive-bred products (Blundell & Mascia, 2005). Discrepancies have been increasingly documented among CITES-listed taxa including clouded leopard (*Neofelis nebulosa*) and radiated tortoise (*Astrochelys radiata*) (D'Cruze & Macdonald, 2015; Nijman & Shepherd, 2007). CITES scientific authorities have often failed to determine whether cases of erratic documentation involving threatened species have been genuine administrative errors or laundering (Nijman & Shepherd, 2009). For example, the vast majority of 54,793 birds exported from the Solomon Islands between 1995 and 2011 are suspected to have been laundered due to the sheer volume of reportedly captive-bred individuals (Figure 1d). This includes Papuan hornbill (*Rhyticeros plicatus*) which, despite scarce breeding facilities on the islands and few successful records of breeding in captivity, were commonly exported as captive-bred (Shepherd et al., 2012).

CITES Parties have permitted trade involving substantial, undocumented captive stocks in countries outside of the species' native range that were likely established illegally, and trade routes that pass through non-CITES Parties, which are subsequently not required to disclose trade data (Nijman & Shepherd, 2010, 2011, 2015a, 2015b; Shepherd et al., 2012). For instance, between 2004 and 2008, 16 species of South American poison arrow frogs (*Dendrobatidae* sp.) totalling 2665 specimens labelled as captive-bred were exported from Kazakhstan (a CITES Party) via Lebanon (not a CITES Party), yet the origin of this substantial captive stock is undocumented (Nijman & Shepherd, 2010). Paucity of data detailing the location and quantity of existing captive populations and the upscaling of genetic verification strategies, such as mandatory genetic family testing, represent key challenges in resolving illegal captive stock acquisition (Hogg et al., 2018). Similarly, exports of captive-bred CITES-listed species that blatantly exceed the capacity of nations' breeding facilities and species' reproductive traits undermine both the conservation potential and trust in wildlife farming programmes (Nijman & Shepherd, 2015a, 2015b).

Quotas are sparingly used in wildlife farming despite their prevalence in the regulation of sustainable wild-sourced offtake. CITES Parties that establish wildlife farming quotas are responsible for monitoring compliance among breeders (Williams & Sas-Rolfes, 2019). Initial quota systems for wildlife farming, such as the Indonesian Captive Breeding Production Programme (CBPP), are designed to promote transparency, deter laundering, and prevent unrestricted restocking from wild populations. However, major doubts remain over the effectiveness of Indonesia's CBPP (Janssen & Chng, 2018). For instance, by failing to incorporate accurate biological data, CBPP quotas exceeded the breeding capacity of captive stocks by an average of 333.29% (range = 101%–6667%) for 21 mammal species, 38 reptile species, and two amphibian species (Figure 1b). Governments provide an avenue for laundering by producing quotas that cannot be feasibly supplied by the existing captive stock, incentivising farmers to fulfil the remaining quota with wild-sourced specimens (Nijman & Shepherd, 2015a, 2015b). It is unclear whether these failings are the product of poor record keeping, scientific malpractice, or governments directly benefiting from excessive quotas. To reform previous failings and avoid the endangerment of threatened wild populations, initiatives tracking captive stocks and output like the CBPP are essential and should be more widespread, although future quotas must incorporate accurate, species-specific biological data.

To ensure the sustainability and legality of wildlife farming, regulatory bodies must overcome the logistical challenge of simultaneously tackling local to national-scale and international-scale laundering. In 2017, CITES Parties adopted Resolution Conf. 17.7, which aimed to reduce the incidence of laundering in wildlife farming by creating a review process for species-country combinations and highlighted specific criteria where captive-sourced trade is suspect (CITES, 2022). These included: rapid increases in captive-sourced trade volume; frequent shifts between different captive production source codes; inconsistencies in the reported source from importers and exporters; and applying the captive production source code to Appendix S1 species when no facilities have been registered with the Secretariat. CITES may issue recommendations to countries to improve the compliance of wildlife-farming facilities with CITES' goals, and if these recommendations are not met, the Secretariat may suspend trade in captive-bred products for the specific species-country combination (CITES, 2017). Whilst the tools are available to identify laundering across taxa, legislative constraints may encumber the restriction of illegal activity within wildlife farming.

3.3 | Verification approaches

Methods used to verify the provenance of captive-bred products vary in cost, species applicability, and timing. Verification strategies employed at breeding facilities include visual assessments of parasite load and damage on specimens (Lyons et al., 2017). Skin measurements can verify provenance as captive-bred specimens are typically harvested before full maturity whilst wild-sourced products are fully grown adults (Webb et al., 2012). Egg counting is employed for reptiles, with each captive offspring required to have a corresponding eggshell (Lyons & Natusch, 2015). As snake species have unique eggshell volume ranges, Lyons and Natusch (2011) suggest recording eggshell volumes to counteract the use of false eggshells. This strategy could be utilised among birds and other reptile groups. Similarly, reptile skin can be physically or chemically branded to identify captive-bred individuals (CITES, 2014). Visits to breeding facilities could also provide an opportunity for regulators to assess the welfare of captive specimens (Whitfort, 2021). Visual verification methods are taxa-specific and unsuitable for use in later stages of the supply chain, creating the need for new verification strategies.

Stable isotope analysis (SIA), microsatellite markers and mass spectrometry have all been used to verify the captive-bred provenance of processed foods and traditional medicine products (Coals et al., 2021; Dittrich et al., 2017). Both SIA and mass spectrometry verify origin by the diet profile of the specimen, which can be aided by breeders feeding captive stocks specific diets or markers. SIA can infer provenance and geographic origin, the requirement of intact samples restricts its application, whilst microsatellites and mass spectrometry both work with processed and damaged sources (Coals et al., 2021; Coetzer et al., 2017; Dittrich et al., 2017). As microsatellites can verify both the parentage and population origin of an individual, they can be used to certify captive-bred specimens which genetically match legitimate captive stock whilst also detecting potential cases of IWT where individuals do not match their stock (Coetzer et al., 2017). However, these methods require expensive reference databases that may not be compatible with the limited budgets of regulatory forces (Hogg et al., 2018).

Current deficiencies within national governance of wildlife trade suggest that the levels of international coordination required to verify the provenance of captive-bred products may be unattainable (Sas-Rolfes et al., 2019). Wildlife farming is distinct from other sources of wildlife crime, having received substantial support from national governments that simultaneously lack the structures and resources required to ensure its sustainability and legality (EIA, 2013). The absence of effective national governance increases the responsibilities of comparatively poorly funded CITES Committees and NGOs in safeguarding wild populations from illegal captive-breeding practices.

4 | DECISION FRAMEWORK FOR WILDLIFE FARMING OUTCOMES

Future strategies to improve the sustainability of captive-breeding must acknowledge the limitations of wildlife farming and restrict its use for biologically unfeasible species. To aid this, we develop a decision framework that can be applied by national governments and NGOs to produce a wildlife farming model with lower risk of illegal activity and unsustainable practice. Reproductive traits, trade pressure, uses and conservation statuses vary markedly between the species involved with captive-breeding. The combination of these factors often determines the scope for captive-breeding to provide conservation and economic value. Our decision framework thus identifies situations in which conservation and economic opportunities and risks align (Figure 2).

4.1 | High-stakes circumstances

Trade of threatened species can be lucrative and has contributed to precipitous population declines among numerous charismatic and lesser-known species (Ayling, 2013; Nellemann et al., 2016; Pietersen et al., 2014). Wildlife farming involving threatened, commonly traded species has the potential to displace wild-caught supply and alleviate pressure on wild populations offering substantial conservation benefit. This is documented by the breeding of American alligator (*Alligator mississippiensis*) in Louisiana, where poaching of locally threatened populations was deterred by substantial increases in the supply of farmed products (Moyle, 2013). Despite these benefits, any additional demand or laundering stimulated by captive-breeding can compound pressure on vulnerable wild populations (e.g. black-winged myna, *Acridotheres melanopterus*), making captive-breeding in such circumstances a high-stakes strategy (Nijman et al., 2018). The failure to prioritise conservation outcomes ahead of economic opportunity has contributed to the limited conservation benefit of Chinese tiger farms (Kirkpatrick & Emerton, 2010). Wildlife farming should be considered in high-stakes circumstances where alternative conservation strategies have failed to halt population decline and where governance capacity is sufficient to prevent laundering. These scenarios require robust data on source stock, biological parameters, projected annual output, and international demand.

4.2 | Low-stakes circumstances

The relative stakes of conducting wildlife farming for fecund, non-threatened species are low. In these circumstances, programmes should prioritise the provision of equitable livelihoods to rural communities. By providing a financial incentive to maintain wild populations, wildlife farming in low-stakes circumstance can encourage the sustainable management of natural resources and provide conservation value in high biodiversity areas (Nogueira & Nogueira-Filho, 2011). This is exemplified by community-based wildlife farming and ranching programmes for yellow-spotted Amazon river turtles (*Podocnemis unifilis*) and red-footed tortoise (*Chelonoidis carbonarius*) (Sinovas et al., 2017). By harvesting and developing captive populations from eggs at sites with poor hatching potential (e.g. close proximity to water), local communities minimised their impact on wild populations whilst generating revenue (Caputo et al., 2005). Wildlife farming of less feasible species is unlikely to provide substantial income but can provide a supplementary food source to rural communities (Aust, 2022). The inherent risk in low-stakes scenarios is born by communities rather than species, as the economic benefits provided by wildlife farming can fluctuate with consumer preferences. For example, demand for captive-bred green iguana (*Iguana iguana*) collapsed from 100,000 in 2005 to zero in 2013 and 2014, due to a shift in consumer demand towards easier-to-keep reptile species such as bearded dragons (*Pogona vitticeps*) (Sinovas et al., 2017). To build economic resilience, farmers and communities reliant on wildlife farming should utilise multiple species or individual species with multiple uses to withstand market trends and maintain a stable income.

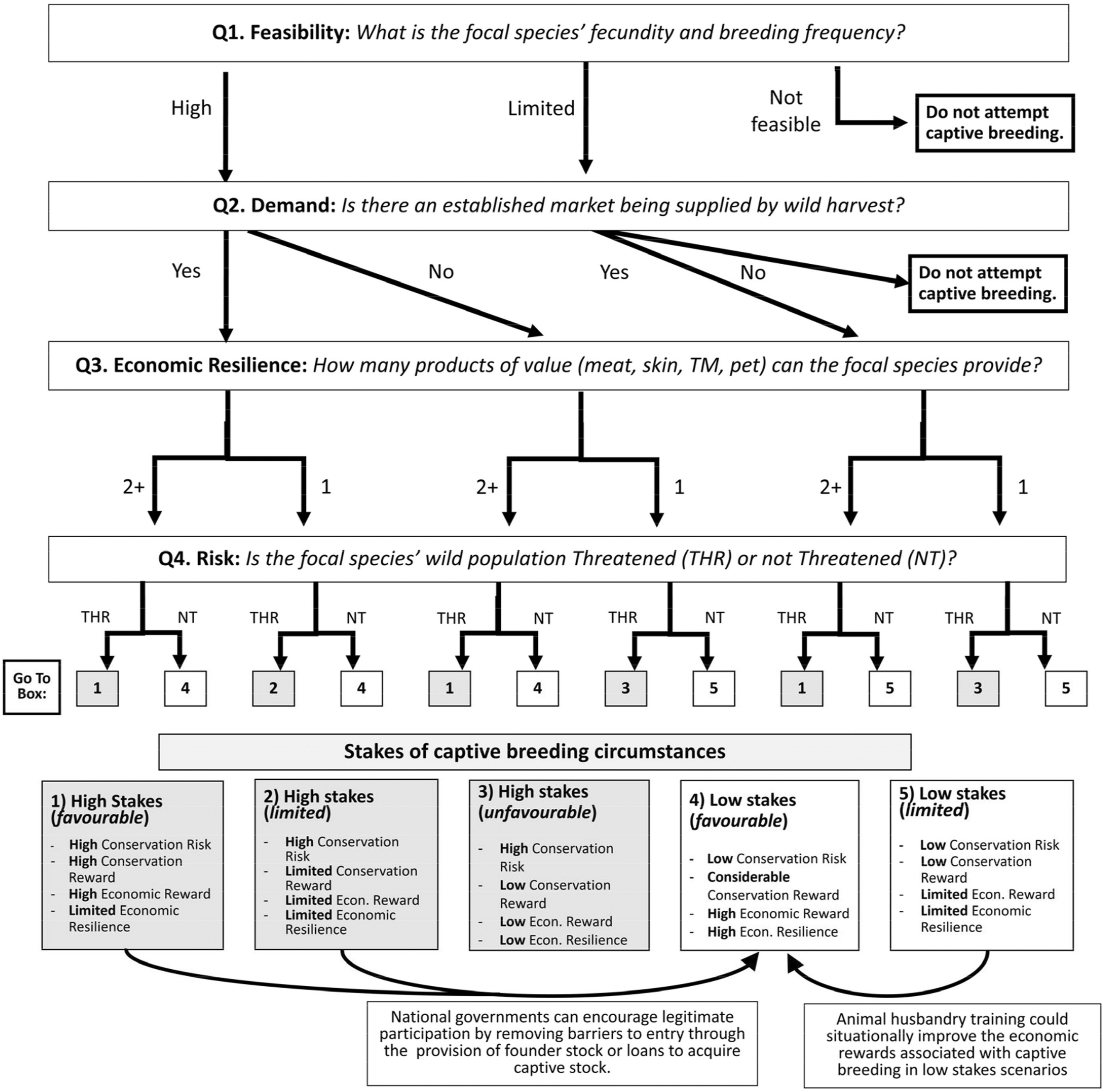

FIGURE 2 A decision framework to aid in the assessment of species suitability for wildlife farming and the identification of suitable objectives in various contextual circumstances. The framework sequentially considers feasibility, demand, economic resilience, and risk to identify whether wildlife farming is high (grey) or low (white) stakes, and whether it is a favourable, limited, or unfavourable strategy in the specific circumstances.

4.3 | Unsustainable circumstances

Wildlife farming is inadvisable for species that are unable to reproduce effectively in captivity (e.g. pangolins) (Challender et al., 2019). Without producing a substantial number of offspring, captive products will be unable to displace wild-caught supply or provide economic livelihoods (Tensen, 2016). As these species cannot generate profits, any traded specimens labelled as captive-bred are likely to have been laundered (Nijman & Shepherd, 2015a). In circumstances involving traded species, where captive breeding is not viable due to a species' slow or complex life history, wildlife farming should not be established regardless of supplementary food provision, with demand instead supplied by well-regulated sustainable offtake (Milner-Gulland et al., 2020). Captive-breeding is also inadvisable for species with limited feasibility and no existing trade market, as stimulating novel demand will incentivise the exploitation of wild populations where profits can be obtained more swiftly than through legitimate wildlife farming. These circumstances are identified in Question 1 and 2 of Figure 2.

4.4 | Removing barriers to participation

To enable legitimate participation with wildlife farming (Figure 2), financial and knowledge barriers must be addressed. To avoid captive stocks being unsustainably founded and supplemented by wild specimens, national governments could establish captive-breeding centres to supply founder stock to new programmes, helping ensure wildlife farming programmes do not compromise wild populations (Nogueira & Nogueira-Filho, 2011). Additionally, removal scenario modelling could reduce the consequences of captive stock establishment in high-stakes circumstances by identifying the specimen type (juvenile, adult, male, female) whose removal will incur the least demographic disturbance for the species' wild population (Colomer et al., 2020). In Vietnam, government loans have enabled prospective breeders to purchase initial captive stock, although the financial restraints faced by many governments may require alternative financing mechanisms to be explored (Lyons & Natusch, 2014). Our decision framework in Figure 2 can be utilised by national governments to efficiently allocate support, in the form of loans and tax breaks, to captive breeding programmes involving species with the greatest potential to deliver positive conservation and economic outcomes. NGOs such as the Sepik Wetlands Management Initiative have provided critical funding to nascent captive-breeding programmes and further NGO involvement would support the delivery of sustainable, rural livelihoods (Daltry et al., 2016). Whilst subsidising sourcing of captive stock will increase legitimate participation and reward established wildlife farms, such payments may finance a growing re-sale market and reduce the volume of captive-bred products displacing wild-sourced commodities, diminishing the conservation value of wildlife farming (Brooks et al., 2010).

Alternatively, in ranching-style systems, communities would collect a regulated number of individuals from wild populations to establish captive stocks or collect a scientifically-informed quota of juveniles/eggs to sell as adults (Lyons & Natusch, 2011). By linking the health of wild populations to profit opportunity, communities would be incentivised to maintain and restore the habitat of the focal species to secure future income (Daltry et al., 2016). The small number of collected juveniles will enable the growth of captive populations and prevent the need for continued offtake that may inhibit the recovery of wild populations (Vera, 2009). Efforts to increase participation must avoid simply withdrawing legislative protection for threatened species, as seen in the most recent revision of China's Wildlife Protection Law, which removed captive-breeding permit requirements for locally protected species and species categorised as having ecological, scientific and social value (White & Yifan, 2022). Instead, participation with either government loans or regulated collection could be contingent on collaboration with regulatory authorities. Whilst enabling broad, legitimate participation with wildlife farming, this measure would also allow captive stocks records to be updated, filling a major regulatory knowledge gap (Thomson, 2008).

Breeder knowledge gaps can lead to sub-optimal wildlife farming practices that undermine income and safety. To address this issue, the Vietnamese Farming Association hosted free workshops for captive breeders to develop python husbandry skills (Lyons & Natusch, 2014). Workshops in this style could be utilised across a range of taxa to provide breeders with the skills required to reduce excess mortality, improve hygiene practices, and increase reproductive output (Robinson et al., 2015b). These sessions could also train community members on how to meet certification standards and connect suppliers directly to markets (Bijman & Wijers, 2019). The removal of financial and knowledge barriers to promote legitimate captive-breeding participation serves as a foundation for resolving issues of illegality and unsustainability within the supply of captive-bred products.

5 | FUTURE DIRECTIONS

5.1 | Linking feasibility and economics into CITES

CITES cannot unilaterally prohibit trade in a species due to its inability to breed in captivity (high-stakes circumstances; Figure 2), alternative strategies for identifying and restricting wildlife farming of unsuitable species must be investigated. If CITES Parties could be encouraged to keep records of breeding facilities and stocks for all Listed species (not just a subset of Appendix S1 species), management authorities could use this in tandem with life history trait databases to create robust quotas and interrogate existing quotas for validity (Janssen & Chng, 2018). Paramount to this would be broadening the criteria used to make listing and quota decisions to safeguard biologically unsuitable species, species more likely to suffer from laundering, and species subject to rapid changes in demand (Cooney et al., 2021). This last point being crucial, as species feasibility for farming is not a static thing, it will change both with advances in ecological understanding and changes in species perceived value. If the potential value of species increases, the farming of species that was previously unfeasible may suddenly become feasible and vice versa.

The creation of a database for biological feasibility assessments, as well as their active incorporation into the CITES review process, would allow for more immediate identification of fraudulent wildlife farming programmes, reducing damage inflicted on wild populations. Governments of exporter countries should incorporate rigorous decision frameworks (Figure 2) into diligent evaluations of species suitability and governance capability before supporting new captive-breeding programmes, and consider scaling down existing operations for unsuitable or threatened species (EIA, 2013).

5.2 | Shifting demand to sustainable captive sources

An array of strategies have been utilised to reduce demand for wild-sourced animal products across different consumer groups throughout society. Education and awareness campaigns target conservation-sympathetic consumers and attempt to deter future consumers, whilst outreach programmes and social influences have targeted consumer groups with culturally entrenched behaviour (Wallen & Daut, 2018). Despite these strategies garnering anecdotal success in reducing demand for specific species, their collective impact is poorly documented and potentially limited by the strategies' specificity (Veríssimo & Wan, 2019). Alternative broad-scale, supply-side mechanisms, such as captive-bred and synthetic wildlife products, should be prioritised as intermediary conservation strategies. For instance, captive-bred exclusivity policies utilised in songbird competitions (Jepson et al., 2011) could be extended to hobbyist collector groups to incentivise the shift away from wild-sourced specimens across a broader range of taxa. Public pledges from community leaders and physicians, advocating for reduced consumption of wildlife products, have influenced traditional consumer groups; however, in an increasingly digital and status-dominated world, certification schemes should explore the role of celebrities in engaging younger consumer groups (Olmedo et al., 2020).

Management strategies must also consider the timing of consumer campaigns, as redirecting demand towards captive-bred products before issues of laundering are resolved will provide consumers with a false sense of sustainability whilst the erosion of wild populations continues (Veríssimo et al., 2020). To prevent the competition of conservation-oriented supply-side mechanisms, the utilisation of synthetic wildlife products, such as synthetic bear bile and tiger bone, should be prioritised in circumstances where wildlife farming is unfeasible (Li et al., 2016). The complementary deployment of supply-side interventions will maximise the availability of alternative wildlife products to consumers and help alleviate trade pressure across a broader range of exploited taxa.

5.3 | Alternative financing mechanisms

Wildlife farming requires investment to alleviate initial establishment costs and provide sustainable livelihoods (Bulte & Damania, 2005). National governments can build on existing financial provisions for captive-breeding by creating channels for NGOs and importer countries to direct funds to community-based programmes to reduce costs associated with acquiring breeding stock, infrastructure and knowledge gaps (Daltry et al., 2016). The limited financial capacity of national governments and NGOs may constrain the extent of development among prospective wildlife farming programmes.

Verified captive-bred products could utilise documented consumer demand elasticity to provide an alternative to ecologically damaging wild-sourced commodities (van der Ven & Cashore, 2018). Certification schemes must engage with breeders of all scales to ensure that the livelihood benefits resulting from access to premium markets extend to small-scale rural breeders. Schemes should base market access on farm efficiency and sustainability to avoid rewarding less efficient programmes and displacing efficient, sustainable farms (Lim et al., 2017). National governments of exporting and importing countries can create opportunities for private investment by offering tax exemptions for certified captive-bred traded products that have been verified via genetic testing or rigorous visual assessment (Larnder-Besner et al., 2020). Such legislation would encourage the adoption of captive-bred products throughout the supply chain and necessitate the rapid development of breeding facilities, creating opportunities for private investors to provide loans to prospective breeders to purchase captive stock and breeding infrastructure (Brancalion et al., 2017). These measures would also attract private investment in the regulation and certification of breeding facilities to reduce competition from wild-caught sources, helping to address the deficiencies of national government enforcement (Edwards & Laurance, 2012). Government safeguards must be implemented to ensure prospective small-scale breeders are afforded equal investment opportunities as industrial-scale operations with more profit potential and to prevent the criminalisation of rural communities financially constrained from legitimate participation in wildlife farming (Duffy et al., 2019). Carefully designed government interventions can leverage private investment to build capacity in breeding facilities, capitalise on consumer plasticity towards captive-bred products, and facilitate the continued displacement of wild-sourced wildlife products.

6 | CONCLUSION

Wildlife farming can deliver equitable and sustainable livelihoods to rural communities and reduce harvesting pressure on threatened wild populations. Currently, these benefits are being widely undermined via the laundering of biologically unfeasible species and the absence of accessible pathways to legitimate participation, which further expose threatened species to exploitation. Moreover, without resolving issues of legality and sustainability, the redirection of consumer demand to captive-bred wildlife products will ultimately fail to benefit wild populations. Our study highlights that regulation of wildlife farming requires reform to reflect the considerable diversity of economic and conservation outcomes of captive breeding and the multitude of factors that contribute to these outcomes. We recommend that regulatory authorities, such as CITES, increase scrutiny into the provenance of international captive-bred trade involving highly threatened or biologically unfeasible species, to fulfil their duty of ensuring that international trade of wild animals does not threaten the survival of the species. We also suggest that by helping address financial and knowledge barriers to captive breeding, national governments and NGOs can make progress toward targets of rural poverty alleviation via the provision of sustainable local livelihoods. The current prevalence of wildlife farming suggests it will remain a feature of the wildlife trade. Whether wildlife farming continues to facilitate IWT or provides economic and conservation value will be determined by the success of intersectional management strategies in reducing the scope of captive-breeding, limiting laundering, and addressing the drivers of illegal participation.

AUTHOR CONTRIBUTIONS

Dominic Meeks conceived the ideas, collected the data and led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interests.

DATA AVAILABILITY STATEMENT

The data set documenting the geographic location, taxonomic focus, research focus and DOI of the 103 studies that met the inclusion criteria of our rapid literature search of wildlife farming can be found in the following data deposit: https://doi.org/10.6084/m9.figshare.24899403.

ORCID

Dominic Meeks https://orcid.org/0009-0002-0141-470X
Oscar Morton https://orcid.org/0000-0001-5483-4498
David P. Edwards https://orcid.org/0000-0001-8562-3853

REFERENCES

Aust, P. W. (2022). *Reptile farming as a response to livelihood insecurity in the Mekong Delta*. https://panorama.solutions/en/solution/reptile-farming-response-livelihood-insecurity-mekong-delta

Ayling, J. (2013). What sustains wildlife crime? Rhino horn trading and the resilience of criminal networks. *Journal of International Wildlife Law & Policy*, *16*(1), 57–80. https://doi.org/10.1080/13880292.2013.764776

Bijman, J., & Wijers, G. (2019). Exploring the inclusiveness of producer cooperatives. *Current Opinion in Environmental Sustainability*, *41*(December), 74–79. https://doi.org/10.1016/j.cosust.2019.11.005

Blundell, A. G., & Mascia, M. B. (2005). Discrepancies in reported levels of international wildlife trade. *Conservation Biology*, *19*(6), 2020–2025. https://doi.org/10.1111/j.1523-1739.2005.00253.x

Brancalion, P. H. S., Lamb, D., Ceccon, E., Boucher, D., Herbohn, J., Strassburg, B., & Edwards, D. P. (2017). Using markets to leverage investment in forest and landscape restoration in the tropics. *Forest Policy and Economics*, *85*(December), 103–113. https://doi.org/10.1016/j.forpol.2017.08.009

Brooks, E. G. E., Roberton, S. I., & Bell, D. J. (2010). The conservation impact of commercial wildlife farming of porcupines in Vietnam. *Biological Conservation*, *143*(11), 2808–2814. https://doi.org/10.1016/j.biocon.2010.07.030

Bulte, E. H., & Damania, R. (2005). An economic assessment of wildlife farming and conservation. *Conservation Biology*, *19*(4), 1222–1233. https://doi.org/10.1111/j.1523-1739.2005.00170.x-i1

Caputo, F. P., Canestrelli, D., & Boitani, L. (2005). Conserving the terecay (*Podocnemis unifilis*, Testudines: *Pelomedusidae*) through a community-based sustainable harvest of its eggs. *Biological Conservation*, *126*(1), 84–92. https://doi.org/10.1016/j.biocon.2005.05.004

Challender, D. W. S., Sas-Rolfes, M., Ades, G. W. J., Chin, J. S. C., Sun, N. C., Chong, J., & Connelly, E. (2019). Evaluating the feasibility of pangolin farming and its potential conservation impact. *Global Ecology and Conservation*, *20*(October), e00714. https://doi.org/10.1016/j.gecco.2019.e00714

CITES. (2000). *Conf. 10.16 (revision)—Specimens of animal species bred in captivity*. CITES. https://cites.org/sites/default/files/document/E-Res-10-16-R11_0.pdf

CITES. (2014). *Notification to parties: Export of skins of caiman crocodilus*. https://cites.org/sites/default/files/notif/E-Notif-2014-033.pdf

CITES. (2017). Review of the provisions of resolution conference 17.7 (Rev. CoP18). https://cites.org/eng/node/55962

CITES. (2022). *CITES representatives and experts discuss trade and captive breeding of CITES-listed animals at UNEP-WCMC in Cambridge, UK*. CITES. https://cites.org/eng/news/CITES_representatives_experts_discuss_trade_captive_breeding_of_CITES-listed_animals_at_UNEP-WCMC_Cambridge_06042017

Coals, P., Loveridge, A., Kurian, D., Williams, V. L., Macdonald, D. W., & Ogden, R. (2021). DART mass spectrometry as a potential tool for the differentiation of captive-bred and wild lion bones. *Biodiversity and Conservation*, *30*(6), 1825–1854. https://doi.org/10.1007/s10531-021-02170-2

Coals, P., Moorhouse, T. P., D'Cruze, N. C., Macdonald, D. W., & Loveridge, A. J. (2020). Preferences for lion and tiger bone wines amongst the urban public in China and Vietnam. *Journal for Nature Conservation*, *57*(October), 125874. https://doi.org/10.1016/j.jnc.2020.125874

Coetzer, W. G., Downs, C. T., Perrin, M. R., & Willows-Munro, S. (2017). Testing of microsatellite multiplexes for individual identification of cape parrots (*Poicephalus robustus*): Paternity testing and monitoring trade. *PeerJ*, *5*(March), e2900. https://doi.org/10.7717/peerj.2900

Colomer, M. À., Oliva-Vidal, P., Jiménez, J., Martínez, J. M., & Margalida, A. (2020). Prioritizing among removal scenarios for the reintroduction of endangered species: Insights from bearded vulture simulation modeling. *Animal Conservation*, *23*(4), 396–406. https://doi.org/10.1111/acv.12549

Cooney, R., Challender, D. W. S., Broad, S., Roe, D., & Natusch, D. J. D. (2021). Think before you act: Improving the conservation outcomes of CITES listing decisions. *Frontiers in Ecology and Evolution*, *9*. https://doi.org/10.3389/fevo.2021.631556

Crudge, B., Nguyen, T., & Cao, T. T. (2020). The challenges and conservation implications of bear bile farming in Viet Nam. *Oryx*, *54*(2), 252–259. https://doi.org/10.1017/S0030605317001752

Daltry, J., Langelet, E., Solmu, G., van der Ploeg, J., van Weerd, M., & Whitaker, R. (2016). Successes and failures of crocodile harvesting strategies in the Asia Pacific region. *Tropical Conservation: Perspectives on Local and Global Priorities*, *21*, 345–362.

Damania, R., & Bulte, E. H. (2007). The economics of wildlife farming and endangered species conservation. *Ecological Economics*, *62*(3), 461–472. https://doi.org/10.1016/j.ecolecon.2006.07.007

Dang Vu, H. N., Gadbert, K., Nielsen, J. V., Nielsen, M. R., & Jacobsen, J. B. (2022). The impact of a legal trade in farmed tigers on consumer preferences for Tiger bone glue—Evidence from a choice experiment in Vietnam. *Journal for Nature Conservation*, *65*(February), 126088. https://doi.org/10.1016/j.jnc.2021.126088

D'Cruze, N., & Macdonald, D. W. (2015). Clouded in mystery: The global trade in clouded leopards. *Biodiversity and Conservation*, *24*(14), 3505–3526. https://doi.org/10.1007/s10531-015-1010-9

Dittrich, C., Struck, U., & Rödel, M. (2017). Stable isotope analyses—A method to distinguish intensively farmed from wild frogs. *Ecology and Evolution*, *7*(8), 2525–2534. https://doi.org/10.1002/ece3.2878

Duffy, R., Massé, F., Smidt, E., Marijnen, E., Büscher, B., Verweijen, J., Ramutsindela, M., Simlai, T., Joanny, L., & Lunstrum, E. (2019). Why we must question the militarisation of conservation. *Biological Conservation*, *232*(April), 66–73. https://doi.org/10.1016/j.biocon.2019.01.013

Edwards, D. P., & Laurance, S. G. (2012). Green labelling, sustainability and the expansion of tropical agriculture: Critical issues for certification schemes. *Biological Conservation*, *151*(1), 60–64. https://doi.org/10.1016/j.biocon.2012.01.017

EIA. (2013). *Hidden in plain site—China's clandestine tiger trade*. Environmental Investigation Agency. https://eia-international.org/wp-content/uploads/EIA-Hidden-in-Plain-Sight-med-res1.pdf

Harfoot, M., Glaser, S. A. M., Tittensor, D. P., Britten, G. L., McLardy, C., Malsch, K., & Burgess, N. D. (2018). Unveiling the patterns and trends in 40 years of global trade in CITES-listed wildlife. *Biological Conservation*, *223*(July), 47–57. https://doi.org/10.1016/j.biocon.2018.04.017

Hinsley, A., & Sas-Rolfes, M. (2020). Wild assumptions? Questioning simplistic narratives about consumer preferences for wildlife products. *People and Nature*, *2*(4), 972–979. https://doi.org/10.1002/pan3.10099

Hinsley, A., Wan, A. K. Y., Garshelis, D., Hoffmann, M., Hu, S., Lee, T. M., & Meginnis, K. (2022). Understanding why consumers in China switch between wild, farmed, and synthetic bear bile products. *Conservation Biology*, *36*(3), e13895. https://doi.org/10.1111/cobi.13895

Hogg, C. J., Dennison, S., Frankham, G. J., Hinds, M., & Johnson, R. N. (2018). Stopping the spin cycle: Genetics and bio-banking as a tool for addressing the laundering of illegally caught wildlife as 'captive-bred'. *Conservation Genetics Resources*, *10*(2), 237–246. https://doi.org/10.1007/s12686-017-0784-3

Janssen, J., & Chng, S. C. L. (2018). Biological parameters used in setting captive-breeding quotas for Indonesia's breeding facilities. *Conservation Biology*, *32*(1), 18–25. https://doi.org/10.1111/cobi.12978

Jepson, P., Ladle, R. J., & Sujatnika. (2011). Assessing market-based conservation governance approaches: A socio-economic profile of Indonesian markets for wild birds. *Oryx*, *45*(4), 482–491. https://doi.org/10.1017/S003060531100038X

Kirkpatrick, C. R., & Emerton, L. (2010). Killing tigers to save them: Fallacies of the farming argument. *Conservation Biology*, *24*(3), 655–659. https://doi.org/10.1111/j.1523-1739.2010.01468.x

Larnder-Besner, M., Tremblay-Gravel, J., & Christians, A. (2020). Funding pandemic prevention: Proposal for a meat and wild animal tax. *Sustainability*, *12*(21), 9016. https://doi.org/10.3390/su12219016

Li, S., Tan, H. Y., Wang, N., Hong, M., Li, L., Cheung, F., & Feng, Y. (2016). Substitutes for bear bile for the treatment of liver diseases: Research progress and future perspective. *Evidence-based Complementary and Alternative Medicine*, *2016*(March), e4305074. https://doi.org/10.1155/2016/4305074

Lim, F. K. S., Carrasco, L. R., McHardy, J., & Edwards, D. P. (2017). Perverse market outcomes from biodiversity conservation interventions. *Conservation Letters*, *10*(5), 506–516. https://doi.org/10.1111/conl.12332

Liu, Z., Jiang, Z., Fang, H., Li, C., Mi, A., Chen, J., & Zhang, X. (2016). Perception, price and preference: Consumption and protection of wild animals used in traditional medicine. *PLoS ONE*, *11*(3), e0145901. https://doi.org/10.1371/journal.pone.0145901

Lyons, J. A., & Natusch, D. J. D. (2011). Wildlife laundering through breeding farms: Illegal harvest, population declines and a means of regulating the trade of green pythons (*Morelia viridis*) from Indonesia. *Biological Conservation*, *144*(12), 3073–3081. https://doi.org/10.1016/j.biocon.2011.10.002

Lyons, J. A., & Natusch, D. J. D. (2014). *Assessment of python breeding farms supplying the international high-end leather industry*. IUCN. https://portals.iucn.org/library/node/43337

Lyons, J. A., & Natusch, D. J. D. (2015). *Methodologies for differentiating between wild and captive-bred CITES-listed snakes*. https://cites.unia.es/cites/file.php/1/files/id_material/Methodologies_differentiating_between_wild_and_captive-bred_snakes.pdf

Lyons, J. A., Natusch, D. J. D., & Jenkins, R. W. G. (2017). *Guidance for inspection of captive breeding and ranching facilities*. CITES. https://cites.org/sites/default/files/eng/prog/captive_breeding/E-InspectionGuidance-FINAL.pdf

Milner-Gulland, E. J., Hughes, P., Bykova, E., Buuveibaatar, B., Chimeddorj, B., Karimova, T. Y., Lushchekina, A. A., Salemgareyev, A., von Meibom, S., & Zuther, S. (2020). *The sustainable use of Saiga antelopes: Perspectives and prospects*. Saiga Conservation Alliance. https://www.cms.int/saiga/sites/default/files/publication/unep-cms_saiga_mos4_outcome2_sustainable-use-saiga-antelopes_e_.pdf

Morton, O., Scheffers, B. R., Haugaasen, T., & Edwards, D. P. (2021). Impacts of wildlife trade on terrestrial biodiversity. *Nature Ecology & Evolution*, *5*(4), 540–548. https://doi.org/10.1038/s41559-021-01399-y

Moyle, B. (2013). Conservation that's more than skin-deep: Alligator farming. *Biodiversity and Conservation*, *22*(8), 1663–1677. https://doi.org/10.1007/s10531-013-0501-9

Naretto, S., Cardozo, G., Blengini, C. S., & Chiaraviglio, M. (2016). Importance of reproductive biology of a harvest lizard, *Tupinambis merianae*, for the management of commercial harvesting. *Wildlife Research*, *42*(8), 697–704. https://doi.org/10.1071/WR15056

Nellemann, C., Henriksen, R., Kreilhuber, A., Stewart, D., Kotsovou, M., Raxter, P., Mrema, E., & Barrat, S. (2016). *The rise of environmental crime*. United Nations. https://www.grida.no/publications/344

Nijman, V., Langgeng, A., Birot, H., Imron, M. A., & Nekaris, K. A. I. (2018). Wildlife trade, captive breeding and the imminent extinction of a songbird. *Global Ecology and Conservation*, *15*(July), e00425. https://doi.org/10.1016/j.gecco.2018.e00425

Nijman, V., & Shepherd, C. R. (2007). Trade in non-native, CITES-listed, wildlife in Asia, as exemplified by the trade in freshwater turtles and tortoises (Chelonidae) in Thailand. *Contributions to Zoology*, *76*(3), 207–211. https://doi.org/10.1163/18759866-07603007

Nijman, V., & Shepherd, C. R. (2009). *Wildlife trade from ASEAN to the EU: Issues with the trade in captive-bred reptiles from Indonesia*. https://policycommons.net/artifacts/1375321/wildlife-trade-from-asean-to-the-eu/1989582/

Nijman, V., & Shepherd, C. R. (2010). The role of Asia in the global trade in CITES II-listed poison arrow frogs: Hopping from Kazakhstan to Lebanon to Thailand and beyond. *Biodiversity and Conservation*, *19*(7), 1963–1970. https://doi.org/10.1007/s10531-010-9814-0

Nijman, V., & Shepherd, C. R. (2011). The role of Thailand in the international trade in CITES-listed live reptiles and amphibians. *PLoS ONE*, *6*(3), e17825. https://doi.org/10.1371/journal.pone.0017825

Nijman, V., & Shepherd, C. R. (2015a). Trade of 'captive-bred' birds from the Solomon Islands: A closer look at the global trade in hornbills. *Malayan Nature Journal*, *67*(December), 260–266.

Nijman, V., & Shepherd, C. R. (2015b). *Adding up the numbers: An investigation into commercial breeding of Tokay Geckos in Indonesia*. http://www.trafficj.org/publication/15_Adding_up_the_Numbers_Tokay Geckos.pdf

Nogueira, S. S. C., & Nogueira-Filho, S. L. G. (2004). Captive breeding programs as an alternative for wildlife conservation in Brazil. *People and Nature*, *11*, 171–190. https://doi.org/10.7312/silv12782-011

Nogueira, S. S. C., & Nogueira-Filho, S. L. G. (2011). Wildlife farming: An alternative to unsustainable hunting and deforestation in neotropical forests? *Biodiversity and Conservation*, *20*(7), 1385–1397. https://doi.org/10.1007/s10531-011-0047-7

Nuno, A., Blumenthal, J. M., Austin, T. J., Bothwell, J., Ebanks-Petrie, G., Godley, B. J., & Broderick, A. C. (2018). Understanding implications of consumer behavior for wildlife farming and sustainable wildlife

trade. *Conservation Biology*, *32*(2), 390–400. https://doi.org/10.1111/cobi.12998
Olmedo, A., Milner-Gulland, E. J., Challender, D. W. S., Cugnière, L., Dao, H. T. T., Nguyen, L. B., & Nuno, A. (2020). A scoping review of celebrity endorsement in environmental campaigns and evidence for its effectiveness. *Conservation Science and Practice*, *2*(10), e261. https://doi.org/10.1111/csp2.261
Pietersen, D. W., McKechnie, A. E., & Jansen, R. (2014). A review of the anthropogenic threats faced by Temminck's ground pangolin, *Smutsia temminckii*, in southern Africa. *South African Journal of Wildlife Research*, *44*(2), 167–178. https://doi.org/10.3957/056.044.0209
Rizzolo, J. B. (2021). Effects of legalization and wildlife farming on conservation. *Global Ecology and Conservation*, *25*(January), e01390. https://doi.org/10.1016/j.gecco.2020.e01390
Robinson, J. E., Griffiths, R. A., St. John, F. A. V., & Roberts, D. L. (2015a). Dynamics of the global trade in live reptiles: Shifting trends in production and consequences for sustainability. *Biological Conservation*, *184*(April), 42–50. https://doi.org/10.1016/j.biocon.2014.12.019
Robinson, J. E., Griffiths, R. A., St. John, F. A. V., & Roberts, D. L. (2015b). Captive reptile mortality rates in the home and implications for the wildlife trade. *PLoS ONE*, *10*(11), e0141460. https://doi.org/10.1371/journal.pone.0141460
Sas-Rolfes, M., Challender, D. W. S., Hinsley, A., Veríssimo, D., & Milner-Gulland, E. J. (2019). Illegal wildlife trade: Scale, processes, and governance. *Annual Review of Environment and Resources*, *44*(1), 201–228. https://doi.org/10.1146/annurev-environ-101718-033253
Scheffers, B. R., Oliveira, B. F., Lamb, I., & Edwards, D. P. (2019). Global wildlife trade across the tree of life. *Science*, *366*(6461), 71–76. https://doi.org/10.1126/science.aav5327
Shairp, R., Veríssimo, D., Fraser, I., Challender, D. W. S., & MacMillan, D. (2016). Understanding urban demand for wild meat in Vietnam: Implications for conservation actions. *PLoS ONE*, *11*(1), e0134787. https://doi.org/10.1371/journal.pone.0134787
Shepherd, C. R., Stengel, C. J., & Nijman, V. (2012). *The export and re-export of CITES-listed birds from the Solomon Islands*. https://policycommons.net/artifacts/1925016/the-export-and-re-export-of-cites-listed-birds-from-the-solomon-islands-pdf-15-mb/2676788/
Sinovas, P., Price, B., King, E., Hinsley, A., & Pavitt, A. (2017). *Wildlife trade in Amazon countries: An analysis of trade in CITES-listed species*. https://doi.org/10.13140/RG.2.2.33501.00482
Summerell, A. E., Frankham, G. J., Gunn, P., & Johnson, R. N. (2019). DNA based method for determining source country of the short beaked echidna (*Tachyglossus aculeatus*) in the illegal wildlife trade. *Forensic Science International*, *295*(February), 46–53. https://doi.org/10.1016/j.forsciint.2018.11.019
Tensen, L. (2016). Under what circumstances can wildlife farming benefit species conservation? *Global Ecology and Conservation*, *6*(April), 286–298. https://doi.org/10.1016/j.gecco.2016.03.007
Thomson, J. (2008). *Captive breeding of selected taxa in Cambodia and Viet Nam: A reference manual for farm operators and CITES authorities*. https://policycommons.net/artifacts/1935446/captive-breeding-of-selected-taxa-in-cambodia-and-viet-nam/2687216/
TRAFFIC. (2008). *What's Driving the Wildlife Trade? A Review of Expert Opinion on Economic and Social Drivers of the Wildlife Trade and Trade Control Efforts in Cambodia, Indonesia, Lao PDR and Vietnam*. East Asia and Pacific Region Sustainable Development Discussion Papers. East Asia and Pacific Region Sustainable Development Department, World Bank, Washington, DC.
van der Ven, H., & Cashore, B. (2018). Forest certification: The challenge of measuring impacts. *Current Opinion in Environmental Sustainability*, *32*(June), 104–111. https://doi.org/10.1016/j.cosust.2018.06.001
van Uhm, D. (2018). Wildlife and laundering: Interaction between the under and upper world. In Spapens, T., White, R., van Uhm, D., & Huisman, W. (Eds.). *Green crimes and dirty money*. Chapter 10 (pp. 197–214). Routledge.
Vera, F. W. M. (2009). The shifting baseline syndrome in restoration ecology. In Hall, M. (Ed.). *Restoration and history*. Chapter 9 (pp. 98–110). Routledge.
Veríssimo, D., Sas-Rolfes, M., & Glikman, J. (2020). Influencing consumer demand is vital for tackling the illegal wildlife trade. *People and Nature*, *2*(December), 872–876. https://doi.org/10.1002/pan3.10171
Veríssimo, D., & Wan, A. K. Y. (2019). Characterizing efforts to reduce consumer demand for wildlife products. *Conservation Biology*, *33*(3), 623–633. https://doi.org/10.1111/cobi.13227
Wallen, K. E., & Daut, E. (2018). The challenge and opportunity of behaviour change methods and frameworks to reduce demand for illegal wildlife. *Nature Conservation*, *26*(April), 55–75. https://doi.org/10.3897/natureconservation.26.22725
Wang, W., Yang, L., Wronski, T., Chen, S., Hu, Y., & Huang, S. (2019). Captive breeding of wildlife resources—China's revised supply-side approach to conservation. *Wildlife Society Bulletin*, *43*(3), 425–435. https://doi.org/10.1002/wsb.988
WCS, Vietnam. (2017). *Commercial wildlife farms in Viet Nam—A problem or solution for conservation*. https://programs.wcs.org/beta/Resources/Publications/Publications-Search-II/ctl/view/mid/13340/pubid/DMX3329600000.aspx
Webb, G., Brien, M., Manolis, C., & Medrano-Bitar, S. (2012). Predicting total lengths of spectacled caiman (*Caiman crocodilus*) from skin measurements: A tool for managing the skin trade. *Herpetological Conservation and Biology*, *7*(April), 16–26. https://herpconbio.org/Volume_7/Issue_1/Webb_etal_2012.pdf
White, A., & Yifan, J. (2022). *Second draft revision of China's wildlife protection law 'a big step backwards'*. China Dialogue. https://chinadialogue.net/en/nature/second-draft-revision-of-chinas-wildlife-protection-law-a-big-step-backwards/
Whitfort, A. (2021). COVID-19 and wildlife farming in China: Legislating to protect wild animal health and welfare in the wake of a global pandemic. *Journal of Environmental Law*, *33*(1), 57–84. https://doi.org/10.1093/jel/eqaa030
Williams, V. L., & Sas-Rolfes, M. (2019). Born captive: A survey of the lion breeding, keeping and hunting industries in South Africa. *PLoS ONE*, *14*(5), e0217409. https://doi.org/10.1371/journal.pone.0217409

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Appendix S1: Rapid literature search methodology.

How to cite this article: Meeks, D., Morton, O., & Edwards, D. P. (2024). Wildlife farming: Balancing economic and conservation interests in the face of illegal wildlife trade. *People and Nature*, *00*, 1–12. https://doi.org/10.1002/pan3.10588