Oxford Review of Economic Policy, Volume 28, Number 1, 2012, pp. 1–21

The economic analysis of biodiversity: an assessment

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Abstract Biodiversity is complex, difficult to define, difficult to measure, and often involves international and intergenerational considerations. Biodiversity loss presents significant economic challenges. A great deal of economics is required to understand the issues, but a simple and important observation is that most species and ecosystems are not traded in markets, so prices are often absent and biodiversity is under-provided. Despite the formidable obstacles to high-quality economic analysis, economics has plenty to offer to biodiversity policy. First, economic valuation techniques can be employed to roughly estimate the value of the benefits provided by biodiversity and ecosystems. Second, assessing the 'optimum' amount of biodiversity involves recognizing that the conversion of natural capital into manufactured and human capital has so far generated vast amounts of wealth. While there may have been 'too much biodiversity' in the past, economic analysis suggests that this is a difficult position to hold now. Third, econometric techniques and carefully designed policy studies can assist in determining what policies are most suited to different contexts to cost-effectively reduce biodiversity loss. Fourth, political economy is helpful because international coordination is often required—ecosystems do not respect national borders and many biodiverse ecosystems are in poorer countries. This paper synthesizes the issues and proposes a research agenda, which includes improving the measurement and accounting of natural capital, improving valuation techniques and theory to provide greater guidance as to the 'optimum' biodiversity, and developing our understanding of the merits of different alternatives for government intervention to reduce biodiversity loss.

Key words: biodiversity, natural capital, valuation, resource economics, CBD, CITES, protected areas, ecosystem services, eco-credits

JEL classification: Q50, Q57, Q58, Q10, Q20, Q30

I. Introduction

Biodiversity loss should be regarded as one of the greatest economic problems of this century for two reasons. First, it is economic growth and development that has caused

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The authors wish to thank Jonathan Colmer for very prompt research assistance and Daniela Miteva, Paul Ferraro, Charles Palmer, Chris Allsopp, Paul Armsworth, Nick Hanley, and Subhrendu Pattanayak for helpful comments. Hepburn acknowledges the financial support of the Grantham Foundation for the Protection of the Environment, as well as the Centre for Climate Change Economics and Policy, which is funded by the UK's Economic and Social Research Council.

doi:10.1093/oxrep/grs014

biodiversity loss and ecosystem degradation.[1] The rapid expansion of the population from around 2 billion in 1900 to over 7 billion today, combined with the enormous growth in income and consumption, have already wreaked havoc on our planet's natural ecosystems. Current environmental pressures will only increase as the human population swells from 7 billion to 9–10 billion by 2050 (UN, 2011) and as the number of so-called 'middle-class' consumers grows from 1 billion to 4 billion people (Kharas, 2010), driven by a materials-intensive growth model.[2] This economic growth has led to dramatic reductions in poverty but also severe ecosystem degradation.[3]

Second, future losses of biodiversity and ecosystems may significantly reduce the productivity of our economic systems. By 2100, on current projections, we may have eliminated half the species on Earth (Wilson, 1994; Thomas *et al*., 2004), and raised global temperatures by more than 3.5 degrees centigrade (IPCC, 2007). The rainforests may have been largely deforested by 2100, the oceans depleted, and land degradation may have significantly affected agricultural productivity. We are living through one of the great extinction episodes in geological history.[4] It is implausible that this will not dramatically affect the patterns of consumption and production around the world.

Technical progress might help to counter some of these negative effects. New techniques of efficiently manufacturing food, and new sources of energy, may facilitate a transition to 'green growth' and development. But this is far from certain. Massive biodiversity loss and climate change represent an unprecedented and enormous experiment with life on Earth, and it is astonishing that biodiversity is not a topic routinely covered in every standard economics textbook. Instead, one of the greatest resource allocation questions has been largely ignored by the mainstream economics profession. Dasgupta (2008) is correct to note that '[n]ature has been ill-served by 20th century economics'.

One of the reasons biodiversity has been relegated to the margins of economics is that there are formidable obstacles in the way of high-quality economic analysis. Biodiversity is a particularly intractable economic problem. It has system properties that defy easy definition. It is more than the aggregate sum of species: some species play a vital role in the survival of ecosystems; some provide key ecosystem services to humans; some are positively harmful to humans; species depend upon each other; and policies aimed at biodiversity are often oblique, aimed at preserving habitats rather than particular species. Biodiversity is a series of overlapping public goods from the local to the global scale.

Conceptually, biodiversity shares a number of core issues with climate change—both involve intergenerational, global public goods—but is even more analytically demanding. Climate change is simple by comparison: the atmosphere comprises a set of gases, changes in its composition can be measured, and empirical estimates can be made of the relation between these changes in composition and temperature changes. The climate change literature has a well-defined research agenda, and has concentrated on the

[1] Ecosystem degradation and biodiversity loss often, but not always, accompany one another. For an assessment of the degree of alignment between policies to protect biodiversity and those to support ecosystem services, see the article by Stephen Polasky, Kris Johnson, Bonnie Keeler, Kent Kovacs, Erik Nelson, Derric Pennington, Andrew J. Plantinga, and John Withey in this issue (Polasky *et al*., 2012). They conclude that: 'In general, investing in conservation to increase the value of ecosystem services is also beneficial for biodiversity conservation, and vice versa.'

[2] Middle-class consumers are defined as those with daily *per capita* spending of between \$10 and \$100 in purchasing power parity terms.

[3] Charles Palmer and Salvatore Di Falco (2012, this issue) explore the relationship between the two.

[4] See Barnosky *et al*. (2011) for comparison with previous extinction episodes.

carbon price and related policy instruments. In contrast, the biodiversity literature is much more heterogeneous. The economic tools available are not yet 'fit for purpose' for the analysis of biodiversity, though they are a helpful starting point.

Furthermore, a precise and operational definition of biodiversity remains elusive. Without a clear definition of biodiversity, it is hard to measure its loss, and harder still to design the appropriate policy instruments and evaluate the impact of such policies. For some, biodiversity loss might be proxied by the number of species or diversity indices; for others biodiversity policy should focus on preserving rainforests, wildernesses, and specific areas as nature reserves and protected habitats.[5]

The structure of this assessment is as follows. Section II addresses the concept of biodiversity, its measurement, and the nature of the resource-allocation problem. Section III reviews the standard valuation techniques that are employed in cost–benefit analysis. Section IV briefly considers the substitutability of man-made and natural capital, depletion, and renewable and non-renewable resources. Section V considers the policy implications: the role for payments for ecosystem services (PES), eco-credits, compensation mechanisms, and the use of prices for the environment, in addition to the designation of protected areas. Section VI considers the institutional dimension, notably the problem of biodiversity treaties and agreements. Section VII looks at implementation and accounting and the embedding of biodiversity within the core of economic policy. Section VIII concludes.

II. What is biodiversity?

Given that governments and international organizations regularly produce biodiversity statements, agreements, and policies, it might be concluded that biodiversity as a concept is both well-defined and measured. It would then be possible to assess policies to see whether they increase or decrease biodiversity. But cursory examination leads to a very different conclusion: there is no obvious and agreed definition of biodiversity, and in practice there are a number of sub-definitions and concepts upon which policy is targeted. For example, according to the Food and Agriculture Organization (FAO), biodiversity encompasses the:

> variety and variability of plants, animals and micro-organisms, at the genetic, species and ecosystem level, that are necessary to sustain the key functions of the agro-ecosystem, including its structure and processes for, and in support of, food production and food security. (FAO, 1999)

This defines biodiversity partly in terms of its contribution to economic production. Other definitions focus simply on the ecological aspects of the system.[6] Any empirical

[5] See for example Weitzman (1998) who argues that the species located further apart on the phylogenetic tree are more valuable than those that are closer genetically to other species, since the former possess more unique genes. See also Brock and Xepapadeas (2003) for another approach to valuing an individual species. For a presentation of a broader ecosystem approach, see Secretariat of the Convention on Biological Diversity (2004). Polasky *et al*. (2012) also consider the relationship between ecosystem conservation and biodiversity protection.

[6] World Resources Institute *et al*. (1992) and Noss (1990).

estimate of the 'amount' of biodiversity is going to be a crude approximation at best. The concept is hugely informationally demanding.

Biodiversity is a contraction of 'biological diversity', and hence has two parts. 'Biological' refers to natural life on Earth and 'diversity' is variety that can be captured by an index. For the 'bio' part, we can count the number of species (though for biologists even this is a contested concept) and estimate their populations. We can also estimate critical thresholds beyond which a species is condemned to extinction, effectively becoming a non-renewable resource. But the essence of biodiversity is the second part, 'diversity'—the variety and interdependency of species, in turn depending upon habitats. Ecologists have studied critical habitats and the thresholds for size and quantity beyond which species loss accelerates. In particular, island populations have been studied, and the relations between the area of a habitat and its species richness have been estimated.[7]

A further difficulty relates to the absence of a 'baseline' against which to evaluate current biodiversity. There is no real 'wilderness' left, and no 'balance of nature'. Humans have been modifying 'nature' since the Pleistocene, eliminating the mega fauna and hence changing the balance between forest and open plains. 'Nature' itself is best viewed as subject to continuous change, not a series of equilibria (Rohde, 2006).

Within current ecosystems, some species are more important than others. The loss of tigers at the peak of a food chain might have a very different effect from the loss of less charismatic species further down, which might support an entire ecosystem—a 'keystone species'. Yet predators can also play key roles—without them herbivores flourish, changing the vegetation (Lotka, 1920; Volterra, 1931). There is also a human value dimension some species are more highly valued by humans than others, both for their intrinsic value and for the services they yield. Thus corn, wheat, and sugar cane are highly valued, to the extent that other species are pushed aside, while mosquitoes are not. Attempts to define 'optimal biodiversity' are therefore inherently difficult, if not impossible, and in practice much of the literature is confined to looking at marginal changes from the *status quo*.[8]

Diversity is a more familiar economic concept, and diversity is measured in financial theory, regulatory economics, and in measures of energy security among others. It is relatively straightforward to develop statistical measures that can be applied to a set of species. On one level, the number of species in a given area of land can simply be counted up. Thus the claim that rainforests are biodiverse might mean that they have more species per hectare than other habitats. Alternative ecological indices include the Shannon index[9] and the Simpson index[10], which gives the probability that two independently sampled individuals are of the same species. So, as with the Herfindahl index in industrial economics, a measure of zero represents infinite diversity (perfect competition), while a measure of unity represents no diversity (monopoly). More sophisticated biodiversity measures might apply ecological weights to the species. Further, we might identify key indicator species—species occupying key niches in an ecosystem—and highlight them in a diversity measure. Another measure, following the FAO definition,

[7] A 'species–area relationship' is often approximated by a power function of the form $s = cA^z$ in which s is the number of species, A is the area, and c and z are fitted constants (Preston, 1962).

[8] It is not always clear what constitutes a marginal change. See Fisher *et al.* (2009).

[9] The Shannon index quantifies the uncertainty in predicting the identity of a species when drawing individual units from a random sample of the total population.

[10] The Simpson index sums the squares of the proportion of each species in a given area.

might work by applying weights based on the economic productivity of the species. And so on.

The important point here is that what we measure depends upon what we define as biodiversity, and that in turn depends upon what the question is that the measure is supposed to answer. Economists typically start the analysis from a human perspective, placing values on individual species, and then aggregate upwards. This aggregation ignores the ecosystem properties, and this in turn means that aggregating individual valuations of species tells us little about the value of biodiversity as a whole. We return to this point below when we consider The Economics of Ecosystems and Biodiversity (TEEB) and related exercises in larger-scale system valuations.

Having explored the challenges in defining biodiversity, the next issue is to explore what sort of economic problems are raised by biodiversity. The economic approach to biodiversity sees the problem as one of market failure and, in particular, of externalities and public goods.[11] A public good is both non-excludable and non-rival, while positive externalities from biodiversity might potentially relate to both rival and non-rival goods. In both cases, there will be over-exploitation and under-provision. Without intervention, biodiversity has little value captured in a price and it will be under-provided by the private sector. Biodiversity shares this with climate change abatement, but climate change relates to a single global atmosphere, whereas biodiversity is a plethora of overlapping public goods. Public goods include species at one level, and national parks at another, right up to the Amazon and Antarctica. Ecosystems can also be described as public goods. These public goods may even conflict: preserving one species may reduce the availability of another. Preserving a national park may eliminate species which depend on human activities prohibited within national parks. The best economists can do is to identify which public goods are being pursued, and at what scale (from local to global), and then determine how best to design policy instruments appropriate for those public goods.

A particularly difficult feature of biodiversity is the critical thresholds: above a critical threshold, a species might be classified as a renewable pubic good; below it is non-renewable and condemned to extinction. These thresholds are uncertain, and hence in making decisions under uncertainty, the question of risk aversion rises. In much of the environmental literature, the precautionary principle—that we should be risk-averse in the face of such uncertainty—is evoked in this context, both at a species level and also more generally.

In sum, biodiversity is difficult to pin down conceptually, and there are various competing definitions that might be employed for quantitative research. A further problem is that there are likely to be many species that have not yet been discovered, particularly insects and amphibians. While difficult, these problems do not prevent useful economic research, using tools and concepts including public goods, externalities, non-linearities and threshold effects, and the economics of information. One economic tool which is critical to biodiversity policy is the various economic valuation methods that have been devised, refined and incorporated into non-market cost–benefit analysis.

[11] See, for example, Fisher *et al.* (2009) for details and discussion on the classification of ecosystem services.

III. Valuing species and landscapes

The valuation of a species is reasonably well-researched in the economics literature, and there has been substantial progress over the last decade (see the article in this issue by Giles Atkinson, Ian Bateman, and Susana Mourato (2012)). Economists tackle this valuation problem in two ways: bottom up, using the traditional tools of valuation of marginal changes in biodiversity; and top down, where systems analysis is employed to attempt to generate an aggregate valuation. The Stern Review on the Economics of Climate Change (Stern, 2007) provided a model of how valuation estimates could capture the political debate. The TEEB exercise, which some regarded as a Stern-type analysis, did not produce an aggregate value on biodiversity for fairly obvious analytical reasons. Given that biodiversity (and, indeed, the climate) are what might be described as necessary conditions for human existence, putting a precise number on the aggregate value of either is open to ridicule. It is a category mistake.

Economic valuation techniques try to place a monetary value on species so that they can be included in resource allocation and their conservation can be traded off against other uses of scarce resources. These 'valuations' are attempts to work out how much should be spent on conserving a species or habitat, given that the monies could be spent to some other—competing—end. They are not estimates of fundamental ethical value or of ecological importance, and confusing the two gives rise to a serious misunderstanding of economic valuation techniques. The question they address is a very limited one.[12]

Valuation techniques are required because most species do not have markets, and hence market prices. Outside agriculture and zoos, they are not typically owned, and rarely traded. There are exceptions—for example, rare animals and plants are subject to collection, and collectors will tend to pay more, the rarer they become. Indeed, this relation of valuation to scarcity has its own dynamic: if a species (or some attribute of a species) can be stockpiled, then there may even be incentives to make it rarer. As Charles Mason, Erwin Bulte, and Richard Horan (2012, in this issue) demonstrate, stockpiling rhino horn and tiger parts may be a profitable strategy if the rhinos and tigers are then pushed towards extinction. A further example is the collapse of the great auk population in the North Atlantic through harvesting and hunting. This led trophy hunters to try to kill the last one. (They succeeded off Iceland in 1844.)

Economic valuation methods fall into one of three categories: revealed preferences methods (including hedonic pricing and the travel cost method); stated preferences (including contingent valuation—carefully constructed surveys—and choice experiments); and production function approaches. Given the technical and informational problems, these techniques are best regarded as snapshots from different angles, each trying to approximate the underlying value, and in practice it is useful to compare and contrast the estimates.

Revealed preference methods take a behavioural approach, trying to use the choices people make to reveal their underlying preferences by making clever use of econometric methods. For instance, the price we pay for assets such as houses near particular

[12] But see Sagoff (2004) who provides an alternative view, claiming in effect that the ethical and the economic are conflated.

habitats might carry a value premium that can be estimated, and the value of time and effort people expend to visit habitats in order to see particular species can also be calculated.[13] These valuation techniques are necessarily imperfect. All sorts of other factors affect house prices and our journeys. Standard controls can eliminate some, but not all, of this bias. But stated preference approaches are vulnerable to biases, too—in the way information is provided to the subjects, and in the way subjects respond. The practitioner is necessarily forced to choose the least-worst method in the specific context, and where possible to compare the different snapshots that result. For this reason, economic valuation is a case-by-case exercise.

In many cases, economic methods produce monetary valuations for species, biodiversity, and ecosystems that are considered 'too low' by 'experts', because they do not and cannot capture all of the relevant benefits of the species. For many ecologists and environmentalists, the problems are so great as to render the techniques at best useless and at worst positively misleading. For some, taking a human-only perspective is to take too narrow a view of nature: many argue that nature has intrinsic value.

These critiques conflate two different issues. Economic valuations are necessarily incomplete, but incompleteness is not a reason for discarding them. Rather, it suggests that the role of the experts is to help to address the question more precisely. In any event, there are many circumstances where incompleteness does not matter. Consider the policy issue of whether to destroy some natural asset in order to build a road. Assuming the cost of building the road is known, all that matters is to discover whether even an incomplete economic valuation of the natural asset exceeds the costs. Where it does, then the decision to preserve the asset is determined. Where it does not, then more detailed analysis may be required.

Intrinsic value is a different issue. To claim something has value above and beyond human consideration raises a host of questions about where such additional value comes from, how it could be justified, and, in particular, the objective basis upon which it relies. In designing biodiversity policy, such ethical considerations cannot, of course, be ruled out, but it is beholden upon those who advance this view to explain what should and should not be conserved, and to explain why resources should be expended in their preferred way, rather than on other alternatives. A number of green philosophical approaches take a cavalier approach to the consequences of strong sustainability and severe restrictions imposed upon permissible trade-offs can be made, but fail to explain how in practice the implications may work out.[14]

It might be argued that the difficulties are so great that cost–benefit analysis (CBA) should be ruled out. But then, what are the alternatives, given that decisions have to be made? Resources are not infinite: preserving a species may mean that houses cannot be built or funds cannot be spent on something else. Resources are unfortunately scarce, and allocations have to be made.

To date, valuation has played relatively little part in biodiversity policy, although, as we shall see, this may be changing. Decisions about species and habitat are typically administered though the application of rules and command-and-control regulation. Planning law gives the job of weighing up the case for and against a development that might harm biodiversity to an official or a judge. The answer tends to be binary: it

[13] Thousands for example travel to Loch Garten in Scotland to see the ospreys.
[14] For a sample of views, see parts 4 and 5 in Dryzek and Schlosberg (1998).

either is, or is not preserved. Too often the implicit result is that the environmental considerations are assigned a value of zero.

This matters in two important ways. First, a lot of biodiversity is destroyed. Too little conservation takes place. Second, where the environment is damaged because there is no value assigned, there is no compensation. Compensation forgone is resources unavailable for conservation more generally. Without a compensation mechanism, there are few opportunities to use economic incentives, such as payments for ecosystem services and eco credits, to benefit species and habitat conservation. CBA provides a route to both better policy-making and to compensation. It is highly imperfect, but it is hard to think of a superior alternative, and not employing CBA has often led to costly mistakes.

CBA forces costs and benefits to be made explicit: qualitative expert evidence tends to be more amenable to use in lobbying and implicit influencing. It is also important to bear in mind that decisions by experts assume that experts are independent guardians of the public interest. But experts have careers, interests, and views of their own, and can be lobbied and influenced. Experts can be hired not just by those who seek to protect biodiversity, but by those who seek to damage it, too. Typically developers have deeper pockets, and hence are better able to muster 'expert evidence' in their favour.

Thus, despite all the caveats, economic valuation and CBA provide an important tool for the design of biodiversity policy. It is one way of characterizing biodiversity problems, and because of the problems of assigning values to non-market goods and services, the assumptions always need to be spelt out. This includes the information basis and the consequences of changes in information. Unfortunately in many cases, monetary valuations are stated without the caveats, especially by politicians and those with interests in the outcomes.

IV. Depletion, substitution, and renewable and non-renewable resources

The optimal amount of biodiversity is not, as some environmentalists claim, that level in some kind of idealized 'state of nature', before humans evolved and began to have their own impact on Earth's ecosystems. As noted above, humans have been exploiting other species and depleting natural resources for their entire history, and now there is no true wilderness left. Indeed, it is not clear that a concept of 'pristine nature'—nature without humans—is in any sense optimal, though much of the ecology and conservation literature takes nature without humans as its base line (and hence assumes it to be optimal).[15] The question is not whether to deplete natural resources, but by how much.

Two areas of resource economics are relevant here: the optimal rules for managing renewable and depleting non-renewable resources (biodiversity resources are primarily renewable resources but may have some non-renewable features); and the substitutability of natural and man-made capital. By defining rules for the use of natural resources, the concept of a sustainable growth rate can be formalized to meet the

[15] Willis *et al*. (2007) and Willis and Birks (2006).

constraint that welfare must not fall in any future period (Pezzey, 1992; Heal, 2012). Other definitions of sustainability impose the constraint that aggregate natural capital must not fall in any period or, even more specifically, that the stock of biodiversity assets is non-decreasing for all future periods. This is where the concept of natural capital comes in (Barbier, 2011).

Renewable resources are those that can restock themselves, provided that resource abstraction is limited and managed. Fisheries are one example. If humans harvest a limited amount of cod, for instance, stock levels of the species can be maintained. But if humans harvest cod excessively, the population may collapse. This is not dissimilar to other predator–prey relationships, in which some sort of balance is maintained in semi-stable systems.[16] The experience of perhaps the greatest cod fishery in the world—the Grand Banks—is an example of the latter.[17] Worse, as noted above, once a species is on a path to extinction, its value may rise, and this in turn provides incentives for even more rapid depletion—a vicious circle explored by Mason *et al*. (2012).

The optimal harvesting rate for a renewable resource is one that ensures that the rate of return from investing in other assets, the market interest rate, is equal to the rate of return from the renewable asset. This often leads to the prescription that, after some initial adjustment, stock levels should be kept constant so that the harvest rate matches the natural growth rate of the stock. The optimal depletion rate for a non-renewable resource rests on the same concept of arbitrage, but generates a very different conclusion. The resource is not going to last, so the issue is not whether to deplete, but how quickly and thus, by implication, which set of people should have the benefit. Hotelling (1931) identified that, under specific assumptions, the optimal extraction path implies that the price of the natural resource increases at the interest rate. These assumptions are reasonably strong, and Livernois (2009) shows that the empirical evidence does not provide overwhelming support for the (simple) rule; modifications (such as better accounting for technological progress in extraction costs) are needed.

An important assumption is that non-renewable resources can be swapped for other physical or financial assets. But how far can man-made capital substitute for natural capital before the returns on man-made capital start to decline? Consider a standard production function, which translates a series of inputs into output. Conventionally, neo-classical economics has two factor inputs, capital and labour. Capital is further disaggregated into human and non-human. In classical economics, there were three factors: land, labour, and capital. Land was subsumed under capital in standard models. However, some economic theory has incorporated resources and/or natural capital as an additional factor of production,[18] of which biodiversity is one example. Biodiversity is then an asset which yields a stream of (eco)services.

The way the factor inputs are separated out reflects differing views of the relationship between humans and the natural world. One—classical view—is that if natural

[16] The concept of 'the balance of nature' is a useful heuristic, but it has limited empirical support, since change is a permanent feature. Even in 'predator–prey' models, the empirical support for the classic Lotka–Volterra equations in ecology is weak. From an economist's perspective this is unfortunate since the concepts of equilibrium and modelling shocks is one that is familiar in economic theory.

[17] See Duncan *et al*. (2011) for background and modelling of the dynamics of the collapse.

[18] See Stiglitz (1974), Barbier (2011), and Hepburn and Bowen (2012).

capital is considered to be a factor of production, the natural environment constrains the possibilities of humans, and once humans move up against the constraints, a Malthusian-style feedback comes into play. Humans can only expand so far, before we run out of land, water, oil and gas, and so on. As biodiversity declines, these constraints become even tighter. Substitution of man-made capital for natural capital is feasible up to a point, but at a critical threshold we damage nature's ability to renew itself. The critical point can be termed the 'carrying capacity' of Earth, and in this view, it implies that the potential growth in the human population is ultimately limited, even if these limits may be in the more or less distant future.

Standard neoclassical models present a rather different view. Output growth is driven by technological progress, which improves the factor inputs of capital (human and non-human). We get better and better at making things. To facilitate this technical progress, we use up natural capital, but end up with much more non-natural capital: cities, infrastructures, goods, and services. We end up with fewer swallows, but more iPads. Constraints remain for non-renewable resources, but even here there is considerable optimism built into the standard neoclassical view. We can increasingly modify genetic material, creating new plants and animals as a result. It is not inconceivable that species could be recreated and that biodiversity, while altered, may be improved. Sequencing the human genome and the developments in the biosciences offer up new opportunities, and move the focus of biodiversity from the species to the genes themselves. In the plant world, it is possible to store seeds for very long periods, and one measure to protect biodiversity has been to create new seed depositories.

The Hartwick–Solow rule formalizes this view and the idea of a growth path based on the substitution between natural and man-made capital. It states that if the rents derived from the efficient extraction of a non-renewable resource are invested entirely in reproducible physical and human capital, and if there is a high degree of substitutability, or a sufficiently fast rate of technological progress, then non-declining, sustainable consumption through time is feasible. This relates to the concept of weak sustainability and the feasibility of such a condition depends very much upon the substitution possibilities open to an economy.

V. Policy instruments

Armed with the economic concepts of externalities and public goods, what policy approaches are available for maintaining biodiversity to maximize social welfare? Economics offers three broad approaches. First, we can employ 'economic instruments' to create incentives to correct biodiversity-related market failures and to ensure optimal provision of biodiversity-related public goods and resources. Economic instruments can be divided into price instruments (such as taxes on damaging behaviour, or subsidies for biodiversity provision), or quantity instruments (such as tradable permits), or some combination of the two. Second, we can regulate through 'command-and-control' (such as the specification of protected areas). Third, we can 'do nothing' on the assumption that the costs of intervention (government failure) are likely to be greater than the costs of the market failure. The third approach is the default, and biodiversity has been suffering.

(i) Biodiversity-related externalities

The market underprovides biodiversity because there are positive externalities—the benefits are not entirely captured by the actor providing biodiversity. Similarly, the market overprovides goods that damage biodiversity because there are negative externalities—the full costs of pollution, fertilizer, pesticides, land conversion, and so on are not borne by the relevant actor. If something is to be done about these biodiversity-related externalities, the choice is between economic instruments or command-and-control.

The choice of approach for biodiversity externalities is related to a much more fundamental topic of economic thought. Debates between the merits of market and planned economies occurred at length in the 1930s. Claims of socialist planners (such as Lange) were pitted against market advocates (such as Hayek). The general superiority of markets, compared to central planning, arises because of the ways in which incentives and information are economized in markets, compared with the computation demands placed upon planners. In a market-based economy, individuals and firms make decentralized decisions based on the vector of prices, which emerge from the many decentralized decisions. In contrast, the planner needs to know all the production and utility functions in order to optimize. Incentives differ too: in markets, individuals and firms pursue utility and profit maximization; in planning, the social welfare function has to be derived from individual utility preference orderings, and the bureaucratic incentives to seek out rents need to be taken into account.

It is for these fundamental reasons that economists often start by looking for 'economic instruments' that correct prices and take advantage of markets, before turning to command-and-control. For biodiversity policy, however, the use of economic instruments and markets turns out to be very challenging. In fact, for understandable reasons, most policy is command-and-control, as Daniela Miteva, Subhrendu Pattanayak, and Paul Ferraro point out in this issue (Miteva *et al*., 2012). This is not to suggest that 'economic instruments' do not have potential, and recent contributions to the policy literature have suggested that more enhanced roles for markets are worth considering.

As noted above, there are two broad categories of 'economic instruments'. The first focuses directly on the prices faced by agents who are degrading biodiversity. A direct price can be established by taxing activities that cause biodiversity loss, or by establishing subsidy payments for ecosystem services (PES). Subsidizing ecosystem services is conceptually distinct from subsidizing biodiversity. Nevertheless, Polasky *et al*. (2012), in their study of conservation funding in Minnesota, USA, find a high degree of alignment between strategies that target the value of ecosystem services and those that target habitat for biodiversity conservation. The appropriate price level has to be estimated, using the valuation techniques discussed above. For instance, the policy-maker might identify an externality (say the damage to bees caused by pesticides and herbicides), conduct a monetary valuation study, and then either impose a tax (in this example on pesticides and herbicides) or grant a subsidy for the under-produced service (in this case, beekeeping). In some cases, the instrument is applied to the cause of the biodiversity loss; in others, it is directed at the consequences. Causes of biodiversity loss are varied, but include agricultural chemicals, conversion and development of land which had supported wildlife, river pollutants, and waste products. Providers of biodiversity can also be supported by a range of measures, including an array of subsidies, conservation auctions, and conservation easements, which provide economic incentives for landowners to conserve biodiversity (see the paper in this issue by Nick Hanley, Simanti Banerjee, Gareth Lennox, and Paul Armsworth (2012)). The so-called set-aside policy, which provides land free from cultivation at field margins, can support biodiversity policy, even if the underlying motive may be primarily to influence agricultural output. Other environmental schemes can subsidize particular farming practices that encourage biodiversity (Natural England, 2009).

A second way of creating appropriate economic incentives is to create new markets. This involves directly fixing the quantity of the externality, and allowing the market to determine the price. For instance, the quantity of pesticide could be fixed, and those wishing to use this chemical would have to apply for (or buy) permits for pesticide use, which could then be traded. The market price for permits is the level at which demand for permits (from agents) equals the supply (set by government). Where the damage is great, the chemicals might be banned, but in many cases, the optimal quantity of pollution is not zero. A variant of this approach is to require developers to purchase eco-credits to offset the impact of their development on biodiversity. Specific harms can be identified, their values estimated through valuation techniques. Developers are then required to purchase eco-credits, generated from biodiversity protection activities, of the same valuation before the development can proceed.

Such economic instruments have several potential drawbacks compared with ‘command-and-control’. First, generalized economic incentives or trading schemes may result in problematic hotspots (Stavins, 2003). This is because biodiversity tends to be highly location-specific, and because the impact of policies to protect biodiversity (e.g. deforestation policies) is also likely to be a function of location. Furthermore, incentives to deforest land and destroy biodiversity vary dramatically from one location to another (see Pfaff and Robalino, 2012, this issue). Location-specific pricing and/or regulation may be required. Unlike climate change, where it does not much matter where carbon dioxide is emitted, spatial location matters enormously to biodiversity and ecosystems. Trading between ecosystems can be a recipe for disaster. However, command-and-control regulation does not completely avoid the location-specific issues either: it simply places these quantity choices (and therefore the implied price) in the hands of a regulator, who has to devise estimates of the location-specific costs and benefits. Location-specific direct regulations may also be preferred because otherwise trading volumes would be too slim for a market to work.[19] Nitrates, for instance, might be banned from use in certain sensitive environments, but be subject to taxes everywhere else. Second, as discussed below, economic instruments alone may not work for biodiversity protection when the biodiversity in question is a public good.

The choice between fixing the price and fixing the quantities—taxes and subsidies versus permits—depends upon a wide range of factors. One factor often cited is efficiency under uncertainty, which depends upon how rapidly the marginal benefits and marginal costs of the activity change as more of that activity occurs (Weitzman, 1974; Hepburn, 2006). Under uncertainty, the choice of instrument, loosely speaking, turns

[19] An interesting case study here is proposals to price the abstraction and use of water (Environment Agency and Ofwat, 2011).

upon what we are most worried about. If it is critical that a certain threshold number of a particular species is preserved, then quantity rather than price instruments are likely to be more efficient under uncertainty. In contrast, if the costs of protection could skyrocket if a particular target turns out to be too stringent, it may be more efficient to fix the price. However, there are a whole host of other factors that need to be considered, not least the administrative feasibility and the politics of the different instruments (Hepburn, 2006). For instance, a major drawback of market-based mechanisms compared with taxes or command-and-control is that constructing artificial markets can involve developing a complicated set of market institutions. Governments may find this much more difficult to manage, especially in developing countries, than, say, simply creating and enforcing a protected area, or sending out the tax collectors.

So, empirically, which instruments work best for biodiversity? Unfortunately, Miteva *et al*. (2012) find that the evidence is too weak to draw clear conclusions. There remains a dire need to evaluate properly the different performance of biodiversity conservation approaches—there simply is not credible empirical evidence of what works and where. Miteva *et al*. find that protected areas do consistently stimulate modest changes in land use that may positively affect biodiversity. Despite billions invested in protecting ecosystems and biodiversity, however, the evidence base for economic instruments and other interventions is simply too shallow to say anything meaningful about whether they are superior to protected areas. Until economic instruments and market-based policies are tested in a manner that allows their subsequent evaluation, it will remain difficult to identify general rules about optimal biodiversity policy.

(ii) Biodiversity-related public goods

So far we have considered policy for protecting biodiversity through the lens of market externalities, focusing on correcting or creating markets, or regulation to require appropriate action of market actors. But, in many cases, characteristics of biodiversity suggest that it can also be viewed as a public good. For instance, the 'existence value' of biodiversity is non-rival—one individual's enjoyment of the existence of a species does not affect another's enjoyment of the existence of the species. Goods with such characteristics provide additional problems for the use of market-based instruments. For instance, even if a competitive market could be constructed, because the marginal costs of provision are zero, in a competitive market the marginal price would also be zero. A market with a price of zero obviously does not create any incentive to invest in the provision of the good.

Much biodiversity has multiple public-good characteristics. This constrains the policy instrument choice. For instance, the Amazon rainforest, the Snowdonia National Park, and sites of special scientific interest (SSSIs) are not obviously amenable to a simple-minded application of economic instruments for externalities, whether taxes, subsidies, or permits (though these instruments may help, and may indirectly provide a source of funding). There are two main options for public-good provision: the state provides the public good for free and recovers the fixed and sunk costs through general taxation; or the public good is turned into a club good, by giving some entity a monopoly right, and the legal power to exclude non-members.

Public ownership plays a key role through national parks and preserved areas on government-owned land. What happens within such parks tends to be command-and-control, though in principle the owner can create incentives, and economy-wide incentives may cover the domain within which the park is located. These public goods can also be provided by non-government charitable institution, such as large-scale environmental organizations. In the UK, the Royal Society for the Protection of Birds (RSPB) has over one million members whose subscriptions fund the 'club' and whose reserves are sometimes made open to the public, but are usually for members only. The National Trust has 3.8m members, has a mix of open and restricted access, and similarly uses a membership fee. Wildlife Trusts are more local in their areas, but again use a mix of funding mechanisms. Together, these non-governmental institutions own a significant amount of land.

Many of these considerations apply in a context in which property rights and the rule of law are generally part of the institutional architecture. In contrast, in developing countries—where much biodiversity is concentrated—the circumstances are less amenable to the use of market instruments. But again resorting to command-and-control does not necessarily *solve* the resource-allocation problems and, in practice, what matters is the empirical evidence in particular circumstances, as Miteva *et al*. (2012) emphasize in this issue. This is an area where more research is urgently needed.

VI. Treaties, targets, international agreements, and institutions

Many biodiversity problems are international, in one of two senses. First, they may be transboundary, such that the causes and solutions involve at least two countries. Migrating species often cross national boundaries, and certain habitats provide biodiversity of potential or actual use to populations beyond national boundaries. The great African migrations of large herbivores are iconic examples, focused not just in the Serengeti, but also the Okavango Delta. Fencing, notably in respect of the Okavango, has major implications for these species. Bird migrations do not respect national boundaries, and the open seas are beyond national jurisdictions.

Second, some biodiversity problems are global, in that the ecosystem concerned provides a global public good with effects on humanity everywhere on Earth. Biodiversity that provides global public goods tends to be concentrated in so-called hotspots. The tropical rainforests are disproportionately important in terms of species densities. Some ecosystems, such as the Amazon, are genuinely global in significance—their services affect the entire climate of Earth. Even smaller, less significant ecosystems within national boundaries may have international significance if they appear in people's preferences. The international feature of biodiversity potentially makes valuation exercises more complicated, since the domain of preferences and the number of people potentially included is very wide.

International public goods introduce a number of issues: the design of international treaties, the bargaining between nation states, the relation between climate change initiatives and biodiversity, the potential trade-offs and connections between poverty reduction and biodiversity, and the design of institutions.

Several major international treaties and initiatives relate to biodiversity. The Convention on Biological Diversity (CBD), which entered into force in 1993, is the most significant international agreement on biodiversity (see the article by Tim Swanson and Ben Groom (2012) in this issue). The CBD establishes that developed nations provide financial resources to support developing nations to meet the incremental costs of protecting biodiversity as required under the Convention. But the financial flows, channelled through the Global Environment Facility, are tiny compared to the value of the natural resources at stake. Swanson and Groom consider these payments in the context of a bargain, or game, between rich and poor countries. They note that the rich tend to offer to pay the poor the 'incremental costs' of protecting biodiversity. But this is an extreme negotiation outcome, in which all the economic surplus is captured by the rich world, and none by the poor. Swanson and Groom argue that this outcome cannot serve as an equilibrium based on narrow national self-interest (a Nash equilibrium), and identify conditions in which both threatened and actual destruction of biological resources by developing countries would be expected to be observed and, indeed, is observed. In short, the CBD is unlikely to provide adequate protection of global natural infrastructure because financial flows are too low and the underpinning concept of incremental cost is not a Nash equilibrium—even if the informational and enforcement problems could be overcome.

The CBD is complemented by other international agreements. The Convention on the International Trade in Endangered Species (CITES) was drawn up in 1973, coming into effect in 1975, and was designed to limit international trade in wild animals and plants threatened with extinction. Other international agreements also impact indirectly on biodiversity. Of these, those whose primary focus is climate change are perhaps the most important. The Reducing Emissions from Deforestation and Degradation (REDD+) scheme under the UN Framework Convention on Climate Change (UNFCCC) is the main vehicle, which was also the focus of discussions relating to the CBD in Nagoya, Japan, in 2010. REDD+ relates to biodiversity in two ways: reducing emissions limits climate change, which in turn protects biodiversity; and protecting key habitats, such as rainforests, typically (but not always) protects biodiversity while limiting emissions.

Considering these international dimensions as public goods provides a basis for considering the extent to which international agreements and treaties may 'solve' biodiversity problems. Scott Barrett categorizes global public goods according to: whether they can be provided unilaterally or by small group of countries; where they depend upon the weakest link; or where they depend upon the combined efforts of all states (Barrett, 2007). This classification is helpful in the biodiversity case. Preserving the American bison can be solved by the United States, and the snow goose depends upon Canada plus the US. Some migratory species depend upon the weakest link: for example, rare breeding birds in the UK are vulnerable to key countries on their migratory routes. Preserving the Amazon rainforest relies upon a number of countries, and is so big as to demand global cooperation to preserve it. Thus, although there is a good case for an overarching international biodiversity framework agreement, different levels of international cooperation are needed for specific cases. Again biodiversity turns out to be much more complicated than climate change.

Biodiversity and climate change are examples of international problems, requiring international solutions. But they arise in a context of many competing international issues and priorities. Other policy goals may conflict with the protection of biodiversity. Indeed, this is why biodiversity tends to suffer when rapid development takes place. Rivers get polluted, forests cut down, air quality declines, and agriculture takes in more land. The trade-off between biodiversity and poverty and between biodiversity policy and poverty is explored by Palmer and di Falco (2012) in this issue.

The nexus between biodiversity and poverty creates particular challenges. Many of the areas rich in biodiversity are in poor countries. Economic growth is associated with the destruction of biodiversity, but so, too, are poverty traps—pressures (perhaps created by population growth) can lead to environmental degradation and biodiversity loss, intensifying poverty, and increasing pressure on the household to meet its subsistence needs, leading to exacerbated degradation and biodiversity losses.

Global and regional agreements require monitoring and enforcement, and supporting research capabilities. Coming to an international agreement depends upon the creation and sustaining of credible institutions. In a number of global public goods cases this has been recognized with mixed success. The UNFCC provided the institutional framework within which the Kyoto Protocol was established. The World Health Organization (WHO) binds those who agree to specific regulations. The diversity and fragmented nature of biodiversity problems makes an overarching institutional framework both hard to construct and difficult to sustain, as was demonstrated at the Rio+20 conference in June 2012. The result has been to focus on regional mechanisms, which will reflect Scott Barrett's classification of public good problems noted above.

Less recognized has been the implications of the application of the theories of bureaucracies and government failure to these institutional examples. Non-governmental organizations (NGOs) pursue limited objectives, and have membership to maximize. It is a notable feature of the biodiversity NGOs that they tend to specialize in one aspect. In the UK, the RSPB looks after birds; Plantlife looks after plants; Buglife looks after insects; and the National Trust focuses on landscapes (and buildings). Within the domain of these environmental groups, there are many conflicts and disputes, and often they fail to cooperate to exploit opportunities for biodiversity in general. This is all the more surprising given that their memberships overlap considerably. Then there are campaign groups—such as WWF, the Sierra Club, Friends of the Earth (FoE), and Greenpeace. Campaign groups campaign, and this requires specific rather than general objectives. Indeed, it is interesting to note that for FoE and Greenpeace it is their anti-nuclear activities which gain the most attention.[20] International conferences—such as Durban on Climate Change and Rio +20 on biodiversity—are important recruiting opportunities for NGOs, who are provided with extensive media coverage.

The result of this institutional fragmentation has been that the political impact of the environmental movement has become less than the sum of its parts. When compared with other large-scale membership organizations, such as trade unions, the contrast

[20] Greenpeace started life as a Canadian pacifist Quaker Group opposed to nuclear weapons and nuclear weapons testing. The 'green' objective was added later, focused on the very vivid images of seals being culled on the Canadian ice.

is stark. Unions have a major impact on governments, companies, and society more generally. Unions sponsor MPs; green NGOs do not. This area of institutional analysis is grossly under-researched—especially its implications for the design of policy instruments and institutions.

VII. Implementation, accounting, and economic policy

In much conventional policy discussion, considerations of biodiversity and the environment are treated as an 'add on'. Once the conventional micro- and macroeconomic problems have been addressed, then the consequences for the environment are considered. An alternative approach is to consider all externalities and public goods on a common basis with all the other goods and services in the economy.

The starting point is to address national income accounts and, in particular, GDP. If the objective of economic policy is to maximize GDP, it will exclude important elements of social welfare (Arrow *et al*., 2004; Helliwell *et al*., 2012). Consideration has been given to the wider, non-market sources of utility in so-called 'happiness' measures.[21] But what limits the relevance of GDP is that it is a cash-based measure, with no balance sheet. An increase in GDP takes no direct account of assets and liabilities, and changes in their values. There is no allowance for capital maintenance or provision for future liabilities.

This is beginning to change, with substantial efforts by the World Bank (2006) and others.[22] In the UK, 'Whole of Government Accounts' (HM Treasury, 2011) have been introduced. These include future pension liabilities. What remains is to include infrastructure, both physical and social, and human capital. Environmental assets are part of that infrastructure, with natural capital considered alongside the other forms of capital. The establishment of the UK Natural Capital Committee is one step towards rectifying this situation.

Valuing natural capital at the net present value of the stream of ecosystem services is in its infancy. The UK National Ecosystem Assessment (UK NEA, 2011) is a tentative step in this process, but what remains is to add the natural assets on a case-by-case basis to the balance sheet. While such an exercise is complex and requires often crude approximations, and will inevitably be built up gradually, it goes in the right direction: it is better to be approximately right, than precisely wrong.

As natural capital is added to the balance sheet, it can be used not just to consider whether and to what extent biodiversity is being preserved, but also to estimate the required capital maintenance as a charge on current spending. We can treat natural assets as 'assets-in-perpetuity' rather than assets which can be depreciated. We want to pass them on to the next generation, to meet a sustainability criterion. The capital

[21] See Layard (2006), and Stiglitz *et al*. (2009). Further, it is argued that the Aristotelian objective of *eudaimonia*, or human flourishing (Oswald, 1962), is broader and more fundamental than mere happiness (e.g. Sen, 1993).

[22] See Pearce *et al*. (1996), Hamilton and Clemens (1999), Arrow *et al*. (2004), World Bank (2006), Dasgupta (2010), and Arrow *et al*. (2012), among others. The World Bank has extended its work with the Wealth Accounting and the Valuation of Ecosystem Services (WAVES) partnership.

maintenance is the sum required to maintain the assets intact, or at least to maintain the value of the service delivered by those assets.

The final step is to check whether the capital maintenance is being met by current spending on natural assets, either directly by government or by appropriate taxes, subsidies, and permit schemes, as described in section VI above.

VIII. Conclusions

The maintenance of biodiversity is one of the most complicated resource-allocation problems. Biodiversity is heterogeneous, arises at a number of different levels, creating multiple externality and public goods problems, and it almost always poses problems in contexts where there are other multiple market failures, too. The tools of economics can help—but they are primitive, in addressing systems properties, over long time periods, within which there is no assumption of stable equilibria. There is no 'balance of nature', against which to define optimal equilibria.

Yet the economic toolbox is not empty. Useful tools to hand include: the concepts of public goods and externalities; cost–benefit analysis and valuation; renewable and non-renewable resource management; substitution rules for sustainable development; policy analysis of market-based incentives and regulation; game theory for agreements; and institutional design.

Given the rapid rate of extinction and the collapse of ecosystems on the one hand, and the failures of the main policy instruments and institutions on the other, the scope for policy improvement is enormous. Though the economic tools are imperfect, they are being developed and refined. The practical application of economics to the world's most important resource allocation problem is long overdue.

In order to make progress, the first step is to fully incorporate the natural environment into the economic calculations, and into the core of government accounts. Natural capital needs to be s*et al*ongside conventional capital, human capital, and labour, extending the work of Kirk Hamilton[23] and the WAVES partnership, funded by the United Kingdom, Japan, and Norway.[24] Such an integrating approach would necessarily overcome the current, all too frequent, assumption of a zero value for natural assets, and requires valuation techniques to be applied.

Once environmental assets are incorporated into national accounts, the next step is to set intergenerational rules. The good news is that the theory has been developing over the last 1–2 decades, and the application of intergenerational policy has already begun for the climate, with carbon prices gradually emerging and being incorporated into economic policy. Mainstreaming natural capital is required—and with it the mainstreaming of biodiversity. This, in turn, requires integrating economic analysis into biodiversity policy—and incorporating biodiversity, and natural capital more generally, into the core of economics.

[23] See Pearce *et al*. (1996) and Hamilton and Clemens (1999).
[24] See http://www.wavespartnership.org.

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