The Economic Value of Biodiversity

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Abstract

Biodiversity is declining world-wide, and the costs of biodiversity losses are increasingly being recognised by economists. In this paper, we first review the multiple meanings of “biodiversity”, moving from simple abundance-weighted species counts to more complex measures which 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, and the functional relationships between changes in biodiversity and economic value. 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 which remain to be solved.
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 an anthropocentric perspective, biodiversity includes the variety of plants, animals, and other life forms (bacteria, archea, 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 (Millennium Ecosystem Assessment 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 the 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 an area 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 ecosystems—the number of taxa that is unique to each system being compared. For two systems, the Sørensen index of beta diversity (Sørensen 1948), $b = \frac{2c}{S_1 + S_2}$, relates the number of species common to both systems, $c$, to the total number of species in those systems. It too 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 in the whole area being evaluated. For two systems it is $g = S_1 + S_2 - c$, a count of the number of distinct species in the whole area (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, $PD_{NODE} = \sum_{i} n_i$, 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, \( D_D = \dim_{i \min} \), due to Solow et al (Solow et al 1993) and Weitzman (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 diversity, on the other hand, explicitly weights each species by its relative abundance, while omega diversity explicitly weights species by their phylogenetic distance from other species. Other measures focus on different characteristics of species. 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 due to (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 paper 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 market commodities, 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 the use made of biodiversity frequently takes 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, Millennium Ecosystem Assessment 2005, Secretariat of the Convention on Biological Diversity 2010, Secretariat of the Convention on Biological Diversity 2014)(Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) 2018) 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 current 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 may be taking 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) put it, “..valuing biodiversity is a key challenge to environmental economists”. The economic value of biodiversity can be characterised as either “direct” or “indirect”, in terms of the way in which changes in biodiversity affect human well-being (Bartowski, 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 the greater the number of bird species in his local forest, or the higher the abundance of individual species populations. It is also possible that individuals derive utility from higher levels of biodiversity due to “what is does” – that is, they 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 (Richardson and Loomis, 2009; Morse-Jones et al 2012); their conservation status (eg scarce versus abundant, or a declining or increasing population trend: Lundhede et al, 2014) or how well they are known (Christie et al, 2006). There is also evidence that people’s WTP for conserving a particular species depends on its cuteness or charisma (Di Minin et al, 2013), and its uniqueness or endemicity (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 of the population abundance of a given species, can be defined in terms of both use and non-use 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 5 rare species of birds on a wetlands visit, rather than 1 rare species). But non-use, 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 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 focussed 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. 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 willingness to pay which depended on (i) whether rare or common species were protected (ii) whether these species were well-known or unfamiliar to most people (iii) whether the policy would merely slow down current rates of loss rather than stopping or reversing this trend; and well as (iv) the consequences of the policy in terms of habitat restoration versus habitat creation, and for (v) ecosystem services supplied. Czajkowski et al (2009) showed that Polish people were willing to pay for biodiversity policy which enhanced and protected “natural ecosystem processes”, protected rare species, or enhanced ecosystem components such as dead wood, natural ponds and clearings within the Bialowieza 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, in Hanley et al (2003) people in Scotland had different utility from wild geese conservation according to whether shooting was used as part of a management policy; whilst in Czajkowski et al (2009), 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. In Hanley et al (2010), the means of protecting an endangered bird of prey (the hen harrier) was, however, an insignificant determinant of WTP. All respondents cared about was the biodiversity outcome.

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 and Requate (2013) look at the problem of multi-species 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). Modelling 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, since this 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 De Falco and Perrings (2005) for farms, and MacPherson et al. (2017) for forests), as is 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 (ES) within a given system and spatial/temporal context, since it is these ecosystem services which provide the connection to human well-being and thus to economic value (Naeem et al, 2017). 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). Baumgartner and 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. This distinction of the economic value of resilience into an insurance value and an effect on expected income from ecosystem services is a useful one.

The form of the relationship 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 biodiversity is complicated to fit into the ecosystem services concept since it plays three roles: as a determinant of ecosystem functioning; as an ecosystem service (eg pollination); and as a final outcome that people care directly about. The relationship between biodiversity and one aspect of economic value – agricultural output, say – may well be very different to the relationship of the same biodiversity indicator to another kind of economic value, such as carbon storage. Whilst some functional relationships will imply trade-offs are the norm, so that increasing biodiversity typically comes at the expense of lower incomes from farming (eg Armsworth, 2012, for upland farming in the UK), in other cases rising biodiversity is consistent with increasing economic values such as non-use and other cultural ecosystem values.

Paul et al (2018) 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 describe a given system depends on whether a 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 ES. 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 specialisation, the adoption of non-native varieties and climate change (Freer-Smith and 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, 2016) 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.

Since the value of any asset lies in the discounted stream of services it offers, 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 amongst the environmental assets that generate ecosystem services. In terms of inter-temporal efficiency, then from the Hotelling principle the decision as to whether environmental assets should be conserved depends on the expected rate of change in their value relative to other assets (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 converted to 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 and 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 which determines how valuable the flow of benefits over time is. 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, 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, or sediment, 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). The question of what such a strategy implies for natural assets including biodiversity has attracted attention since the 1970s, when conditions allowing aggregate wealth to be non-declining even if natural capital were being depleted were identified. These conditions are that rents earned on the depletion of natural resources should be reinvested in other, reproducible elements of comprehensive wealth (Hartwick 1977, Hartwick 1978, Solow 1974, Solow 1986). The implications of this for biodiversity and other ecosystem components follows 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 still be sustainable 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 non-market value of wild species
3.1: methods

Economists often distinguish three approaches to estimating values for non-market goods such as the conservation of wild plants and animals. Two of these (i) stated preference methods, and (ii) revealed preference methods are designed to uncover peoples’ willingness to pay for/accept compensation for the loss of goods and services for which there are no markets. A third, (iii) production function methods, are used either to infer the implied willingness to pay for environmental inputs to the production of particular goods or services, or to project the environmental consequences of particular behaviors. 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 programmes since the 1970s. Indeed, some of the earliest applications of the method were concerned with these two issues (see Hanley and Spash, 1993). Contingent valuation (CV) can be used to estimate WTP for preventing a decline, avoiding damage to, or achieving an increase in a specific wildlife species (e.g., a species conservation programme for marine mammals: Samples et al, 1986) or for habitat protection (e.g., Carson, Wilks and Imber, 1994; Hanley and Craig, 1991). Choice Experiments (CE) were first used in environmental economics in the early 1990s, and are another stated preference method which 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 a species is endangered or not, 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 visits, 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 and Markandya, 2006). Hedonic pricing is rarely applied to estimating direct values of biodiversity, although many studies relate house prices to forest cover, proximity to farmland and urban green spaces, variables which may all be related to measurable variations in biodiversity.

Where the value of some good or service is known, 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 bumble bees or hover flies. 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 (external) effects of the decline or disappearance of particular species of wild pollinator on commercial crop outputs. Ideally we would also wish to count the contribution that wild insect pollinators make to the pollination of non-commercial plants; and their direct utility value to the public since people like bees (Hanley et al, 2014).
For further examples linking wild species change to market values using a production function approach (eg for mangrove conservation in SE Asia), see Hanley and Barbier (2009).

3.2 Examples of non-market value estimates for wild species.

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 WTP 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. 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 programmes.

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 studies have been published looking at issues such as the benefits of marine protected areas, the environmental impacts of off-shore wind farms (Borger et al, 2014) and non-use values of deep sea biodiversity (Jobstvogt et al, 2014). Aanesen et al (2015) study the benefits of safeguarding cold water coral communities (CWC) around the Norwegian coastline. Threats to CWC include deep sea trawling, oil and gas exploration, mining and cable-laying. Since these communities are found at least 200m below the ocean’s surface, they are “remote” from most peoples’ experiences. A valuation workshop approach to data collection was therefore 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 CWC were “well-informed”. The choice experiment showed that targeting CWC locations that were particularly valuable as a habitat for fish was the most highly-valued attribute; whilst 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 since 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.

Issues with direct valuation.

When we think about the economic value of conserving a particular species, then if that species is known to people, we would expect that the identity of the species matter. Elephants are probably more loved than jelly fish, a phenomenon termed a “characteristic megafauna effect” (Metrick and Weitman, 1998). Many SP 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, since 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 a list of all 25 un-named species.

Another problem with SP 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) and MacMillan et al., 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 domesticated species
Production systems developed to deliver foods, fuels, and fibers 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 quite similar to the problem posed by the loss of wild species. Production systems are extremely simplified ecosystems in which species with desirable traits are encouraged, while competitors, predators, pests, and pathogens are discouraged, and many of the ecosystem processes and functions critical to growth in natural systems are replaced by artificial controls.

There is a large literature on the relation between crop diversity, the mean and variance of agricultural yields, and farm incomes. While 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, di Falco & Perrings 2005). Farmers will opt for some level of diversity. 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 2018, Altieri 1999). 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 things: 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 so is more productive through a niche differentiation effect. 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).

Since farmers and foresters are typically interested in particular, highly valued plant traits, they will be willing to trade off a loss in biomass for an increase in the abundance of the valued trait. 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 tending to occur in poorer countries, or 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, genetically variable and dynamic, and have distinctive properties in terms of yield, date of maturity, pest and disease resistance (Birol et al 2006a, Birol et al 2006b).

A number of studies in developing countries have investigated farmer incentives to cultivate landraces, and found that primary drivers are the value of land races in direct consumption and the management of short-term private risk (Gauchan et al 2005, Smale et al 2004). Longitudinal survey based research in North-Eastern and Central Highlands communities in Ethiopia, for example, found the number of landraces cultivated to be increasing at the same time as there has been progressive land fragmentation, and a reduction in average field size. While average field area planted in 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 and di Falco 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 offer 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 (Chavas & di Falco 2012).

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 farm declines, so does the capacity to combine genetic material in new ways to address secular 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 non-targeted species (species that are incidentally impacted by the actions of farmers, fishers, aquaculturists, or foresters). Non-targeted species include wild species affected by loss of habitat, by the off-site effects of on-farm applications of nutrients, herbicides, pesticides, or by 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 something like 50,000 introduced species in the USA accounting for annual damage costs in the order of $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 1300 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 .001% in the case of Singapore (Paini et al 2016). As might be 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 impacts on human health. In sub-saharan Africa, for example, 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 South East Asia more than a quarter of the burden of disease is accounted for by the same infectious agents. While 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, while India and Sub-Saharan Africa together accounted for 44% of the global burden of disease, they accounted for only 2% of global health expenditures (Murray & Lopez 1997). In the same year the USA accounted for just under 3% of the global burden of disease but 38% of health expenditures (World Health Organization 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 2012, Molinari et al 2007). The estimates generated by these methods are significantly different, though 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 cost 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 Accounts (SEEA) and the World Bank’s wealth accounts. The SEEA (United Nations et al 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 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 (United Nations et al 2014). Work is on-going 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. Since 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. Marine-protected areas are not included. 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). Using the valuation method applied in the wealth accounts this suggests a sharp increase in the value of wild species for these income groups. 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 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 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 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 meet the Aichi Targets of the Convention on Biological Diversity (Convention on Biological Diversity 2018). These include, among other things, 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).

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 amongst 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 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 habitat; for many species, no valuation estimates exist; whilst 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 role of pollinator diversity (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 2005, Kinzig et al 2002). Although this 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 of the impact of changes in the diversity of specific functional groups for environmental risk (Admiraal et al 2013).

The second is improvements to the transferability of direct biodiversity values between contexts. Research shows that people’s willingness to pay for gains in a specific species are very much context- and knowledge-dependent (Lundhede et al, 2014; LaRiviere et al., 2014). This is at 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.
FIGURE ONE – CHOICE CARD FROM LUNDHEDE ET AL (2014) showing choices respondents made between alternative conservation policies designed to counter the adverse effects of climate change on birds in Denmark.
Natural capital in protected areas by income group. Value is per capita in constant 2014 US$.

Source: (World Bank 2018)
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