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The Economic Value of  
Biodiversity

Nick Hanley<sup>1</sup> and Charles Perrings<sup>2</sup>

<sup>1</sup>Institute of Biodiversity, Animal Health and Comparative Medicine, University of Glasgow, Glasgow G12 8QQ, United Kingdom; email: Nicholas.Hanley@glasgow.ac.uk

<sup>2</sup>School of Life Sciences, Arizona State University, Tempe, Arizona 85287, USA

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### Abstract

Biodiversity is declining worldwide, and the costs of biodiversity losses are increasingly being recognized by economists. In this article, we first review the multiple meanings of biodiversity, moving from species richness and simple abundance-weighted species counts to more complex measures that take account of taxonomic distance and functionality. We then explain the ways in which protecting biodiversity generates economic benefits in terms of direct and indirect values. Empirical approaches to estimating direct and indirect values are presented, along with a selection of recent evidence on how substantial these values are. The use of asset accounting approaches to track biodiversity values over time is discussed, in the context of sustainable development paths. Finally, we review some important challenges in valuing biodiversity that remain to be solved.

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## 1. DEFINING BIODIVERSITY

Biodiversity refers to the variety of life forms at all levels: molecular, organismic, population, species, and ecosystem. The biodiversity of some region is accordingly a measure of all the genes, species, and ecosystems of that region. It spans the morphological diversity of individuals and populations within a species, the taxonomic diversity of species within a community or ecosystem, the functional diversity of groups of species within an ecosystem, and the diversity of ecosystems themselves. From a human well-being perspective, biodiversity includes the variety of plants, animals, and other life forms (bacteria, archaea, Protozoa, Chromista, and Fungi) used directly or indirectly: (a) in the production of foods, fuels, fibers, pharmaceuticals, and industrial goods; (b) in the supply of services such as pollination, water quality regulation, soil erosion control, storm buffering, and climate regulation; or (c) that have aesthetic, scientific, totemic, or amenity value (Millenn. Ecosyst. Assess. 2005).

Biodiversity includes the wild species that are the primary focus of conservation biology. But it also includes the genetic diversity of cultivated crops and wild crop relatives, the diversity of pathogens that induce diseases of humans, animals and plants, and the diversity of antibiotics used to counter those pathogens. By the same token it includes the variety of life forms involved in ecosystem functions and processes—decomposition, production, nutrient cycling, and energy fluxes—that underpin crop production, or the abundance of pathogens and their controllers.

The traditional approach to the measurement of biodiversity focuses on four dimensions of diversity—referred to as alpha, beta, gamma, and omega diversity—at the species level. Alpha diversity is the taxonomic diversity of species in a particular system and is measured by an index of the number of species and their abundance. The two most commonly used indices of alpha diversity are the Shannon and Simpson's indices. The Shannon index,  $H = -\sum_{i=1}^S p_i \ln p_i$ , defines  $p_i$  to be the proportion of the number of individuals of the  $i$ th species to the total number of individuals,  $S$ . The index is higher the greater the species richness in a location and the more even the relative abundance of species. Simpson's index,  $D = \sum_{i=1}^S p_i^2$ , defines  $p_i$  to be the fraction of all organisms accounted for by the  $i$ th species and takes a value between 0 (zero diversity) and 1 (infinite diversity). Beta diversity measures the difference in species diversity between systems—the number of taxa that is unique to each system being compared. For two systems, the Sørensen index of beta diversity (Sørensen 1948),  $\beta = \frac{2c}{S_1+S_2}$ , relates the number of species common to both systems,  $c$ , to the total number of species in those systems. It also takes a value of 0 when there is no species overlap between the communities, and a value of 1 when exactly the same species are found in both communities. Gamma diversity is a measure of taxonomic diversity across all systems being evaluated. For two systems it is  $\gamma = S_1 + S_2 - c$ , a count of the number of distinct species in both systems (Whittaker 1972). Finally, omega diversity is a measure of the taxonomic difference between species or phylogenetic diversity (Schweiger et al. 2008). The most common measures use either a minimum spanning path approach or a pairwise distance approach. The minimum spanning path approach sums the branch lengths of the phylogenetic tree that contains all of the species in the area or the number of nodes separating species. One example,  $PD_{\text{NODE}} = \sum_i n_i$ , sums the number of nodes within the minimum spanning path (Faith 1992). The pairwise distance approach measures the taxonomic distinctiveness or distance of a species to its nearest neighbor. One example of this is the pure diversity index,  $D_D = \sum_i d_{i\min}$ , from Solow et al. (1993) and Weitzman (1992), where  $d_{i\min}$  is the nearest neighbor distance of species  $i$  to all other species.

Each of these indices explicitly or implicitly weights species in some way. Gamma diversity, for example, implicitly weights all species equally. Alpha and beta diversity, on the other hand, explicitly weight each species by its relative abundance, while omega diversity explicitly weights species by their phylogenetic distance from other species. Which diversity index and which

weighting system is appropriate depend on the problem at hand. Weitzman's Noah's Ark Problem, for example, focuses on the cost effectiveness of projects that are ranked in terms of their contribution to the maintenance of phylogenetic or omega diversity (Weitzman 1998). Other approaches to project ranking have similarly focused on phylogenetic distance (Hartmann 2013, Polasky & Solow 1995). By contrast, Brock & Xepapadeas's unified genetic approach (2003) focuses on the ecosystem services associated with differences in species richness and abundance. Given two systems differing in alpha diversity, the value of the additional species richness or abundance in the more biodiverse system is measured by the difference in the Bellman state valuation functions for the optimal management of the two systems (Brock & Xepapadeas 2003).

A focus on the utility of goods and services associated with different combinations of species suggests that alternative measures may be preferred. Ecosystem functioning depends more on the traits of the plants, animals, and microorganisms performing particular functions than on their taxonomy (Bracken et al. 2008, Bunker et al. 2005, McIntyre et al. 2007, Solan et al. 2004). The biodiversity supporting freshwater provision, for example, differs from the biodiversity needed for timber supply, but in both cases is determined by traits such as root structure, above ground biomass, wood production and composition, crown architecture, and leaf display. This has led to the development of alternative measures of diversity within functional groups (Mouchet et al. 2010). Following the measure of functional attribute diversity (Walker et al. 1999), a number of distinct indices have been developed that address either species richness within functional groups (Schmera et al. 2009) or assembly rules based on trait values (Villéger et al. 2008).

The weights adopted in the traditional alpha-omega measures are likely to differ from the weights that resource users would use, which depend on the same factors that fix the value of other assets: preferences, technology, endowments, and property rights. In this review, we consider what is currently known about the economic value of biodiversity, given the properties of the ecosystems concerned and the many ways in which biodiversity contributes to human well-being. What makes this value difficult to quantify is that (*a*) most biodiversity is not directly marketed or subject to well-defined property rights, (*b*) there is still considerable uncertainty about the role of biodiversity in the production of marketed goods and services, and (*c*) many of the services generated by functional groups of species are public goods. While there are well-defined markets for particular foods, fuels, fibers, and pharmaceuticals, and so for the species involved directly in their production, there are no well-defined markets for the supporting ecosystem processes and functions, or for species affecting production as predators or prey, competitors, pests or pathogens, mutualists, or symbionts. Although the abundance of such species might affect land values or the market value of specific use rights (such as hunting, wild capture, or access rights), the species themselves are unpriced. The net result is that decisions made by private landowners that have an impact on biodiversity frequently take no account of impacts beyond those that directly affect the private costs and benefits of resource users. Indeed, biodiversity may be subject to more externalities than any other area of environmental concern.

Successive assessments of the state of biodiversity (Heywood & Watson 1995; IPBES 2018a,b,c; Millenn. Ecosyst. Assess. 2005; Secr. Conv. Biol. Divers. 2010, 2014) show declining biodiversity globally as measured by all of the standard ecological indices. Although response indicators such as the area of land committed to conservation may be improving, almost all areas continue to experience declining species richness. The IUCN Red List, which assesses the number of species that are currently threatened (defined as critically endangered, endangered, or vulnerable), has consistently reported increasing levels of threat. To control for the effect of improvements in information, a Red List Index has been developed for four taxonomic groups: birds, mammals, amphibians, and corals. For each of these groups the index has been declining in the period for which data are available—precipitously in the case of corals (IUCN 2018).

There are many drivers of the loss of biodiversity, but all are associated with the fact that the values people place on species favor the modification and simplification of ecosystems. Land is managed so as to increase the abundance of beneficial species (crops, livestock, timber trees, farmed or wild-caught marine species, ornamental species, commensals) and decrease the abundance of harmful species (pests, weeds, pathogens, predators, competitors). Continuing growth in the area of land committed to agriculture, aquaculture, forestry, mining, transport, industry, commerce, or housing means a reduction in wildlands and thus continuing losses of habitat for wild species. The biodiversity problem addressed by economists is not that land conversion involves some loss of biodiversity. It is that private resource users take insufficient account of the wider and longer-term consequences of actions that lead to the loss of biodiversity, and in so doing may be imposing significant costs on current and future generations.

Economists are concerned about the effects of biodiversity loss on the supply of private and public ecosystem services. Quantifying the size of these effects requires us to be able to estimate economic values for biodiversity change. The next section sets out a conceptual framework for measuring such values. Subsequent sections explain methods for estimating these values, and provide examples.

## **2. THE CONCEPTUAL LINKS BETWEEN BIODIVERSITY AND ECONOMIC VALUE**

As Jacobsen et al. (2008, p. 247) state, “...valuing biodiversity is a key challenge to environmental economists.” The economic value of biodiversity can be characterized as either direct or indirect in terms of the way in which changes in biodiversity affect human well-being (Bartkowski 2017).

### **2.1. Direct Economic Values of Biodiversity**

Biodiversity generates direct economic value when people care about it. This means that some measure of biodiversity enters into at least some person's utility function. So, an individual is happier when the number of bird species in his local forest is greater or when the abundance of individual species populations is higher. It is also possible that individuals care about the contribution of biodiversity to ecosystem functioning (Czajkowski et al. 2009).

However, more likely is that individuals care about the individual species that make up the biodiversity assemblage of a specific ecosystem. This could be in terms of the specifics/identities of individual species (Morse-Jones et al. 2012, Richardson & Loomis 2009); their conservation status [e.g., scarce versus abundant, or a declining or increasing population trend (Lundhede et al. 2014)] or how well they are known (Christie et al. 2006). Evidence also indicates that people's willingness to pay (WTP) for conserving a particular species depends on its cuteness or charisma (Di Minin et al. 2013) and its uniqueness or endemism (Morse-Jones et al. 2012). Individuals also care about the habitats in which species are found, for example, having preferences for specific types of forest relative to other forest types. As more species are conserved, it may well be that the marginal value of protecting additional species declines (Jacobsen et al. 2008).

The direct values of biodiversity, or of the presence/absence of a given species or the population abundance of a given species, can be defined in terms of both use and nonuse values. Direct use values for biodiversity occur when an individual engages in recreational hunting of a particular species (e.g., red deer in Scotland) or derives utility from watching a species in situ (e.g., the utility to a bird watcher of being able to see five rare species of birds on a wetlands visit, rather than one rare species). But nonuse, direct values are also likely to be present and may be relevant for

a bigger group of individuals. People care about the existence of individual species such as snow leopards or killer whales even if they will never see one in the wild, and they may derive utility from knowing more wetland bird species are conserved in Mallorca even if they never visit the island.

A large volume of work has focused on the direct benefit of biodiversity conservation in terms of WTP for specific habitats or species, rather than on the characteristics of biodiversity itself. Examples of these kinds of studies are provided in Section 3. The study by Christie et al. (2006) was one of the first attempts to try to measure the direct economic value of the characteristics of biodiversity, rather than the direct economic value of individual species or habitats. They showed that members of the UK general public had a WTP that depended on (a) whether rare or common species were protected, (b) whether these species were well-known or unfamiliar to most people, (c) whether the policy would merely slow down current rates of loss rather than stopping or reversing this trend, and (d) the consequences of the policy in terms of habitat restoration versus habitat creation and for the ecosystem services supplied. Czajkowski et al. (2009) showed that Polish people were willing to pay for biodiversity policy that enhanced and protected natural ecosystem processes, protected rare species, or enhanced ecosystem components such as dead wood, natural ponds, and clearings within the Białowieża Forest (Poland). It is also possible that people care about how biodiversity conservation objectives are achieved (the policy choice), irrespective of the outcome. For instance, as shown by Hanley et al. (2003), peoples' WTP for wild geese conservation in Scotland varied according to whether shooting was used as part of a management policy, whereas Czajkowski et al. (2009) show that people's preferences for protecting forest biodiversity were stronger when this was achieved by extending the national parks system, rather than by some other method.

A rather different route by which biodiversity generates utility and thus direct value is in the context of provisioning services when consumers have a preference for species diversity. Quaas & Requate (2013) look at the problem of multispecies fishery management when consumers prefer more diversity in their fish consumption (for a given total consumed, utility is higher when more different species make up this total consumption bundle). Modeling this love for diversity as equivalent to a low elasticity of substitution in consumption between fish species, they show that this can drive species to be more liable to collapse, as the low elasticity of substitution in consumption reduces the effect of rising prices (as a species gets scarcer) on demand. This means one can get a cascade of collapsing fish species if the consumers' love for diversity is strong enough—a worrying result.

## 2.2. The Indirect Economic Value of Biodiversity

Biodiversity is an input to aspects of ecosystem functioning and thus to the supply of ecosystem services (such as pollination), which in turn provide benefits to people (such as outputs of insect-pollinated crops). These benefits may be either valued by markets (e.g., agricultural crops) or not (wild flowers). The role of crop species and genetic diversity in reducing risks to commercial outputs from farms and forests is included in this indirect value (see Di Falco & Perrings 2005 for farms and MacPherson et al. 2017 for forests), as are any negative effects of increases in some species such as invasive pests and pathogens on crop outputs. Changes to biodiversity result in changes to economic values as moderated by the processes linking biodiversity as an input to a valuable economic output.

Key to indirect values is therefore the precise nature of the links between biodiversity, ecosystem functioning and the supply of ecosystem services within a given system and spatial/temporal

context, because it is these ecosystem services that provide the connection to human well-being and thus to economic value (Naeem et al. 2016). The contribution, if any, of higher biodiversity to greater ecosystem resilience would also be classified as an indirect value, as the economic value of resilience rests on the ability of a system to maintain its functioning in the presence of shocks (Walker et al. 2010). Baumgärtner & Strunz (2014) point out that one can think of higher resilience as having two, separable potential benefits. One of these is an insurance value: higher resilience affects the uncertainty of income flows from the ecosystem and thus affects the income risk to people who derive benefits from the ecosystem service. This insurance value depends both on how higher resilience impacts the probability that a system will move from one more desirable domain to another less desirable domain given some external shock and on peoples' risk preferences. It is equivalent to the change in a representative agent's risk premium for a marginal change in the level of resilience. An additional, separable economic value of resilience relates to the effects of a change in ecosystem dynamics on the expected income from the supply of ecosystem services across alternative states of the world.

The form of the relationships between biodiversity and ecosystem function and between ecosystem function and the supply of ecosystem service is complex and ecosystem specific. Moreover, Mace et al. (2012) have argued that it is difficult to fit biodiversity into the now-dominant ecosystem services concept because it plays three roles: as a determinant of ecosystem functioning, as an ecosystem service (e.g., pollination), and as a final outcome that people care directly about. The relationship between biodiversity and one aspect of economic value, for example, agricultural output may well be very different to the relationship of the same biodiversity indicator to another kind of economic value, such as carbon storage. While some functional relationships will imply that trade-offs are the norm, so that increasing biodiversity typically comes at the expense of lower incomes from farming (e.g., Armsworth et al. 2012 for upland farming in the United Kingdom), in other cases rising biodiversity is consistent with increasing economic values such as nonuse and other cultural ecosystem values.

Paul et al. (2019) develop a synthesis of alternative possible functional forms for the biodiversity–economic value (BEV) relationship. These possibilities are

- Positive concave
- Positive convex
- Negative concave
- Negative convex
- Strictly concave
- V-shaped

Paul et al. argue that which of these relationships best describes a given system depends on whether single or multiple ecosystem services are being included in economic value and whether there are complementarities or trade-offs present in the supply of multiple ecosystem services. Increasing biodiversity can be associated with falling agricultural and forest outputs (falling economic values) when the biodiversity increase is due to invasive species, such as invasive pests and diseases. The cumulative incidence of such pests and diseases is rising over time in many countries, due to expanded world trade links, crop specialization, the adoption of non-native varieties, and climate change (Freer-Smith & Webber 2017). Such increases in pest and disease incidence reduce crop yields and/or cause valuable resources to be devoted to managing these risks. This can change the optimal management of forests, in terms of the optimal rotation period (MacPherson et al. 2018) and the optimal planting mix (MacPherson et al. 2017). Invasive species can also produce losses in direct biodiversity values, if for example they adversely affect the quality or availability of

recreational resources (waterways, forests) or result in the loss of valued native species of wildlife (Chakir et al. 2016).

### 2.3. Biodiversity, Asset Values, and Sustainability

Because the value of any asset lies in the discounted stream of services it offers over time, the valuation of ecosystem services conveys information on the value of underlying assets and the efficiency with which they are used. Stocks of species, along with water, soil, and mineral resources, are among the environmental assets that generate ecosystem services. In terms of intertemporal efficiency, the decision as to whether environmental assets should be conserved depends on the expected rate of change in their value relative to other assets, according to the Hotelling principle (Hotelling 1931). Conservation of an asset in any given state will be optimal if the proportional growth in its expected value in that state is more than the rate of return it yields if it is converted to an alternative state. Environmental assets are frequently poorly understood components of wealth, largely because many of the services they offer lie outside the market and are public goods of one kind or another (Perrings 2014).

The identification of environmental assets requires that they be connected to the structure of property rights. Barbier & Markandya (2013) have argued that the appropriate assets are ecosystems. Ecosystems generate flows of benefits over time, the discounted value of which defines the capital value of the ecosystem. Biodiversity is one characteristic of an ecosystem that determines the value of the flow of benefits over time. While it is true that the source of valuable ecosystem services is the underlying ecosystem, it is not clear that the system itself is the appropriate asset to focus on. In some cases, whole ecosystems are contained within a single land holding, but most ecosystems span the boundaries between private and public landholdings and between jurisdictions. Indeed, many ecosystems span national jurisdictions and areas beyond national jurisdiction. They are subject to competing claims, and competing uses. The convention has therefore been to take the underlying assets to be the rights (private, common, or communal titles) to land on which ecosystems occur. If there are ecosystem processes or functions that connect different titles—upstream-downstream flows of water, nutrients, sediment, and movements of animals or birds and so on—this is likely to affect the value of those titles. But private decisions about land conversion or conservation are taken at the level of the title.

The primary goal of a sustainable development strategy is to maintain or increase the comprehensive wealth of a country, of which natural capital is a partly substitutable component (Arrow et al. 2012). Natural capital is the stock of all natural assets that yield a flow of benefits to society. It comprises both biotic and abiotic assets—forests, grasslands, wetlands, soils, minerals, and water, among others. Like other forms of capital, natural capital can depreciate or appreciate over time. Because natural capital is at least partially substitutable by other forms of capital, aggregate wealth can be nondeclining even if natural capital is being depleted. A necessary condition for this is that rents earned on the depletion of natural resources can be reinvested in other, reproducible assets (Hartwick 1977, 1978; Solow 1974, 1986).

The implications for biodiversity and other ecosystem components follow directly. By the Hotelling principle, if the discounted stream of services generated by land in some state is expected to increase in value at a rate at least equal to the rate of return to be had if the land is converted, it will be optimal to conserve it in that state. If not, it will be optimal to convert it, investing the rents in alternative assets. Conversion of land to agriculture, (production) forestry, industry, infrastructure, or residential uses implies that the value of the discounted stream of services expected in those states dominates the value of services in the unconverted state. But conversion may not involve the instantaneous transformation of the system. By the Hartwick rule, progressive mining

of the biotic elements of an ecosystem may be sustainable as long as the rents generated in the process are invested in reproducible assets. Declining biodiversity may be consistent with overall sustainability in this sense. The danger at present is that land use change in many countries is sustainable in neither sense. The discounted value of the stream of services to be had from the unconverted system is systematically underestimated, and the net proceeds from depleting the biotic and abiotic components of ecosystems are not being invested in reproducible assets. How this process can be tracked empirically is outlined in Section 5.

### 3. ESTIMATING THE NONMARKET VALUE OF WILD SPECIES

#### 3.1. Methods

Economists often distinguish three approaches to estimating values for nonmarket goods such as the conservation of wild plants and animals. Two of these—stated preference methods and revealed preference methods—are designed to uncover peoples' WTP for gains in/willingness to accept compensation for the loss of goods and services for which there are no markets. A third class of approach, production function methods, are used to infer the implied WTP for environmental inputs to the production of particular goods or services (valuing the environment as an input). All three can be used to measure different aspects of the value of conserving wild species, of protecting biodiversity "in the wild."

Stated preference methods such as contingent valuation and choice experiments have been used to estimate the value of habitat and species conservation programs since the 1970s. Indeed, some of the earliest applications of the method were concerned with these two issues (see Hanley & Spash 1993). Contingent valuation can be used to estimate WTP for preventing a decline or avoiding damage to or achieving an increase in a specific wildlife species [e.g., a species conservation program for marine mammals (Samples et al. 1986)] or for habitat protection (e.g., Carson et al. 1994, Hanley & Craig 1991). Choice experiments were first used in environmental economics in the early 1990s and are another stated preference method that can be used to study direct biodiversity values. For example, Hanley et al. (2003) compare the preferences of Scottish people for geese conservation policies according to whether or not a species is endangered, where it is found across Scotland, how it is managed, and its population level. This ability to specify such possible determinants of direct biodiversity values as attributes within the survey design is one of the main attractions of using choice experiments in this context.

Revealed preference methods of environmental valuation include recreational demand (travel cost) models and hedonic pricing. Travel cost methods are frequently applied to outdoor recreational use of natural resources, and some measure of biodiversity can be included as a potentially relevant predictor of site choice and consumers' surplus per visit. A fall or rise in this biodiversity indicator can be used to simulate a change in predicted trips and, thus, a change in direct economic values (Johnstone & Markandya 2006). Hedonic pricing is rarely applied to estimating direct values of biodiversity, although many studies relate house prices to forest cover and proximity to farmland and urban green spaces, variables that may all be related to measurable variations in biodiversity.

Where the value of some good or service is priced by the market, the implied value of the functional groups of species used in its supply can be obtained by specifying and estimating the appropriate production function (Allen & Loomis 2006). Similarly, once the production function is known, it can be used to project the future consequences of changes in functional groups of species. For example, consider a change in the species diversity and/or population abundance of wild insect pollinators such as bumblebees or hoverflies. If we were able to identify the relationship between






abundance and diversity of insect pollinators in a landscape and the pollination of commercial crops such as strawberries or apples, then we could estimate the economic costs of the decline or disappearance of particular species of wild pollinator in terms of lost commercial crop outputs (Hanley et al. 2015). For further examples linking wild species change to market values using a production function approach (e.g., for mangrove conservation in Southeast Asia), see Hanley & Barbier (2009).

### 3.2. Examples of Nonmarket Value Estimates for Wild Species

In the following sections, we provide some examples of the application of economic valuation tools to the task of estimating non-market values for biodiversity conservation. We then discuss some of the problems in using such methods in this context.

**3.2.1. Terrestrial biodiversity: direct values.** Lundhede et al. (2014) apply the choice experiment method in a study of the preferences of Danish people for bird conservation policy, in the context of the pressures brought about by climate change. What is interesting in this study is that the choice experiment design allows the authors to compare values of native Danish birds whose presence in Denmark is threatened by climate change, relative to immigrant birds who might well move to Denmark as a result of climate change. They also compare WTP values according to the conservation status of birds in Denmark and their conservation status in the wider European context. **Figure 1** gives an example of the choice cards used in their analysis. People were willing to pay more to conserve native Danish birds than immigrants. In the case of abundant populations, WTP for protecting immigrant bird populations was actually zero. Generally, WTP is higher to protect a given species in Denmark when it is declining rather than stable elsewhere in Europe.

	Stable	Decreasing	European population	No policy	Policy 1	Policy 2
				Danish population in 15 years	Danish population in 15 years	Danish population in 15 years
Living in Denmark Serin and dipper  Frequent in Europe				<input type="checkbox"/>	<input checked="" type="checkbox"/>	<input type="checkbox"/>
Living in Denmark Eider and common snipe  Scarce in Europe				<input type="checkbox"/>	<input type="checkbox"/>	<input checked="" type="checkbox"/>
Can immigrate to Denmark Woodchat shrike and partridge  Frequent in Europe				<input type="checkbox"/>	<input type="checkbox"/>	<input checked="" type="checkbox"/>
<b>Additional yearly household tax</b>				0 DKK	500 DKK	750 DKK
Please choose only one				<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>

**Figure 1**

Choice card from Lundhede et al. (2014) showing the choices respondents made between alternative conservation policies designed to counter the adverse effects of climate change on birds in Denmark. Adapted with permission from Lundhede et al. (2014) under the terms of the Creative Commons Attribution (CC BY) License, <http://creativecommons.org/licenses/by/4.0>.

Using the same data set, Lundhede et al. (2015) showed that uncertainty over the effectiveness of conservation policies to offset the impacts of climate change on bird populations in Denmark also affected WTP for these conservation programs.

**3.2.2. Marine biodiversity: direct values.** There are many challenges in applying stated preference methods to value changes in marine biodiversity, not least of which is the lack of access of most people to marine flora and fauna and their lack of familiarity (Ressurreição et al. 2011). Nevertheless, an increasing number of published studies have looked at issues such as the benefits of marine protected areas, the environmental impacts of offshore wind farms (Borger et al. 2014), and nonuse values of deep sea biodiversity (Jobstvogt et al. 2014). Aanesen et al. (2015) study the benefits of safeguarding cold water coral communities (CWCs) around the Norwegian coastline. Threats to CWCs include deep sea trawling, oil and gas exploration, mining, and cable-laying. Because these coral communities are found at least 200 m below the ocean's surface, they are remote from peoples' experiences. A valuation workshop approach to data collection was used, in which people's ex ante knowledge could be measured, and through which it was possible to ensure that responses to a stated preference choice experiment on CWCs were well-informed. The choice experiment showed that targeting CWC locations that were particularly important as a habitat for fish was the most highly valued attribute. People also preferred protection to be targeted at areas of the sea of highest value to the oil and gas sector and to deep-sea fisheries (presumably because these were likely to be the areas under greatest threat from biodiversity loss). However, WTP was not particularly sensitive to the size of the protected area.

**3.2.3. Issues with direct valuation.** When we think about the economic value of conserving a particular species, we would expect that the identity of the species matters. Elephants are more loved than jellyfish, a phenomenon termed a characteristic megafauna effect (Metrick & Weitzman 1998). Many stated preference studies have shown such effects to be present and/or that which species we conserve matters in terms of WTP. This is not a problem, as we would expect the direct utility value of a conservation action to depend on what is being protected. Jacobsen et al. (2008) raise an interesting issue relating to whether or not we name species in a list of those to be conserved by some action, even when the named species are unfamiliar. The authors focus on the conservation of heathland in Denmark. They use a split-sample design in a choice experiment. One group of respondents is told that "4 butterfly species, 14 lichen species, 6 beetle species, and one plant species" will be put at risk by not conserving the habitat. The other group is told instead that two specific species, both relatively obscure (the spring anemone and the moth *Euxoa lidia*), are threatened. Results show that stated WTP is almost as high for protecting just one named species as for a list of all 25 unnamed species.

Another problem with stated preference approaches to valuing changes in biodiversity is whether ordinary citizens know enough for their preferences and WTP values to be informative for policy development. Welfare economics has not traditionally taken the view that only the preferences of the sufficiently well-informed should count in a social welfare function. However, some aspects of biodiversity are likely to be very unfamiliar to many people. Approaches such as the valuation workshop were developed specifically to address the problem of applying stated preferences to unfamiliar goods such as biodiversity (MacMillan et al. 2003, 2006). Lundhede et al. (2014) use a quiz to measure participants' knowledge of birds in Denmark and find that better-informed participants had significantly higher WTP for protecting native Danish birds from the effects of climate change than less-informed participants. LaRiviere et al. (2014) measure peoples' ex ante knowledge about cold water corals and found that ex ante better-informed people had a higher WTP and gave less random answers to the choice experiment.

#### 4. MARKET VALUES OF WILD AND DOMESTICATED SPECIES

Production systems developed to deliver foods, fuels, fibers, and pharmaceuticals are considered to be the main cause of the loss of wild habitat worldwide. But the same production systems face a biodiversity problem that is similar to the problem posed by the loss of wild species in natural systems. Production systems are typically simplified ecosystems in which species with desirable traits are exploited, while competitors, predators, pests, and pathogens are controlled, and many of the ecosystem processes and functions critical to growth in natural systems are replaced by artificial substitutes. While the nature of the problem differs from one production system to another, all involve the choice of which species and which traits to promote and which species and which traits to suppress or ignore.

Amongst the least intensively managed production systems are bioprospecting concessions aimed at the identification of compounds useful for the development of pharmaceuticals. One approach to the problem has assumed that all species are equally likely to be the source of a compound that could yield a successful new drug and asked what the marginal value of an additional species might be. Each new species sampled is treated as an independent Bernoulli trial with equal probability of success. The marginal value of a species is then the increment in the expected value of the sample due to the addition of one more species to sample. This approach suggests that while the marginal value of a species is increasing in the profitability of a successful drug, it is likely to be very small (Simpson et al. 1996). A later reevaluation of the same problem relaxes the assumption that the species tested are from an undirected random sample. It finds that if sample selection is directed by research on species with particular promising traits, the information rents on such traits rise, and the marginal value of an additional species with those traits can be quite substantial (Rausser & Small 2000).

Amongst more intensively managed production systems, where traits are frequently manipulated through plant and animal breeding or through direct genetic modification, a similar argument would suggest that the marginal value of conserving, for example, crop wild relatives would be greater than the marginal value of conserving a species selected at random. The optimal size of the pool of exploited species is then a function of the capacity of producers to control for environmental variability.

There is a large literature on the relation between crop diversity, the mean and variance of agricultural yields, and farm incomes. Although homogenization can increase yields in the short run, it is often at the cost of increasing longer-run vulnerability to pests and pathogens, with implications for farm incomes and hence for the private value of crop genetic diversity (Di Falco & Chavas 2007; Di Falco & Perrings 2003, 2005). In low-income countries, farm incomes have been shown to have increasing crop genetic diversity (Birtal et al. 2015, Michler & Josephson 2017). Crop rotations, for example, remain common even in industrial agriculture and are the primary means of managing soil fertility, weed, pest, and disease control in organic systems (Altieri 1999, 2018). The degree to which farmers choose to specialize reflects the private costs and benefits of alternative combinations. While crop prices are critical, the degree of crop genetic diversity depends on two other factors: the effect on yield variance or risk and potential economies of scope (Chavas & Di Falco 2012).

The ecological basis for economies of scope lies in complementary interspecific interactions that increase yields (Li et al. 2014) and niche differentiation. The impact of spatial niche differentiation was initially shown for grasslands, where productivity increases monotonically with the number of species planted (Hector 2011, Tilman et al. 2001). The reason is that in heterogeneous environments, greater species richness better utilizes all ecological niches and is thus more productive. Niche differentiation also operates over time (Yu et al. 2015) and has been shown to have

the same productivity effect in natural forests as in grasslands (Liang et al. 2016). Because farmers and foresters are typically interested in particular, highly valued plant traits, they will be willing to trade off a loss in aggregate biomass for an increase in the abundance of the valued trait. An example of this in rangelands would be a loss in aggregate grassy biomass. Nevertheless, both complementarities and niche differentiation promote on-farm diversity, though this incentive differs across farming systems. Empirically, there is a negative correlation between agrobiodiversity and income, with smaller, more diverse farming systems generally found in poorer countries or in poorer regions within countries. For example, a study of the diversity of crops in small farms in Hungary, using a combined revealed and stated preference approach, found that farmers in the most economically, geographically, and agri-ecologically marginalized communities placed the highest value on crop diversity and specifically on landraces—plants that are morphologically distinct and genetically variable and dynamic and that have distinctive properties in terms of yield, date of maturity, and pest and disease resistance (Birol et al. 2006a,b).

A number of studies in developing countries have investigated farmer incentives to cultivate landraces and found that primary drivers are the value of landraces in consumption and the management of short-term private risk (Gauchan et al. 2005, Smale et al. 2004). Longitudinal survey-based research in Northeastern and Central Highlands communities in Ethiopia, for example, found that the number of landraces cultivated increased at the same time as there was progressive land fragmentation; it also found a reduction in average field size. While average field area planted with sorghum landraces fell by 42% between 1990 and 2010, 56% of farmers increased sorghum landrace richness over the period (Teshome et al. 2016). A study by Chavas & Di Falco (2012) that used panel data from Ethiopia to estimate a stochastic joint production function found evidence that planting multiple varieties generated complementarities among crops and reduced both variance and skewness in yields. Both effects mean positive incentives to diversify. Complementarities among crops are a source of economies of scope, while negative impacts on variance and skewness reduce risk generally, and downside risk in particular.

The private benefits of diversification accrue disproportionately to farmers who are unable to take out insurance against crop failure or who rely heavily on own-production for consumption. That is, they accrue to those who have limited access to financial or product markets. The social benefits of diversification accrue to everyone on the planet. Landraces are a primary source of the genetic material that fueled the development of high-yielding varieties in the Green Revolution. While there are some areas, like Ethiopia, where farmers still have an incentive to cultivate landraces, in most places they have been displaced by high-yielding varieties. The adoption of high-yielding varieties has improved average yields for farmers (while increasing vulnerability to environmental shocks and the spatial correlation of risks—specifically the capacity to deal with drought, pests, or disease). The genetic erosion of landraces has more serious consequences for global longer-term risk. As the number of landraces conserved on farms declines, so does the capacity to combine genetic material in new ways to address global risks such as climate change (Bellon 2009, Bellon et al. 2017). Symmetrically, the on-farm conservation of landraces (and wild crop relatives) involves a positive externality: the maintenance of a continually evolving gene pool (Jackson et al. 2007).

Though not analyzed in the same detail, similar problems exist in other areas of agriculture (Rege & Gibson 2003), in aquaculture and fisheries (Blanchard et al. 2017), and in forestry (Potter et al. 2017). As in crop production, these are all activities in which the range of biodiversity externalities spans both the targeted species (species that are cultivated, hunted, or controlled) and nontargeted species (species that are incidentally impacted by the actions of farmers, fishers, aquaculturists, or foresters). Nontargeted species include wild species affected by loss of habitat, the

off-site effects of on-farm applications of nutrients, herbicides, pesticides, or changes in the abundance of species accidentally selected for or against.

There have been very few systematic attempts to estimate the value of changes in the abundance and richness of species accidentally selected for or against. Most of these focus on accidentally introduced invasive species and rely more on revealed preferences (estimates of resources committed to mitigate or repair damage) than on stated preferences. In 2005, Pimentel estimated that there are approximately 50,000 introduced species in the United States, accounting for annual damage costs in the order of US\$120 billion, and noted that nearly half of all species identified as endangered under the Endangered Species Act are at risk because of competition or predation by invasive species (Pimentel 2014, Pimentel et al. 2005). More recently, a study of the effects of a subset of 1,300 invasive insect pests and pathogens in 124 countries calculated the total invasion cost as a percentage of GDP. The share of GDP identified as at risk in this way ranged from 36% in the case of Malawi to 0.001% in the case of Singapore (Paini et al. 2016). As expected, most of the countries in which more than 10% of GDP is at risk from invasive pests and pathogens are low-income countries characterized by high levels of dependence on agriculture.

Biodiversity also has an impact on human health. In Sub-Saharan Africa, more than half the burden of disease is accounted for by communicable diseases in which the infectious agent is a life form of some kind—bacteria, viruses, fungi, parasites, and prions. In South and Southeast Asia, more than a quarter of the burden of disease is attributed to the same infectious agents. Although there are no data on the value of pathogen diversity in different regions, regional estimates of health expenditures imply substantial differences in value. In 2013, for example, India and Sub-Saharan Africa together accounted for 44% of the global burden of disease, but only 2% of global health expenditures (Murray & Lopez 1997). In the same year, the United States accounted for just under 3% of the global burden of disease but 38% of health expenditures (WHO 2018).

Three methods for estimating the cost of specific diseases relevant here are the cost of illness (COI) method (Hoffmann et al. 2012), computable general equilibrium models (Lee & McKibbin 2004, McKibbin & Sidorenko 2006), and estimates based on the value of a statistical life (Bloom et al. 2011, Molinari et al. 2007). The estimates generated by these methods are significantly different, although all track the association between income and disease cost. At the level of the household, studies using the COI method have found that, relative to income, infectious diseases impose the highest relative costs in the poorest countries (Russell 2004). Indeed, it has been argued that infectious disease in poor countries generates a poverty trap in which per capita income falls as disease prevalence rises.

## 5. ACCOUNTING FOR CHANGING NATIONAL BIODIVERSITY VALUES OVER TIME

Two main approaches have been developed to capture the national consequences of changes in aspects of biodiversity over time: the United Nations System of Environmental-Economic Accounting (SEEA) and the World Bank's wealth accounts. The SEEA (UN 2014) involves four sets of accounts:

- asset accounts that record the volume and value of natural resource stocks, along with changes in those stocks
- flow accounts for pollution, energy, and materials
- environmental protection and resource management expenditure accounts
- indicators that can be integrated in the System of National Accounts (SNA) such as depletion-adjusted net national income and depletion-adjusted net savings (Lange 2014)

The living assets included in the SEEA comprise individual stocks such as forests, livestock, and crops, but they also include wild animals. Although these in principle encompass both assets in private ownership and assets in common property, in practice they only cover assets subject to well-defined property rights. They are valued by the discounted stream of rents generated by the asset (UN 2014). Work is ongoing to better integrate the SEEA with indicators of biodiversity change (Vardon et al. 2017).

The World Bank's wealth accounts build on the concept of adjusted net savings (also known as genuine savings) to show whether countries are investing sufficient amounts to maintain the value of the aggregate capital stock when changes in natural capital are taken into account (Hamilton & Clemens 1999, Hamilton & Hartwick 2005). To obtain adjusted net savings the World Bank starts with the measure of gross saving recorded in the national income and product accounts, and then makes four adjustments:

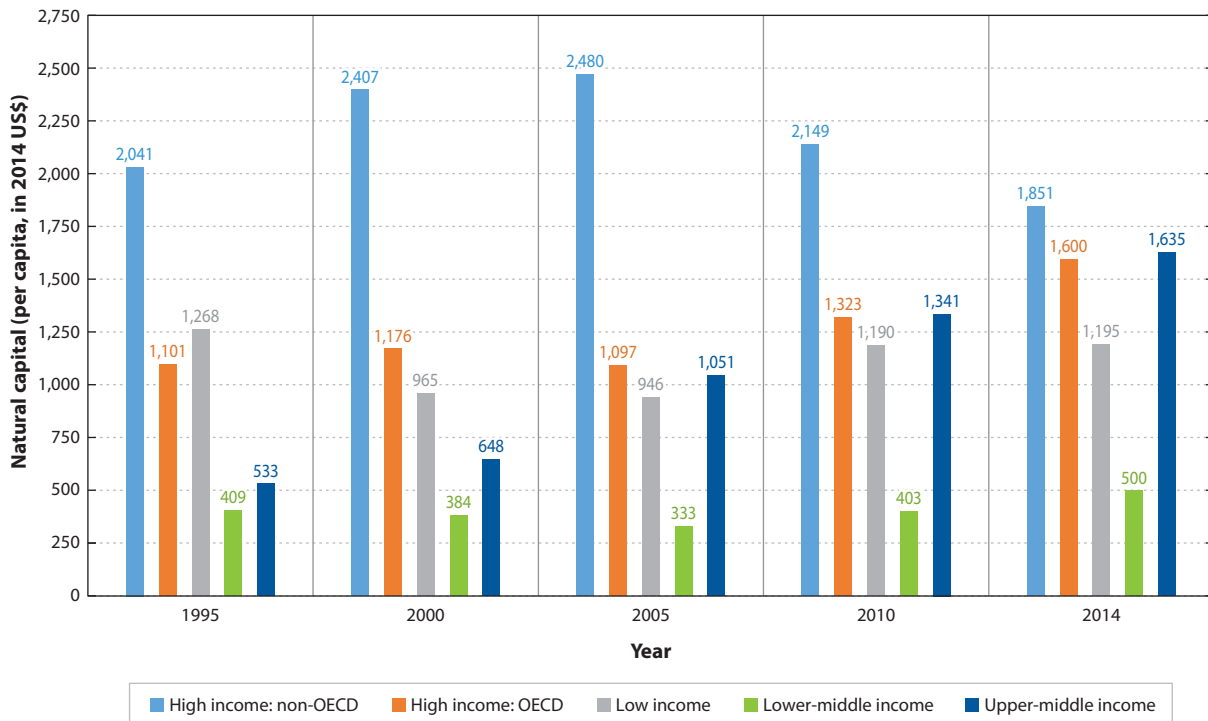
- deduction of fixed capital consumed
- addition of current public expenditure on education (as a proxy for investment in human capital)
- deduction of the depletion or degradation of natural resources
- deductions for damages from carbon dioxide and particulate emissions

Adjusted net savings are then expressed as a percentage of gross national income (World Bank 2011). If the net savings are negative, the inference is that the stock (wealth inclusive of natural capital) is declining. Because the system of national accounts currently fails to measure many changes in stocks, the inference is not always warranted, but negative adjusted net savings is still a reasonable indication that stocks of natural assets are being depleted without building compensating stocks of produced assets (Lange et al. 2018).

Biodiversity is not separately recorded in the wealth accounts but is instead embedded in natural capital. Natural capital comprises energy sources (oil, gas, coal), minerals, agricultural land (cropland and pastureland), forests, and terrestrial protected areas. The relevant natural capital stocks are therefore areas of land committed to different uses. Each is valued at the discounted sum of the rents generated over the expected life of the asset (an infinite horizon is assumed for land), protected areas being valued by reference to the least productive agricultural land in the same area—a quasi-opportunity cost. The best current proxy for the value of all biodiversity is therefore the aggregate value of lands included in natural capital.

A first approximation of the value of wild species in this approach is the value of protected areas. In the latest wealth accounts, it is noted that protected areas in low- and middle-income countries increased by 65% between 1995 and 2014 and now comprise 17% of all land (Lange et al. 2018). On the basis of the valuation method applied in the wealth accounts, this suggests a sharp increase in the value of wild species for these income groups. As a first approximation, the income that people in low- and middle-income countries are willing to forgo by committing land to conservation has increased by around two-thirds. Given population growth rates in low- and lower-middle-income countries, however, if values are calculated in per capita terms only upper-middle-income countries show a significant increase. **Figure 2** indicates that the per capita value of protected areas in low- and lower-middle-income countries in fact changed very little in this interval. The dual effect of high population growth rates and low agricultural productivity—which fixes the quasi-option value of protected areas—depresses the per capita value of protected areas in these income groups.

The disconnect between the trends revealed by the wealth accounts and changes in the status of endangered species reported in the IUCN Red List follows directly from differences in the methods employed to generate the two indicators. Absent estimates of the value of protected



**Figure 2**

Natural capital in protected areas by income group. Value is per capita in constant 2014 US\$. Data from World Bank (2018).

areas that reflect changes in the richness and abundance of the species they contain, or in the wider ecosystem services they provide, the World Bank uses a coarse proxy for the opportunity cost of the land. There is a similar disconnect between the observation in the wealth accounts that the per capita value of agricultural land increased by 27% between 1995 and 2014 and conclusions reached from a review of changes in crop genetic diversity (Pingali 2017). In neither case does the estimated value of land reflect future consequences of changes in the species cultivated or maintained on that land. Acquiring data on the longer term impacts of changes in biodiversity on the value of natural assets for use in national sustainability accounting remains a priority for the future.

## 6. CONCLUSIONS

Because many of the goods and services supported by functional groups of species are unpriced public goods, they are undersupplied by the market. Valuation makes it possible to identify the economic costs of biodiversity decline and provides an argument for devoting significant resources to corrective measures. Undertaking cost-benefit analysis of specific conservation actions also becomes possible (if challenging), as does a comparison of the relative merits of alternative conservation actions on preference-based grounds (for example, decisions over which species or systems to target for conservation actions, or the acceptability of specific management measures such as lethal control).

Until recently, the main focus of biodiversity policy was the provision of habitat for endangered wild species through protected areas. In the last decade, however, a broader range of biodiversity policies have been adopted to try to meet the Aichi Targets of the Convention on Biological Diversity (Perrings et al. 2011). These include efforts to eliminate subsidies harmful to biodiversity, to preserve genetic diversity in agriculture, aquaculture and forestry, and to control invasive species pathways (Tittensor et al. 2014). All require instruments designed to change landholders' incentives to reflect the social cost of biodiversity loss. Understanding the implications for human well-being over time depends on improvements in valuation and resource accounting at the national level (Vardon et al. 2017). As noted above, we still have far to go in this regard.

Most early economic research on the value of biodiversity sought to uncover peoples' preferences for wild species and the goods and services they produce. Attention focused on charismatic megafauna, exotic birds, marine mammals, and nationally important landscapes known to be under threat. Relatively little attention was paid to the diversity among species responsible for ecosystem processes and functions, to biodiversity in agriculture, aquaculture, forestry and fisheries, or to negatively valued species—pests and pathogens (Kumar 2010). As our review makes clear, this imbalance is now being gradually corrected. However, it should be obvious that the evidence base on the value of biodiversity is very far from being comprehensive. We have patchy evidence on the value of conserving a small number of species and habitats; for many species, no valuation estimates exist, whereas very few studies have tried to estimate economic values for changes in biodiversity indicators rather than individual species. Such incompleteness in coverage makes widespread application of cost-benefit analysis to biodiversity conservation problems impossible for the foreseeable future. It does, however, invite further research effort to fill the more important evidence gaps.

Looking forward, we identify two areas where we expect to see progress made. The first is the economic value of diversity within functional groups of species. While there are studies of particular functional groups, such as pollinators, the focus has been on the agricultural value at risk from pollinator decline (Gallai et al. 2009), rather than the role of pollinator diversity in providing pollination services over time (Kleczkowski et al. 2017). Ecological studies have shown that the diversity of functional groups increases both stability (Tilman et al. 2005) and resilience (Carpenter & Gunderson 2001) through one of three mechanisms: overyielding, statistical averaging, and compensatory dynamics (Cardinale et al. 2012, Hooper et al. 2005, Kinzig et al. 2002). Although this insurance function is the basis for the value of the regulating services, referred to earlier, and although it maps well to portfolio theory, there are few studies on the impact of changes in the diversity of specific functional groups for environmental risk (Admiraal et al. 2013).

The second area is improvements to the transferability of direct biodiversity values between contexts. Research shows that people's WTP for gains in a specific species are very much context and knowledge dependent (LaRiviere et al. 2014, Lundhede et al. 2014). This is as it should be, one suspects: how much I am willing to pay to conserve woodpeckers in my local area is likely to depend on their conservation status locally, the extent to which they are declining nationally, and how much I understand about their role in the ecosystem. Yet this makes it hard for policy analysts to transfer values of biodiversity conservation across the many contexts in which conservation decisions are made and between the many dimensions of conservation policy. A better understanding of what drives heterogeneity in direct biodiversity values across people and contexts is one key step forward. Knowing how best to incorporate this knowledge within practical benefits transfer models is a harder step to accomplish, but one that is essential if economic insights are to be more fully integrated into biodiversity policy.



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### Errata

An online log of corrections to *Annual Review of Resource Economics* articles may be found at <http://www.annualreviews.org/errata/resource>