Cambridge Prisms: Extinction

www.cambridge.org/ext

Review

Cite this article: Fraser IM, Roberts DL and Brock M (2023). The economics of species extinction: An economist's viewpoint. *Cambridge Prisms: Extinction*, **1**, e20, 1–10
https://doi.org/10.1017/ext.2023.18

Received: 03 November 2022
Revised: 25 May 2023
Accepted: 30 June 2023

Keywords:
species extinction; economic valuation; bioeconomic models; renewable resources; scale

Corresponding author:
Iain Fraser;
Email: i.m.fraser@kent.ac.uk

The economics of species extinction: An economist's viewpoint

Iain M. Fraser[1], David L. Roberts[2] and Michael Brock[3]

[1]School of Economics and Durrell Institute of Conservation and Ecology (DICE), University of Kent, Canterbury, UK; [2]DICE and the School of Anthropology and Conservation, University of Kent, Canterbury, UK and [3]School of Economics, University of East Anglia, Norwich, UK

Abstract

There is growing evidence to suggest that there is an increase in species extinction occurring globally. In this article, we briefly review the literature on the economics of species extinction, examining what is meant by extinction before explaining how economics has conceptualised this. The initial economics literature on species extinction focuses largely on renewable resources, in particular fisheries, but has subsequently evolved to cover many aspects of biodiversity across all physical scales, employing an increasing array of methodological tools. We also consider aspects of cultural and societal extinctions (e.g. local languages, local knowledge) and how this is positively correlated with loss of biodiversity, as well as an economist's outlook on the potential to re-capture value post-extinction.

Impact statement

In this article, we review aspects of the economic literature that examine species extinction. We focus on research that explicitly considers species extinction, which sits within the wider literature on species loss considered by the literature on the economics of biodiversity. Our review explains how economics helps researchers and policymakers understand why species extinction can occur and what needs to be done to potentially reverse the losses that we are currently experiencing. The importance placed upon correctly valuing species within the wider economy is seen as key to reversing many of the pressures that cause their decline and eventual extinction. The valuation task facing economists is significant and not necessarily viewed positively by all in the conservation community. However, the practice of economic valuation of species has improved significantly and as such its role in helping to reduce future species extinction is enhanced. Furthermore, a fundamentally important change is occurring within environmental economics and how the profession sees the relationship between the economy and the environment. Specifically, the economy is now viewed as being embedded in the environment and this fundamental change in emphasis means that environmental economics (and economics in general) should now see species extinction less as one-off events with limited repercussions, but rather as signals that current patterns of economic activity are unsustainable and pushing the environment beyond its ability to support future generations. As such, we contend that the wider conservation community should positively embrace the contribution that economics can make to tackling species extinction given global recognition of the urgency of transitioning to nature-positive economies.

Introduction

The general consensus is that the planet is entering a time of immense environmental upheaval. Birdlife International (2022) highlights that one in eight bird species is threatened with extinction, while Dasgupta (2021) reports the current rates of species extinction could be between 100 and 1,000 times greater than the underlying trend (Pimm et al., 2014; IPBES, 2019). This is happening at the same time as the planet is also experiencing unprecedented climate change, and the links between this and species extinction have been increasingly noted by scientists and economists alike (e.g. Thomas et al., 2004; Urban, 2015; Ng, 2019; Sol, 2019; Tilman, 2022). However, the economic analysis and modelling of species extinction predate the emergence of climate change concerns, with much of the early economic literature focussing on the (un) sustainable management of renewable natural resources such as fisheries and forests (e.g. Gordon, 1954; Clark, 1973). Subsequently, economists have examined issues surrounding resource management extensively, although with relatively little exact focus on species extinction (Conrad, 2018).

In this article, we review the economic literature that focusses on species extinction. Thus, although the scope of economic research that examines species loss is large, especially within the literature on the economics of biodiversity, we focus upon the specific research area that explicitly considered species extinction. It is this literature that we review in this article, revealing how economists frame species extinction: why it might occur; why extinction occurs; and what we can do to avoid it occurring.

We define species extinction, following the IUCN (2012):

> *A taxon is Extinct when there is no reasonable doubt that the last individual has died. A taxon is presumed Extinct when exhaustive surveys in known and/or expected habitat, at appropriate times (diurnal, seasonal, annual), throughout its historic range have failed to record an individual. Surveys should be over a time frame appropriate to the taxon's life cycle and life form.* (p. 14).

Importantly, we are not concerned with the natural process of evolution and extinction; rather we focus on extinction due to human action, and where such action creates rates of extinction higher than would be expected to occur naturally (Dasgupta, 2021). We also do not consider species extinction that might be considered beneficial. For example, society *might* positively value the eradication of anopheles mosquitoes that transmit malaria to humans. However, given the benefits provided by mosquitoes, such as pollination, there are likely to be different ways of dealing with the diseases that anopheles mosquitoes spread (Peach, 2019). Models of economic enhancement are built on the importance of discovering new technology that in turn results in older technologies becoming obsolete. It is accepted that technological advances will result in technological extinction. In this article, we are not characterising extinction as an economic process of creative destruction (Ormerod, 2007).

In the next section, we review the key economic underpinnings that explain why species extinction can occur. In Section 3, we consider the various ways in which an economics analysis of species extinction has evolved in the literature. We then consider species extinction across contexts including geographical scales: local; national; and global, indirect impacts and post-extinction. Finally, Section 5 concludes.

The economics of species extinction: Basic concepts

How efficiently or inefficiently society 'uses' the environment, including species, can be explained by the role of property rights (or more importantly the lack of), externalities and market failure (Baumol and Oates, 1988; Hanley and Perrings, 2019; Dasgupta, 2021; Groom and Turk, 2021). These concepts help economists to explain why many species are frequently undervalued and not protected. There are four main characteristics of a property rights regime: Ownership; Excludability; Transferability; and Enforceability. The absence of any one characteristic will mean that the 'signalling function' of the market will not work, something economists term 'market failure' (Dasgupta, 2021). Take a resource such as a fish stock in the high seas that is not owned by any specific country. No one can be excluded from exploiting the stock, hence the resource may be over-exploited, potentially leading to negative externalities and species extinction (Swanson, 1994). A negative externality is an unintentional side effect of production or consumption that falls on a third party (Baumol and Oates, 1988). Importantly, many externalities do not require there to be a physical link. Barbier (2022) gives the example of habitat destruction in the Brazilian Amazon as a result of illegal logging, land clearing for farming and mining. The loss of the rainforest gives rise to a loss of biodiversity that imposes a negative externality on many in society who will never visit the Amazon. Another example is cited by Dasgupta (2021), who explains that the vicuña, a member of the camelid family, found in the high Andes, was almost hunted to extinction. This situation was only reversed when local communities were granted usufructuary rights to shear and sell the highly valued fibre from the animal. Changing the way the property rights worked meant that the negative externality was 'internalised' (i.e. resolved).

Fundamentally, market failure and the resulting exploitation and possible extinction of species occur because the 'value' attached does not fully capture societal preferences. For example, society may be concerned about excessive and inhumane whale hunting (Cook et al., 2020) or the excessive deforestation of Amazonia (Barbier, 2022; Brouwer et al., 2022), but their concerns (i.e. their preferences) are not reflected in the costs and benefits from the activity as experienced by the perpetrators. Many economists therefore contend that part of the solution is ensuring that we properly value biodiversity and this in turn means the need to undertake valuation so that societal preferences can be properly expressed (Dasgupta, 2021; Ando, 2022).

Deriving values for biodiversity and the environment so that societal preferences can be expressed can be done either by using existing market data or by asking what society would be willing to pay (WTP) to keep something or willing to accept (WTA) for its loss. The economics profession has developed a wide array of techniques to measure the components of total economic value of the environment (OECD, 2018; Hanley et al., 2019). Economics can measure the use value associated with the direct utilisation of a resource such as walking or bird watching in forests, or the indirect use of the forest sequestering carbon. They can also measure non-use values, such as those associated with the continued existence of the birds in the forest without the need for members of society to actually ever visit the forest. Non-use existence values are of particular significance when it comes to species extinction (Alexander, 2000; Lopes and Atallah, 2020). The importance of estimating environmental values is that they can then be fed into policy (e.g. cost-benefit analysis) that might be attempting to deal with the problem of species extinction.

This market failure also arises through the way macroeconomic activity is measured by economists. For example, our metric of 'growth' via changes in Gross Domestic Product (GDP) means many uses of biodiversity are seriously undervalued, contributing to species loss and possible extinction (Dasgupta, 2021; IPBES, 2022). Fundamentally, as argued by Dasgupta (2021), there is a need to understand that the economy is embedded within the environment. This means the environment places real constraints on economic activity, and unless we understand that species extinction is a signal that we are going beyond the real limits of what can be supported, the prospects confronting future generations are bleak.

Some key themes in the literature

Bioeconomic models and species extinction

Initially, economic research focussed on renewable resources (such as fisheries) and associated resource harvesting and management by employing bioeconomic models. A bioeconomic model combines biological and economic features capturing the interaction between the two systems. Early research by Gordon (1954) and Clark (1973) used bioeconomic models to reveal how, under specific circumstances, the economically optimal management of a resource could in fact result in species extinction. As noted by

https://doi.org/10.1017/ext.2023.18 Published online by Cambridge University Press

Polasky and Dampha (2021), it was shown by Clark (1973) that the optimal use of a renewable resource might be to drive the resource stock to zero (i.e. extinction). As noted by Clark (1973), Swanson (1994) arrives at this result given three factors:

> *(i) open access to the resource;*
>
> *(ii) relative price to cost of harvesting the resource; and*
>
> *(iii) relative growth rate of the resource.*
>
> *These three factors are sufficient to produce a bioeconomic model showing that the optimal stock of the resource goes to zero in the steady-state.* (p. 802).

Swanson (1994) significantly extended the model to consider species as productive assets as well 'onshore' biodiversity (e.g. elephants). This extension shows, for example, that a resource (or species) will be kept at a stock level that generates a return equal to the discount rate. What this implies is that resources that have relatively slow growth rates compared to other productive assets in an economy are a potential pathway to 'extinction'. Swanson (1994) summarises this point as follows:

> *The fundamental cause of extinction is that a species will be seen as an inferior asset, and thus be excluded from the human portfolio of assets.* (p. 809).

He also notes that there are several possible reasons for this to occur (i.e. resource mining; land-use conversions; and over-exploitation) and that only species that are of high value to humans and that also have a high growth rate are likely to survive, unless appropriate policy interventions are introduced.

Following the important contribution of Swanson (1994) to model development there have been many additional enhancements. For example, bioeconomic models no longer focus on single species; they are applied to multi-species problems, something seen as necessary both ecologically and economically (Fleming and Alexander, 2003). An example of a multi-species bioeconomic model examining species extinction is provided by Bertram and Quaas (2017). They demonstrate that extinction is never an optimal outcome once the biodiversity value of species abundance is explicitly considered. Other examples include Horan and Melstrom (2011), who consider disease risk and how to undertake culling and translocation as a means to avoid extinction. Melstrom et al. (2022) also consider the trade-off for policymakers between economic development and land-use conservation for an endangered species. Bioeconomic models have also been used to examine existing extinction episodes. This facet of the literature reveals that the variables that combine to generate species extinction are varied and heterogeneous (Bulte et al., 2006). A bioeconomic model has also been used to examine the rise and fall of the Easter Island human population (Brander and Taylor, 1998). Employing the classic predator–prey Lotka–Volterra model, they demonstrate how over-exploitation of the island's natural resource base plus an increasing human population almost led to population collapse. In this case the model is able to demonstrate the interaction between predator (humans) and prey (island resource base) and how mis-management can cause such an extreme outcome.

Even though bioeconomic models can generate results indicating that a species will become extinct, many researchers believe it unlikely that excessive exploitation alone will deliver this (Bulte et al., 2003). This is because the cost of harvesting an ever-rarer species will increase such that the benefits from harvesting are significantly outweighed by the costs. Thus, although many natural resources (such as fisheries) can be seriously depleted as a result of overharvesting, it is unlikely that this solely causes extinction. In most cases, species extinction instead arises when excessive harvesting is compounded by effects such as habitat loss (Pimm et al., 2014). An example of this is considered by Bulte et al. (2003) using a bioeconomic model to examine the likelihood that the Tasmanian Thylacine became extinct because of hunting. Their analysis suggests that the Tasmanian government's 'bounty scheme' policy was unlikely to have led to the extinction of the Thylacine. They further note that ongoing attempts to eradicate invasive species, for example many of the introduced species in Australia, do not often drive these species populations to zero. Typically, what can and does cause species to become extinct, particularly at a local level, is poor resource management (Ando and Langpap, 2018).

Economic value and species extinction

The determination, meaning and use of economic value, both use and non-use values, plays a prominent role when examining species extinction. For example, with regard to bioeconomic models, Alexander (2000) extends the work by Swanson (1994) to examine the importance of non-consumptive existence values (i.e. non-use values) of a species. This extension meant that the value attributed to a particular species no longer mattered only to the resource owner. Non-use values are, by definition, values attributed to a species by society even when they do not directly use or come into contact with that species. This adjustment in how the values attached to a species are understood essentially changed the 'golden rule' equation that is key to expressing how a specific renewable resource should be managed over time, potentially reducing the propensity for extinction. Another example is Bertram and Quaas (2017), who extend Alexander (2000) by including an aggregate value for biodiversity broadly defined in a multi-species bioeconomic model.

The inclusion of non-market values in bioeconomic models is (largely) motivated by the extensive non-market valuation literature. In this context, as a species nears extinction, economists recognise the rising value is in part being driven by increasing species rarity, something that has been extensively researched (see Richardson and Loomis (2009) for a meta-analysis of the literature). Importantly, there are high economic gains from promoting the value of rarity, as has been shown in areas such as ecotourism and safaris (Pathirana et al., 2021) and from rare sightings (Brock et al., 2021). Although it is very possible that tourism can offset and reverse extinction threat (Naidoo et al., 2016), there is also an argument made within the literature of a more perverse relationship, whereby manufacturing rarity creates potential extinction scenarios (Courchamp et al., 2006; Angulo and Courchamp, 2009). This effect is known as the anthropogenic Allee effect and essentially assumes that, given rarity is valued, this could in turn lead to the extinction of rare species. A study by Lyons and Natusch (2013) concluded that an emphasis on the value of rare species could lead to greater levels of exploitation. A more recent study on this effect by Krishna et al. (2019) suggests that the trade in rare wild birds (plus extensive habitat loss) could lead to several species becoming extinct. There are also studies that explicitly consider species extinction. Examples include Bristol et al. (2014) (Seychelles) and Zander et al. (2022) (Australia), who both consider people's WTP to avoid species extinction.

An important conceptual aspect of non-market value and species extinction is that of 'Option Value' (Arrow and Fisher, 1974). This indicates the worth people attach to the possibility of utilising a resource in the future, even if they have not done so until now (or do

https://doi.org/10.1017/ext.2023.18 Published online by Cambridge University Press

so very infrequently). Intrinsically linked to this value is the concept of uncertainty and irreversibility (Pindyck, 2007). Dasgupta (2021) provides a neat example to explain the meaning of option value. An economic agent wishes to drain a wetland. However, if they then wanted to reinstate the wetland, they would face significant costs and these would be greater than if the wetland had been retained. In addition, over time our economic agent learns that the wetland is more valuable than initially thought. This means that the value of the wetland is greater than our economic agent understands today. The extra value that is revealed over time is the option value associated with this environmental asset. In addition, by retaining biodiversity, the diversity of an ecosystem is also maintained and that may well be of fundamental importance in ensuring the capacity and function of an ecosystem in the future. In doing so, we are maintaining the resilience of the ecosystem which can help in dealing with exogenous stresses and shocks. As such, maintaining diversity and species richness is akin to putting in place an insurance policy to protect ecosystem function (King et al., 2019).

The meaning of value can and does differ between disciplines as there is divergence between 'human' and 'ecological' value for a species or natural resource. This 'value' differential placed on preventing extinction can differ vastly depending on the extent to which (ecologically) the species performs the role of a keystone species (White et al., 1997; Morse-Jones et al., 2012), or (anthropocentrically) if we perceive there to be a good proxy species to take its place in our valuation. Importantly, the value for many species within an ecosystem can adjust to the loss of one species, irrespective of how crucial this species role is within the ecological system itself. The reverse can also be true when we perceive a species to be 'charismatic', and this creates a 'human-value' bias towards megafauna or aesthetically pleasing creatures (Christie et al., 2006; Jacobsen et al., 2008).

Underpinning the conceptual importance of economic valuation for species extinction is the non-market valuation literature. In this literature, there are many studies that examine the economic costs of biodiversity loss, and specific studies considering species extinction (e.g. Bristol et al., 2014; Zander et al., 2022). As noted by Hanley and Perrings (2019) by undertaking economic valuation it is feasible to assess the costs of biodiversity loss and in turn provide evidence in support of policies to correct market failure and externalities. However, despite the key role that valuation can play in identifying the costs of species loss and extinction, there are significant theoretical and practical limitations that need to be addressed. For example, Hanley and Perrings (2019) note that, although we can value the potential extinction of a single species, there are still significant limitations in our understanding of the ecological consequences and the associated value that needs to be attributed to '*diversity within functional groups of species*' (p. 370). Even with some evidence about the rate of extinction for a functional group of species such as insects, Cardoso et al. (2020) estimate these economic benefits of \$500 billion per annum are still only likely to be an underestimate given our lack of understanding about insects and the key role they play in global ecosystems.

Other limitations that need to be considered when undertaking non-market valuation include the difficulty of transferring values by benefit transfer for individual species in specific contexts (Hanley and Perrings, 2019); also, how species are presented within valuation exercises, the ability of survey participants to construct meaningful preference for species they know very little about and scale and scope of the issue being examined (Dasgupta, 2021). How valuation research is conducted in practice also needs to re-examine the groups in society who are targeted. Ando (2022), for example, argues that such research frequently fails to include underrepresented groups in society, as well as seeing the need for more research to employ a WTA approach when considering biodiversity loss. This matters, as the property rights associated with biodiversity loss imply that a WTA approach is appropriate and that improvements in how we conduct non-market valuation mean that the practical guidance that has dominated economic research over the last 30 years need to be reconsidered.

The elasticity of substitution and extinction

The discussion of value will frequently frame the worth of a rare species relative to other species or economic resources. Here, the key economic principle of product substitution becomes relevant. The important economic parameter that captures choices with regard to extinction is therefore the elasticity of substitution, which can be thought of as the ease (or difficulty) with which we can respond to the removal of one species in society by utilising or substituting another (or others) as a replacement. In the world of sustainable growth and development, this is inherently linked to the debate around weak and strong sustainability, and the extent to which our 'natural capital' assets can be substituted for other forms of capital (i.e. physical capital or human capital) (Dasgupta, 2021). The elasticity of substitution also allows us to consider whether alternatives to a resource facing extinction can be seen as 'perfect', offering the same function and values as its predecessor, or 'imperfect' in having limitations through its replacement. Thus, the elasticity of substitution between general consumption goods and biodiversity is likely to work as follows. If preferences are assumed to be elastic (i.e. the elasticity of substitution is greater than unity), then as society becomes richer, the optimal amount of biodiversity decreases, while it increases if inelastic.

The importance of substitution elasticity is apparent when we consider irreversibility. Conrad (2018) employed a real options approach to examine the optimal timing of a policy intervention to alleviate species extinction. This analysis requires a value to be placed on a species if it becomes extinct. If the elasticity of substitution for a species becomes very large, then the resulting value attached to a species if it is likely to become extinct is also very (very) large. Conrad (2018) examined the case of the California condor, and in his analysis the value placed on this species is \$4,800,000. Importantly for policy, this analysis indicates that the number of birds left in the wild should have resulted in a captive breeding programme sooner than occurred. Xu (2021) extends this by proposing the optimal intervention policy. Still linked to the condor, they include the conditions by which the intervention responds to population change and consider how such intervention flexibility facilitates a more accurate value of the intervention policy itself. The multi-species bioeconomic model of Bertram and Quaas (2017) also considers elasticity of substitution, in this case between stocks of species. In their model, if this parameter is greater than one, then they can show that the value of the species to their biodiversity index tends to infinity as the species heads towards extinction.

The elasticity substitution is also vital within the wider environmental economics literature. For example, Sterner and Persson (2008) demonstrate that the elasticity of substitution is as important as the discount rate in relation to results generated when conducting climate-economy modelling. In economic analysis the discount rate allows costs and benefits that are accrued in different time periods to be expressed in value terms today (OECD, 2018). Importantly, the smaller (higher) the discount rate, the more (less)

https://doi.org/10.1017/ext.2023.18 Published online by Cambridge University Press

we are concerned about future time periods in terms of the costs and benefits that occur. A discount rate is always required for cost-benefit analysis that occurs over time and as such is a key parameter when examining conservation management options and associated species protection (Armsworth, 2018).

Another role that the elasticity of substitution plays in helping to understand species extinction is in regard to the environmental Kuznets Curve (EKC). The EKC is an empirical and theoretical relationship that suggests that as society becomes wealthier, the damage imposed on the environment that occurs as a result of economic growth is reversed (Shibayama and Fraser, 2014). Importantly, the EKC has been used to examine species extinction by Sol (2019) using 2016 data on the IUCN Red List species. Sol (2019) finds no evidence in support of the EKC, wealthier countries do not provide examples of a decline in species loss or rates of extinction, and as a result, Sol (2019) advocates that we need to reconsider how we think about the relationship between the environment and the economy much in keeping with Dasgupta (2021) and the IPBES (2022).

Finally, in terms of sustainability and substitutability, Cohen et al. (2019) provide interesting empirical evidence, reporting.

> *We find that most available substitutability estimates do not stand up to careful scrutiny. Moreover, accurate substitutability estimates are even more difficult to produce for unpriced or mispriced resources. Finally, we provide evidence from industrial energy use, and agricultural land use, that suggests substitutability of natural capital with other forms of capital may be low to moderate.*

These findings matter in that they not only provide empirical support for taking a 'strong sustainability' perspective, but they also justify Dasgupta (2021) arguing that the economy is embedded within the environment and the limits this imposes must be recognised.

Policy-making and extinction

For economists, another particularly important part of the literature relates to policy-making when faced with species extinction. As an example, some of the earlier literature (Swanson, 1994; Kremer and Morcom, 2000) examined the possibility of elephants becoming extinct as a result of poaching to harvest ivory. Here, economic analysis needs to consider not only the elephant as a biological resource but also the ivory as a non-renewable resource that can be stored over time (Bulte and van Kooten, 1999; Kremer and Morcom, 2000). This strand of research has proven particularly informative as it helps to direct policymakers on how best to simultaneously address issues of species management when faced with negative externalities associated with illegal behaviour. These potential negative consequences are summarised by Mason et al. (2012):

> *When supply from private stores competes with supply from "wild populations" (in nature) and when speculators are able to collude, it may be optimal to coordinate on an extinction strategy.* (p. 180).

There continues to be extensive research on how best to deal with illegal poaching and its associated impact on wildlife. For example, Conrad and Lopes (2017) develop a bioeconomic model and examine the typical policy options advocated to deal with poaching. Do et al. (2021) have estimated the demand for ivory yielding an own price elasticity of 0.4 plus an inelastic supply elasticity. These results imply that policies that attempt to reduce demand will need to be large, given the inelastic supply curve for ivory. However, as noted by Ando and Langpap (2018), bioeconomic models such as those used by Conrad and Lopes (2017) can generate qualitatively different results when examining various policy options to deal with species extinction. This means that there is generally no one policy that can solve all species extinction issues. For a broader review of the literature on illegal wildlife trade, see MacFarlane et al. (2022).

Another aspect of policy that has been examined is the speed at which management regimes are put in place to protect an endangered species. Lewis et al. (2019) show that upfront investment in conservation management, although expensive, can yield faster benefits and the results are more highly valued by the public. Similar findings in terms of attempting to reverse possible species extinction are reported by Lewis et al. (2022). They again demonstrate that immediate actions to reverse species decline will in turn generate larger benefits for society than policy actions that gradually attempt to reverse a path towards extinction.

Returning to the issue of multiple species, Gordon et al. (2020) examine how there is a bias in terms of which taxonomic groups attract conservation funding. The implication of this research is that certain, possibly less, charismatic species may well be close to extinction but may not receive the appropriate funding or policymaker support. Gordon et al. (2020) observe that a more effective approach would be to spread the funding across a greater number of species. However, they also note that charismatic species can act as flagships that can in turn help with wider conservation objectives (Veríssimo et al., 2009).

It is also the case that government policy can in fact create perverse incentives with regard to species conservation. This possibility is frequently noted in relation to the US Endangered Species Act which creates an incentive for landholders who have an endangered species on their land to extinguish the species (Byl, 2019). However, the US government has in this case attempted to reduce these negative incentives by introducing legislation via the use of safe harbour programmes. Another example is provided by Lopes and Atallah (2020), who examine the impact of the potential eviction of the indigenous Soligas tribe from a protected area on the local tiger population. Their ecological-economic model yields a result that sees the tiger population facing extinction. In contrast, if the Soligas tribe retain the property right to manage the land, then the localised extinction is avoided. Another example is provided by Barbier (2022), who notes that based on data from the OECD, government expenditure on agricultural support is having negative effects on the environment and species. Barbier (2022) argues that this is happening because of the 'under-pricing' of nature and the resulting negative externalities and market failure.

Finally, when it comes to the framing of policy and species extinction Turnhout and Purvis (2020) sound a note of caution. As they observe, focussing on species and biodiversity loss as species extinction enables communication of the crisis being faced. However, loss does not always imply extinction and they cite the Global Analysis of the IPBES (2019) as an example. IPBES (2019) produced an estimated 1 million species facing extinction. Much like the ecological footprint, a big number has an impact, especially with the media, but it does not help frame or implement policy solutions. But how the number is arrived at can become the focus of attention which in turn detracts from the problem at hand. Furthermore, Turnhout and Purvis (2020) argue that there is no point in suggesting that a given number of species are going to become extinct each year. The number does not help inform important policy choices, such as resource allocation and management strategies, and it does not reflect the actual costs associated with the loss. The simple message can in fact get in the way of what really matters.

Extinction and context

Geographical scale

Local scale evxtinction

The loss of a species or resource at a local level has two conflicting impacts when considering values lost. On the one hand, individuals within society frequently feel pride or attachment to natural, cultural or geographic commodities that are deemed 'local'. In the case of biodiversity, even common species of wildlife evoke a special sense of value among people (Clucas et al., 2015; Brock et al., 2017). Even when the public do not interact with local species, they can express a significant value for protecting the species. For example, Vincent et al. (2014) presented local estimates of WTP to prevent extinction associated with forest loss that has very limited public access. In this case the estimate of WTP capture measure of existence, option and bequest values.

Species that are deemed unique or rare on a national level but exist in a specific area or region can not only provide revenues in the form of tourism but also instil a sense of identity within local populations of people. Furthermore, there is evidence that this type of 'nature connectivity' is pivotal to establishing a broader environmental awareness among people (Dutcher et al., 2007). Thus, the extinction of key species delivering such connections could be harmful for the 'reach' of tackling environmental degradation.

The counterargument here is that local extinction (if indeed just at a particular local level) is less catastrophic than at national or global levels because there exists an opportunity to reintroduce this resource in the future. Although management and leadership are key to the success of this (Sutton, 2015), and that success also differs considerably by taxonomy (Wolf et al., 1996), this in essence weakens the concerns over irreversible action. It also reduces our concerns over possible substitutes if we have reassurance that one day reintroduction could be applied if required. There is also a cost argument in terms of local extinction relating to when we should stop managing an area with a specific species in mind. As explained by Chadès et al. (2008), once a species becomes difficult to detect, the benefits from management and monitoring become uncertain and as such scarce resources may well be better employed to protect other vulnerable groups.

It is also important to note that the scale of species distribution is a relevant factor. For example, if a species is distributed globally, then the implications of localised extinction may have large local consequences (e.g. loss of direct use values derived from a species), and yet the global impact is far less significant. However, for species that are endemic in specific regions and only exist in this geographically constrained area, then not only does the extinction have an impact locally, but there are also global consequences. In cases such as this, the value attached to a particular species facing extinction will be significant.

Although somewhat true at any level of the extinction scale, an important consideration here is the (in)ability for species populations to be measured over time. Field et al. (2004) note that this can be an important consideration for the value attributed to preserving a species, and if it is hard to establish the exact trajectory of population change, then this is likely to have a large bearing on the willingness for policy to allocate time and resources into preventing a species extinction.

One way in which localised species extinction can be reversed is by undertaking reintroductions. For biodiversity, we have seen many successful reintroductions. In the UK alone, there have been reintroductions of the Beaver, Red Kite and Wild Boar since the 1990s (Goulding et al., 2003) plus the Seychelles Paradise Flycatcher (Bristol et al., 2014) and White-tailed eagles to Ireland (O'Rourke, 2014). An interesting question that arises with reintroduction/re-creation at the local levels is that do they lead to a return of values that were present before a species initially became extinct in the local area? This is a difficult question to answer and is likely to be context-specific. But it is important to try and recognise that the entity itself is not always what drives the value pre-extinction, but this can be interlinked with cultural or societal values which, when extinction occurs, may not be recreated by a return of a species or close substitute.

National/regional scale extinction

To some extent, the issues raised at a local scale will also apply at the regional and national scales. However, the extent to which society establishes an affiliation with wildlife on a national scale could be seen as diminished when compared to that at a local level. Nevertheless, it is true to assume that society will consider a species to be an 'inhabitant' (indigenous) so long as it lives or breeds somewhere within the confines of a given geographical boundary, securing any social or cultural tie to that animal. Indeed, this is one reason that many species can appear on logos, flags or motifs to represent their region, state or nation. Furthermore, if this standpoint is to be followed, we might assume that national or regional attachment to a species is as strong (if not stronger) than at a local level. An example of how species protection is more highly valued for native versus non-native species is provided by Hanley and Perrings (2019) in regard to Danish conservation for birds.

Also, when it is necessary to consider reintroductions at the national scale it is probable that this will require far greater international co-ordination, and this brings with it additional legal and/or political difficulty. In an effort to enable national-level reintroductions, the IUCN/SSC (2013) has developed guidelines. Furthermore, if a species has disappeared from an entire country or region, this may be because of ecological reasons (i.e. due to changes in the climatic conditions) or human-induced effects. In either case, reintroductions might be supported. It is also the case that facilitated species migration might be supported as climate change reduces the viability of an existing population (Ando and Langpap, 2018). Finally, there is also a growing movement to rewild in an effort to protect biodiversity (Schulte to Bühne et al., 2022). In some cases, the reintroductions can be like for like or it might be relatively close to a now regionally extinct species. In these cases, the economic analysis not only examines the costs of the reintroduction, but it can consider the associated benefits (ecological and economic).

Global-scale extinction

International or global-scale extinction is perhaps what most people think of when they think of the term 'extinction'. Of course, this form of extinction creates the biggest issues when considering 're-creation' or reintroduction, purely because the absolute level of the resource is tending to zero. However, whether this is more or less problematic to solve is an open question. On the one hand, economists would recognise the greater value attached to preventing global extinction, and thus a higher propensity to direct resources to prevent this from occurring. From an environmental valuation standpoint, the fact that those with both use and non-use values would drive action to prevent extinction when at a global level would support that view. On the other hand, preventing global extinction, as with much global action, requires a greater sense of co-operation and co-ordination, which may be very hard to facilitate. As shown in most applied public or common good problems, the actors who need to prevent the trend to extinction are regularly those who benefit from the resource (i.e. for food, tourism, medicine, trade), or would have to bear the biggest social costs from protecting the species (Kraak, 2011).

As noted earlier, the issue of global extinction has increasingly been considered by economists in relation to climate change. Stern (2007) explicitly raised the possibility of human extinction in his seminal work on the economics of climate change. He explicitly stated that humanity faced a 10% chance of extinction by 2,100. This subjective assessment and those of other researchers are summarised by Polasky and Dampha (2021). Subsequently, Weitzman (2009) examined how economists can deal with the difficulty inherent in modelling rare but catastrophic events.

More recently, Ng (2019) reconsiders the link between human extinction and its importance in terms of climate change. Like Stern (2007), he argues that by reducing the probabilities of global extinction, the resulting benefits to future generations will hugely dominate any benefits that will be forthcoming from protecting future consumption. The result occurs simply because reducing this probability increases future expected utility and, as future utilities are important, they should only be discounted at a very low level especially when compared to future consumption. In fact, how to treat future generations has been the topic of intensive philosophical debate with the view being expressed that we should avoid discriminating against future generations simply because they are born later (Polasky and Dampha, 2021; Stern, 2022). As such there is a strong argument for setting the rate of pure time preference to be equal to zero as there is no basis for assuming any form of pure time preference. However, as discussed by Stern (2022), if we assume there is some non-human cause of human extinction, for example, as the result of an asteroid hitting the earth, then the rate of pure time preference can be set greater than zero. However, if the cause of the possible extinction is endogenous, that is as a result of human action, then employing a value of zero remains valid. We can think of this as being the difference between expected extinction versus sudden extinction.

Finally, despite all the discussion about the possibility of human extinction as a result of climate change, there is in fact very little research that attempts to explain how this may occur. An example of recent research attempting to place the subjective probabilities in context is by Beard et al. (2021), who consider the claimed threats to humanity and how they can be linked to climate change. In addition, Rising et al. (2022) considered issues on how to economically evaluate the risks associated with climate change given the divergence between natural scientists who frequently claim an increased likelihood of extinction, whereas economists present empirical evidence that does not support many of the more extreme outcomes.

Recapturing values post-extinction

The final aspect of extinction economics we consider is how it might be possible to re-create values once a resource has become extinct. Of course, the scope of opportunities here largely depends on the scale of the extinction, but also the technological capabilities to re-create a resource. For example, recent advances in genetic science mean scientists believe it may be possible to completely re-produce species that have long been extinct (Waterhouse and Mitchell, 2022).

Although 'capability' is one variable that determines the ability to re-create value, there are others that we must consider as an economist. For example, it is highly possible that people within a region may not attach the same worth from a reintroduced population as they did from the original 'indigenous' group. By contrast, the argument of 'bio-prospecting' (Rausser and Small, 2000) can mean that reintroductions at a later time or in a different region might offer equal or even better opportunities for humans (i.e. propensity to create medicine or production) than in its original state. The argument here is that re-creation, like extinction, can be argued through 'bio-prospecting' and preserving the opportunities for future economic gains from natural resources.

How do we value something when it is extinct? Reintroduction programmes, value of museums Beidleman, 2004) are all tangible ideas, but this does omit the connectivity or cultural importance attached to the retention of a species. This is already being reflected in the changing role of natural history museums (Winker, 2004).

Concluding comments

In this article, we have reviewed various concepts and ideas as they relate to the economics of species extinction. Economic principles such as property rights, externalities and the appropriate functioning of markets help to explain the analysis and framing that is found in the economic literature on species extinction. This is in conjunction with an acknowledgement and understanding of how values that are attached to species that are close to extinction impact society. Importantly, economics can help to explain many of the potential sources of pressure that lead to our increasing worries about species extinction. Fundamentally, it is typically a combination of reasons that explain why a particular species is at risk from extinction. In the vast majority of cases, it is competition for resources and land use (i.e. the opportunity cost) that are key. At the same time, economics is also able to provide solutions to deal with species extinction that are varied and inventive.

Our review has been intentionally focussed on two areas of economic research that consider species extinction: bioeconomic models and valuation studies. As we have noted, the bioeconomic literature has developed rapidly since the initial fisheries-based work of Gordon (1954) and Clark (1973). The bioeconomic models that are now being used are far more sophisticated, extending to multi-species and policy-oriented modelling. These models have in turn yielded a far more nuanced understanding of species extinction. We have also considered how economics values species extinction. There has been an increase in studies generating values that consider species extinction as well as species rarity.

Finally, we acknowledge that our review is far from exhaustive, but it does reveal the important role that economics can play in understanding the problems that need to be addressed if we are to deal with species extinction. We also note that the economic literature on species extinction is part of a much bigger literature on biodiversity loss in general. And as we have already explained, if we are to avert a global crisis as a result of biodiversity loss, we must heed the key message in Dasgupta (2021), that is, we must change how we see the relationship between the economy and the environment.

Glossary of economic terminology

Benefit Transfer — An economic valuation methodology which takes estimates obtained from previously conducted research and employs them in a new context.

Bequest Value — An example of non-use value, where individuals attach value to the fact that a resource will be available to future generations.

https://doi.org/10.1017/ext.2023.18 Published online by Cambridge University Press

Capital (human, natural, physical)	A measure of the stock of a resource required to produce output and/or the component parts of an ecosystem.
Discounting	A mathematical method used to convert future costs and/or benefits into present values using a discount rate.
Discount Rate	The discount rate is the interest rate employed when discounting future costs and benefits.
Elasticity (Elastic or Inelastic)	An economic measure to describe how a change in behaviour of an economic agent to a change in, for example, price or income.
Elasticity of Substitution	A measure of ease or difficulty of how an economic agent responds to the loss of a good or service by substituting it for another.
Existence Value	The value an economic agent derives from knowing that a resource exists, even though they have no current or planned use for it.
Externalities	An unintended outcome of an action that affects other economic agents (either positively or negativeyxly) which is neither compensated nor penalised.
Market Failure	When a market either does not function efficiently (i.e. inefficient allocation of resources) and/or a market for a good and/or service does not exist.
Non-Market Valuation	How economists generate estimates of value for goods and/or services that are not valued by conventional markets.
Non-Use Values	Value derived from the knowledge that a good and/or service is maintained even when there is no direct use. Types of non-use values include bequest value, altruistic value and existence value.
Opportunity Cost	Value of a resource in its next best available use.
Option Value	The value of preserving the option to use services in the future either by oneself (option value) or by others or heirs (bequest value).
Quasi-Option Value	The value of avoiding irreversible decisions until new information reveals whether certain ecosystem services have values society is not currently aware of.
Property Rights	The set of characteristics determining how property is owned and managed. The main characteristics of a property rights regime are: Ownership; Excludability; Transferability; and Enforceability. Examples of property right regimes include private, collective, common, public, and state ownership.
Pure (Societal) Time Preference Rate	This is the rate a society values the present compared to the future.
Use Values	The values attributed to non-market goods and/or services from direct use.
Utility	A measure of the satisfaction that is gained from a good or service. It is synonymous with wellbeing.
Willingness to Accept (WTA)	The monetary measure of the value of forgoing an environmental gain or allowing a loss.
Willingness to Pay (WTP)	The monetary measure of the value of obtaining an environmental gain or avoiding a loss.

Open peer review. To view the open peer review materials for this article, please visit http://doi.org/10.1017/ext.2023.18.

Data availability statement. No data was used in the production of this manuscript.

Acknowledgements. We thank two very generous reviewers for providing excellent suggestions on earlier versions of this manuscript. We also thank Steve King for his insightful comments on the manuscript.

Author contribution. I.M.F., D.L.R. and M.B. all contributed equally to the production of this manuscript.

Financial support. No financial support was provided for the production of this manuscript.

Competing interest. D.L.R. is a senior editor of the journal *Cambridge Prisms: Extinction.*

References

Alexander RR (2000) Modelling species extinction: The case for non-consumptive values. *Ecological Economics* **35**(2), 259–269.

Ando AW (2022) Equity and cost-effectiveness in valuation and action planning to preserve biodiversity. *Environmental and Resource Economics* **83**(4), 999–1015. https://doi.org/10.1007/s10640-022-00674-1

Ando AW and Langpap C (2018) The economics of species conservation. *Annual Review of Resource Economics* **10**, 445–467.

Armsworth PR (2018) Time discounting and the decision to protect areas that are near and threatened or remote and cheap to acquire. *Conservation Biology* **32**(5), 1063–1073.

Angulo E and Courchamp F (2009) Rare species are valued big time. *PLoS One* **4**(4), e5215. https://doi.org/10.1371/journal.pone.0005215.

Arrow KJ and Fisher AC (1974) Environmental preservation, uncertainty, and irreversibility. In *Classic Papers in Natural Resource Economics*. London: Palgrave Macmillan, pp. 76–84.

Barbier EB (2022) The policy implications of the Dasgupta review: Land use change and biodiversity: Invited paper for the special issue on “the economics of biodiversity: Building on the Dasgupta Review” in environmental and resource economics. *Environmental and Resource Economics* **83**, 911–935.

Baumol WJ and Oates WE (1988) *The Theory of Environmental Policy*. Cambridge: Cambridge University Press.

Beard SJ, Holt L, Tzachor A, Kemp L, Avin S, Torres P and Belfield H (2021) Assessing climate change’s contribution to global catastrophic risk. *Futures* **127**, 102673.

Beidleman R (2004) More than specimens in natural history museums. *Bioscience* **54**(7), 612–612.

Bertram C and Quaas MF (2017) Biodiversity and optimal multi-species ecosystem management. *Environmental and Resource Economics* **67**(2), 321–350.

BirdLife International (2022) *State of the World’s Birds 2022: Insights and Solutions for the Biodiversity Crisis*. Cambridge, UK: BirdLife International.

Brander JA and Taylor MS (1998) The simple economics of Easter Island: A Ricardo-Malthus model of renewable resource use. *American Economic Review* **88**(1), 119–138.

Bristol RM, Fraser I, Groombridge JJ and Veríssimo D (2014) An economic analysis of species conservation and translocation for island communities: The Seychelles paradise flycatchers as a case study. *Journal of Environmental Economics and Policy* **3**(3), 237–252.

Brock M, **Perino G and Sugden R** (2017) The Warden attitude: An investigation of the value of interaction with everyday wildlife. *Environmental and Resource Economics* **67**(1), 127–155.

Brock M, **Fraser I**, **Law C**, **Mitchell S and Roberts DL** (2021) An economic analysis of twitching behaviour and species rarity. *Journal of Environmental Economics and Policy* **10**(1), 54–73.

Brouwer R, **Pinto R**, **Dugstad A and Navrud S** (2022) The economic value of the Brazilian Amazon rainforest ecosystem services: A meta-analysis of the Brazilian literature. *PLoS One* **17**(5), e0268425. https://doi.org/10.1371/journal.pone.0268425.

Bulte EH and van Kooten GC (1999) Economics of antipoaching enforcement and the ivory trade ban. *American Journal of Agricultural Economics* **81**, 453–466.

Bulte EH, **Horan RD and Shogren JF** (2003) Is the Tasmanian tiger extinct? A biological–economic re-evaluation. *Ecological Economics* **45**(2), 271–279.

Bulte E, **Horan RD and Shogren JF** (2006) Megafauna extinction: A paleoeconomic theory of human overkill in the Pleistocene. *Journal of Economic Behavior and Organization* **59**(3), 297–323.

Byl JP (2019) Perverse incentives and safe harbors in the endangered species act: Evidence from timber harvests near woodpeckers. *Ecological Economics* **157**, 100–108.

Cardoso P, **Barton PS**, **Birkhofer K**, **Chichorro F**, **Deacon C**, **Fartmann T**, **Fukushima CS**, **Gaigher R**, **Habel JC**, **Hallmann CA and Hill MJ** (2020) Scientists' warning to humanity on insect extinctions. *Biological Conservation* **242**, 108426.

Chadès I, **McDonald-Madden E**, **McCarthy MA**, **Wintle B**, **Linkie M and Possingham HP** (2008) When to stop managing or surveying cryptic threatened species. *Proceedings of the National Academy of Sciences* **105**(37), 13936–13940.

Christie M, **Hanley N**, **Warren J**, **Murphy K**, **Wright R and Hyde T** (2006) Valuing diversity of biodiversity. *Ecological Economics* **58**, 304–317.

Clark CW (1973) Profit maximization and the extinction of animal species. *Journal of Political Economy* **81**(4), 950–961.

Clucas B, **Rabotyagov S and Marzluff JM** (2015) How much is that birdie in my backyard? A cross-continental economic valuation of native urban songbirds. *Urban Ecosystems* **18**, 251–266. https://doi.org/10.1007/s11252-014-0392-x.

Cohen F, **Hepburn CJ and Teytelboym A** (2019) Is natural capital really substitutable? Annual Review of Environment and Resources *44*(1), 425–448.

Conrad JM (2018) Real options for endangered species. *Ecological Economics* **144**, 59–64.

Conrad JM and Lopes AA (2017) Poaching and the dynamics of a protected species. *Resource and Energy Economics* **48**, 55–67.

Cook D, **Malinauskaite L**, **Davíðsdóttir B**, **Ögmundardóttir H and Roman J** (2020) Reflections on the ecosystem services of whales and valuing their contribution to human well-being. *Ocean & Coastal Management* **186**, 105100.

Courchamp F, **Angulo E**, **Rivalan P**, **Hall RJ**, **Signoret L**, **Bull L and Meinard Y** (2006) Rarity value and species extinction: The anthropogenic Allee effect. *PLoS Biology* **4**(12), e415.

Dasgupta P (2021) *The Economics of Biodiversity: The Dasgupta Review*. London: HM Treasury.

Do QT, **Levchenko AA**, **Ma L**, **Blanc J**, **Dublin H and Milliken T** (2021) The price elasticity of African elephant poaching. *The World Bank Economic Review* **35**(3), 545–562.

Dutcher D, **Finley J**, **Luloff A and Johnson J** (2007) Connectivity with nature as a measure of environmental values. *Environment and Behaviour* **39**(4), 474–493.

Field SA, **Tyre AJ**, **Jonzen N**, **Rhodes JR and Possingham HP** (2004) Minimizing the cost of environmental management decisions by optimizing statistical thresholds. *Ecology Letters* **7**(8), 669–675.

Fleming CM and Alexander RR (2003) Single-species versus multiple-species models: The economic implications. *Ecological Modelling* **170**(2–3), 203–211.

Gordon ER, **Butt N**, **Rosner-Katz H**, **Binley AD and Bennett JR** (2020) Relative costs of conserving threatened species across taxonomic groups. *Conservation Biology* **34**(1), 276–281.

Gordon HS (1954) The economic theory of a common-property resource: The fishery. *Journal of Political Economy* **62**(2), 124–142.

Goulding MJ, **Roper TJ**, **Smith GC and Baker SJ** (2003) Presence of free-living wild boar *Sus scrofa* in southern England. *Wildlife Biology* **9**, 15–20.

Groom B and Turk Z (2021) Reflections on the Dasgupta review on the economics of biodiversity. *Environmental and Resource Economics* **79**(1), 1–23.

Hanley N, **Shogren J and White B** (2019) *Introduction to Environmental Economics*. New York: Oxford University Press.

Hanley N and Perrings C (2019) The economic value of biodiversity. *Annual Review of Resource Economics* **11**, 355–375.

Horan RD and Melstrom RT (2011) No sympathy for the devil. *Journal of Environmental Economics and Management* **62**(3), 367–385.

IPBES (2019) Global assessment report on biodiversity and ecosystem services of the intergovernmental science-policy platform on biodiversity and ecosystem services. In Brondizio ES, Settele J, Díaz S and Ngo HT (eds), *IPBES Secretariat*, Vol. **1148**. Bonn, Germany: IPBES secretariat. https://doi.org/10.5281/zenodo.3831673#.

IPBES (2022) Summary for policymakers of the methodological assessment of the diverse values and valuation of nature of the intergovernmental science-policy platform on biodiversity and ecosystem services. In Pascual U, Balvanera P, Christie M, Baptiste B, Gonzalez-Jimenez D, Anderson C, Athayde S, Barton D, Chaplin-Kramer R, Jacobs S, Kelemen E, Kumar R, Lazos E, Martin A, Mwampamba T, Nakangu B, O'Farrell P, Raymond C, Subramanian S and Vatn A (eds), *IPBES Secretariat*. Bonn, Germany. https://doi.org/10.5281/zenodo.6522392.

IUCN (2012) *IUCN Red List Categories and Criteria: Version 3.1*, 2nd Edn. Gland, Switzerland and Cambridge, UK: IUCN, iv + 32 pp.

IUCN/SSC (2013) *Guidelines for Reintroductions and Other Conservation Translocations, Version 1.0*. Gland, Switzerland: IUCN Species Survival Commission, viiii + 57 pp.

Jacobsen JB, **Boiesen JH**, **Thorsen BJ and Strange N** (2008) What's in a name? The use of quantitative measures versus 'iconised' species when valuing biodiversity. *Environmental Resource Economics* **39**, 247–263.

King S, **Ferrier S**, **Turner K and Badura T** (2019) Discussion paper 11: Research paper on habitat and biodiversity related ecosystem services. Paper Submitted to the Expert Meeting on Advancing the Measurement of Ecosystem Services for Ecosystem Accounting. New York, 22–24 January 2019 and subsequently revised. Version of 15 March 2019. Available at https://seea.un.org/events/expert-meeting-advancing-measurement-ecosystem-services-ecosystem-accounting (accessed 28 June 2023).

Kraak SB (2011) Exploring the 'public goods game model to overcome the tragedy of the commons in fisheries management. *Fish and Fisheries* **12**(1), 18–33.

Kremer M and Morcom C (2000) Elephants. *American Economic Review* **90**(1), 212–234.

Krishna VV, **Darras K**, **Grass I**, **Mulyani YA**, **Prawiradilaga DM**, **Tscharntke T and Qaim M** (2019) Wildlife trade and consumer preference for species rarity: An examination of caged-bird markets in Sumatra. *Environment and Development Economics* **24**(4), 339–360.

Lewis DJ, **Dundas SJ**, **Kling DM**, **Lew DK and Hacker SD** (2019) The non-market benefits of early and partial gains in managing threatened salmon. *PLoS One* **14**(8), e0220260.

Lewis DJ, **Kling DM**, **Dundas SJ and Lew DK** (2022) Estimating the value of threatened species abundance dynamics. *Journal of Environmental Economics and Management* **113**, 102639.

Lopes AA and Atallah SS (2020) Worshipping the tiger: Modeling non-use existence values of wildlife spiritual services. *Environmental and Resource Economics* **76**(1), 69–90.

Lyons JA and Natusch DJD (2013) Effects of consumer preferences for rarity on the harvest of wild populations within a species. *Ecological Economics* **93**, 278–283.

Mason CF, **Bulte EH and Horan RD** (2012) Banking on extinction: Endangered species and speculation. *Oxford Review of Economic Policy* **28**(1), 180–192.

MacFarlane D, **Hurlstone MJ**, **Ecker UK**, **Ferraro PJ**, **van Der Linden S**, **Wan AK**, **Veríssimo D**, **Burgess G**, **Chen F**, **Hall W and Hollands GJ** (2022) Reducing demand for overexploited wildlife products: Lessons from systematic reviews from outside conservation science. *Conservation Science and Practice* **4**(3), e627.

Melstrom RT, **Shanafelt DW and Reeling CJ** (2022) Coordinating investments in habitat management and economic development. *Journal of Bioeconomics* **24**(1), 67–91.

Morse-Jones S, **Bateman I**, **Kontoleon A**, **Ferrini S**, **Bergess N and Kerry-Turner R** (2012) Stated preferences for tropical wildlife conservation amongst distant beneficiaries; charisma, endemism, scope and substitution effects. *Ecological Economics* **78**, 9–18.

Naidoo RL, **Weaver C**, **Diggle RW**, **Matongo G**, **Stuart-Hill G and Thouless C** (2016) Complementary benefits of tourism and hunting to communal conservancies in Namibia. *Conservation Biology* **30**(3), 628–638.

Ng YK (2019) KEYNOTE: Global extinction and animal welfare: Two priorities for effective altruism. *Global Policy* **10**(2), 258–266.

OECD (2018) *Cost-Benefit Analysis and the Environment: Further Developments and Policy Use*. Paris: OECD Publishing. https://doi.org/10.1787/9789264085169-en.

Ormerod P (2007) *Why Most Things Fail: Evolution, Extinction and Economics*. Hoboken, NJ: John Wiley & Sons.

O'Rourke E (2014) The reintroduction of the white-tailed sea eagle to Ireland: People and wildlife. *Land Use Policy* **38**, 129–137.

Peach DAH (2019) The bizarre and ecologically important hidden lives of mosquitoes. The Conversation, December. Available at https://theconversation.com/the-bizarre-and-ecologically-important-hidden-lives-of-mosquitoes-127599 (accessed 28 June 2023).

Pimm SL, **Jenkins CN**, **Abell R**, **Brooks TM**, **Gittleman JL**, **Joppa LN**, **Raven PH**, **Roberts CM and Sexton JO** (2014) The biodiversity of species and their rates of extinction, distribution, and protection. *Science* **344**(6187), 1246752.

Pindyck RS (2007) Uncertainty in environmental economics. *Review of Environmental Economics and Policy* **1**(1), 45–65.

Polasky S and Dampha NK (2021) Discounting and global environmental change. *Annual Review of Environment and Resources* **46**(1), 691–717.

Pathirana MT, **Herath HMRP**, **Scott N and Gardiner S** (2021) Economic sustainability in safari tourism in Southeast Asia: The case of Minneriya National Park Sri Lanka. *Journal of Tourism, Hospitality and Culinary Arts* **13**(3), 201–227.

Rausser GC and Small AA (2000) Valuing research leads: Bioprospecting and the conservation of genetic resources. *Journal of Political Economy* **108**(1), 173–206.

Richardson L and Loomis J (2009) The total economic value of threatened, endangered and rare species: An updated meta-analysis. *Ecological Economics* **68**(5), 1535–1548.

Rising JA, **Taylor C**, **Ives MC and Ward RE** (2022) Challenges and innovations in the economic evaluation of the risks of climate change. *Ecological Economics* **197**, 107437.

Schulte to Bühne H, **Ross B**, **Sandom CJ and Pettorelli N** (2022) Monitoring rewilding from space: The Knepp estate as a case study. *Journal of Environmental Management* **312**, 114867.

Shibayama K and Fraser IM (2014) Nonhomothetic growth models for the environmental Kuznets curve. *International Economic Review* **55**(3), 919–942.

Sol J (2019) Economics in the anthropocene: Species extinction or steady state economics. *Ecological Economics* **165**, 106392.

Stern N (2007) The economics of climate change. In *The Stern Review*. Cambridge: Cambridge University Press.

Stern N (2022) A time for action on climate change and a time for change in economics. *The Economic Journal* **132**(644), 1259–1289.

Sterner T and Persson UM (2008) An even sterner review: Introducing relative prices into the discounting debate. *Review of Environmental Economics and Policy* **2**(1), 61–76.

Sutton AE (2015) Leadership and management influences the outcome of wildlife reintroduction programs: Findings from the Sea Eagle Recovery Project. *PeerJ* **3**, e1012.

Swanson TM (1994) The economics of extinction revisited and revised: A generalised framework for the analysis of the problems of endangered species and biodiversity losses. *Oxford Economic Papers* **46**, 800–821.

Thomas C, **Cameron A**, **Green R**, **Bakkenes M**, **Beaumont LJ**, **Collingham YC**, **Erasmus BF**, **De Siqueira MF**, **Grainger A**, **Hannah L and Hughes L** (2004) Extinction risk from climate change. *Nature* **427**, 145–148.

Tilman D (2022) Extinction, climate change and the ecology of Homo sapiens. *Journal of Ecology* **110**(4), 744–750.

Turnhout E and Purvis A (2020) Biodiversity and species extinction: Categorisation, calculation, and communication. *Griffith Law Review* **29**(4), 669–685.

Urban MC (2015) Accelerating extinction risk from climate change. *Science* **348**(6234), 571–573.

Veríssimo D, **Fraser I**, **Groombridge J**, **Bristol R and MacMillan DC** (2009) Birds as tourism flagship species: A case study of tropical islands. *Animal Conservation* **12**(6), 549–558.

Vincent JR, **Carson RT**, **DeShazo JR**, **Schwabe KA**, **Ahmad I**, **Chong SK**, **Chang YT and Potts MD** (2014) Tropical countries may be willing to pay more to protect their forests. *Proceedings of the National Academy of Sciences* **111**(28), 10113–10118.

Waterhouse J and Mitchell C (2022) Has anybody seen a Tasmanian tiger lately? Ethical and ontological considerations of Thylacine de-extinction. *Green Letters* **26**(1), 14–27.

Weitzman ML (2009) On modeling and interpreting the economics of catastrophic climate change. *The Review of Economics and Statistics* **91**(1), 1–19.

White PCL, **Gregory KW**, **Lindley PJ and Richards G** (1997) Economic values of threatened mammals in Britain: A case study of the otter *Lutra lutra* and the water vole *Arvoicola terrestris*. *Biological Conservation* **82**, 345–354.

Winker K (2004) Natural history museums in a post biodiversity era. *Bioscience* **54**(5), 455–459.

Wolf CM, **Griffith B**, **Reed C and Temple SA** (1996) Avian and mammalian translocations: Update and reanalysis of 1987 survey data. *Conservation Biology* **10**(4), 1142–1154.

Xu J (2021) A comment on:"Real options for endangered species". *Ecological Economics* **189**, 107175.

Zander KK, **Burton M**, **Pandit R**, **Gunawardena A**, **Pannell D and Garnett ST** (2022) How public values for threatened species are affected by conservation strategies. *Journal of Environmental Management* **319**, 115659.