Environ Resource Econ (2007) 37:313–333
DOI 10.1007/s10640-007-9118-3

Do we really care about Biodiversity?

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Published online: 16 May 2007

Abstract This paper addresses from an economic perspective the issue of global biodiversity conservation. It challenges the perception that the world really cares a great deal about biodiversity and is prepared to pay the full cost of maintaining this stock of natural capital. Despite the existence of a plethora of international agreements there still seems to be a global 'deficit of care' surrounding efforts to combat challenges such as those posed by global warming and biodiversity conservation. More light can be thrown on the degree of care by measuring both the actual expenditures and the stated willingness to pay for biodiversity conservation. However, actual expenditures are much lower than willingness to pay estimates recorded in the published literature. Using the criteria that the 'right' amount of conservation effort is one where the marginal economic benefits from conservation just equal the marginal costs of conservation, the paper explores the biodiversity conservation conundrum and concluded that, on the available evidence, the world does not care too much about this natural capital stock and bequests to future generations.

Keywords Biodiversity conservation · Conservation management costs · Willingness to pay for conservation · Ecosystem service · Cost benefit analysis

1 Introduction: the issue

The world community is allegedly very concerned about the fate of the world's biological diversity. Evidence for this concern arises from the ratification of various international treaties on biodiversity conservation. Among these are the truly global treaties: The International Convention for the Regulation of Whaling 1946 *et seq.*; The International Convention for the Protection of Birds 1950; The Convention on Wetlands of International Importance (the 'Ramsar Convention') 1971; The Convention on International Trade in Endangered Species ('CITES') 1973; Convention on the Conservation of Migratory Species of Wild Animals 1979; and the Convention on Biological Diversity 1992 (The 'CBD'). Details of all these

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Conventions and the various regional treaties can be found in Sands et al. (1994). Equally relevant treaties affecting biodiversity less directly are the Framework Convention on Climate Change (the 'FCCC') 1992 and its first Protocol, the Kyoto Protocol of 1997; and the Convention to Combat Desertification 1994. Barrett (2003) lists over three hundred international conventions relating to the environment in one form or the other. Barrett states that only one treaty 'offers a comprehensive approach to biodiversity conservation. This is the Biodiversity Convention' (Barrett 2003, p. 350).

But just how serious is the world in respect of biodiversity conservation? We argue that the only true indicators of concern must relate to action taken. Rhetoric about the fate of the world's environments is politically cheap unless the electorate calls the politician to account. Action tends to be costly, despite the claims of some that so much of what can be done is 'win win', i.e. will pay for itself. Moreover, we need to measure action carefully. It might seem that negotiating a treaty is a sign of firm action. But many commentators now doubt that the international efforts on conservation are effective or, at least, are as effective as is claimed. In some cases, the design of treaties has been criticised as addressing the wrong problem or a problem of lesser importance— see, for example, Hutton and Dickson (2000) on CITES. The more crucial issue is the counterfactual, i.e. what would have happened if the treaties did not exist? While no-one can be sure in every case, the available evidence is consistent with the suggestion that most treaties achieve little more than the counterfactual. Thus, despite the wealth of treaty-making and national laws embodying the treaty intentions, Barrett declares that 'most treaties fail to alter state behaviour appreciably' (Barrett 2003, p. xi). Similarly, Sandler (1997) argues that 'many international treaties concerning the environment have merely codified actions that the ratifiers had already accomplished or were soon to achieve' (Sandler 1997, p. 213). Elsewhere, we have suggested that there is a global 'deficit of care' to resolve global warming problems (Pearce 2003).

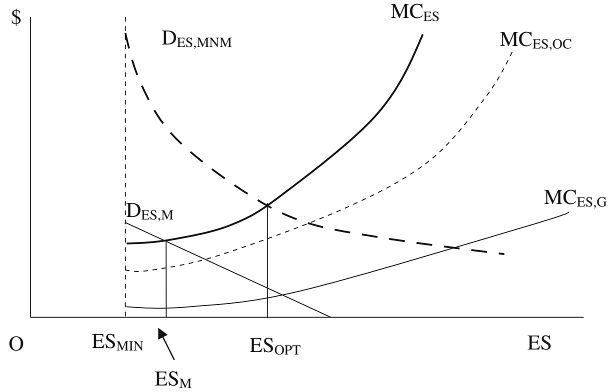
In this chapter we try to measure the degree of care by measuring action taken, using two economic indicators: actual expenditures and stated, or implied, willingness to pay for biodiversity conservation. In so doing, we also try to resolve an apparent conundrum. A recent and widely discussed literature has suggested that the world's willingness to pay for ecosystem conservation generally runs into many trillions of dollars, suggesting that the world does recognise the importance of ecosystem services and is willing to pay for them. But when we look at the actual expenditures on ecosystem conservation, they appear to be measured in, at best, a few billions of dollars annually. How can willingness to pay and actual payments differ by several orders of magnitude?

A prior question is why measuring the degree of care matters. One answer is that problems will not be solved unless we are aware of their true extent. If the world spends too little solving the biodiversity problem, but believes it is spending enough, then the problem will not be resolved and there will be no incentive to spend the right amount until, perhaps, it is too late. False beliefs about the adequacy of the global effort to save biodiversity are simply encouraged by political rhetoric. In turn, politicians have an incentive to say they are doing a lot, while doing little. The rhetoric may get them re-elected. Spending the 'right' amount of taxpayers' money, on the other hand, may get them deselected because the implied tax burdens might not be acceptable. In short, the political system has in-built incentives for the truth not to be told. Only by seeking some measure of 'conservation effort' can we call the politicians to account.

In what follows we make some simplifying assumptions.

First, we focus only on international conservation efforts. Partly this is because gathering the relevant data for biodiversity conservation efforts at the national level is too time-consuming. More importantly, the international focus is justified by the fact that a large part of the

Fig. 1 Stylised costs and benefits of ecosystem service provision



world's biological diversity resides in tropical countries where there are the least indigenous resources to conserve them. Hence, international flows of finance are one of the prime means of securing that conservation.

Second, we acknowledge that cash flows cannot be the whole story in terms of measuring the degree of care. Policy measures, for which associated cash flows may be difficult to identify and measure, will also affect conservation. But it might equally be argued that many policy measures actively encourage the destruction of biodiversity, e.g. the agricultural and industrial subsidy regimes employed primarily by rich countries. In other words, there may be as many 'bad' policies as there are 'good' ones from the standpoint of conservation.

Third, we will speak throughout of 'biodiversity' without dwelling too much on its definition. In its widest sense it refers to biological resources, whilst its proper sense would be confined to a measure of the diversity of those resources. Maximising the stock of biological resources is not the same as maximising diversity.

Fourth, we adopt the view that the 'right' amount of conservation effort is one where the marginal economic benefits from conservation just equal the marginal costs of conservation, i.e. the point where the net benefits of conservation are maximised. This is not a criterion of optimality that will appeal to many who do not like the economist's approach to these issues. But we argue this is an appropriate benchmark when trying to measure the degree of care since the economist's notions of costs and benefits relate directly to human preferences.

2 A diagrammatic construct

We begin with a diagram that tries to encapsulate the various flows of costs and benefits from biodiversity conservation. We will use the terms 'biodiversity conservation' and 'ecosystem conservation' interchangeably. We take an ecosystem to be broadly defined as 'a biotic community and its abiotic environment' (Krebs 1994, p. 12). All ecosystems generate flows of services to humankind, and hence all ecosystem services have an economic value. The issue is, just how large is this value?

Figure 1 shows the relevant constructs. On the vertical axis we measure economic value in dollars. On the horizontal axis we measure the flow of ecosystem services (ES) which we assume can be conflated into a single measure for purposes of exposition.

The first construct is a demand curve for ecosystem services $D_{ES,M}$. This is a demand curve for the *commercial*, or *marketed*, services of ecosystems, i.e. those services that have associated with already established markets in which formal exchange takes place using the medium of money. Thus, if we have an ecosystem producing timber or fuelwood or wildmeat, and, say, tourism, and if these products have markets, then the demand for these products would be shown by $D_{ES,M}$. Another name for a demand curve is a 'marginal willingness to pay' curve because the curve shows how much individuals are willing to pay for incremental amounts of the good in question, ES.

The second construct is another demand curve but this time for all ecosystem services, regardless of whether they currently have markets or not. This is $D_{ES,MNM}$ which is the demand curve for marketed (M) and non-marketed (NM) ecosystem services. There are various non-market services such as watershed protection, carbon sequestration and storage, scientific knowledge, the aesthetics of natural ecosystems, and so on. We know that $D_{ES,MNM}$ lies everywhere above $D_{ES,M}$. This is because, historically, ES have been abundant and hence there has been only a limited incentive for humans to establish property rights over them. As humans systematically expand their 'appropriation' of ecosystems, however, there is an incentive to establish property rights because ES become scarce relative to human demands on them. Humans have intervened in virtually all terrestrial ecosystems, especially as global population expands, appropriating around a third to a half of the net primary product (NPP) (Vitousek 1986;Vitousek et al. 1997).[1] Nonetheless, a vast array of ES is not marketed, so there is a gap between $D_{ES,MNM}$ and $D_{ES,M}$.

We need to consider the shape of the two demand curves shown in Fig. 1. Both are downward sloping, as we would expect. The more ES there are the less humans are likely to value an *additional unit* of ES. We have no reason to suppose that ES are any different in this respect to other goods and services: they should obey the 'law of demand'. But notice what happens if we have a very low level of ES. Imagine a world with very few forests, very little unpolluted oceans, a much reduced stock of coral reefs, an atmosphere with a very much higher concentration of carbon dioxide and other greenhouse gases. In the limit, if there were no unpolluted oceans, no forests, extremely high concentrations of greenhouse gases, then the willingness to pay for one more unit of ES would be extremely high. Simply put, while a few may survive in some kind of artificial Earth bubble, humans would, by and large, disappear. For this reason, $D_{ES,MNM}$ bends sharply upwards as we go to points closer to the origin on the horizontal axis. Essentially, $D_{ES,MNM}$ is unbounded. There is some irreducible minimum ES below which marginal WTP would rise dramatically. This irreducible minimum corresponds to the kinds of limits that ecologists and others have tried to define for, say; climate change. For example, O'Neill and Oppenheiner (2002) set this limit at +1°C for long run warming, well below the 'business as usual' level of warming.

Some suggest that at ES_{MIN} the demand curve would become infinitely elastic (for example, see Turner et al. 2003) But as long as it is a (marginal) willingness to pay curve, this cannot strictly be correct since incomes and wealth would still be bounded. It is technically more correct to say that there is no meaning to the notion of economic value in the unbounded area of Fig. 1.

As we shall see, this undefined region turns out to be rather important when we come to investigate the claims that the economic values attached to ES are extremely high.

In order to maintain ES of value to humans we know that certain costs are incurred. Figure 1 shows the first category of these costs as $MC_{ES,G}$—the marginal costs of managing ES. In the absence of a very strong evidence about the shape of $MC_{ES,G}$, we show it as a gently rising line. The second category of costs is of considerable importance and comprises the opportunity costs of providing ES. The assumption is that ES are best secured by

[1] Net primary production is the energy or carbon fixed in photosynthesis less the energy (or carbon) used up by plants in respiration. NPP is like a surplus or a net investment after depreciation (what is required for maintenance of function).

conserving the ecosystems that generate them. This is not consistent with using the ecosystem for some other purpose, e.g. agriculture. Hence, a potentially significant cost of having ES is the forgone profits (more technically, the forgone social value) of the alternative use of the ecosystem. We refer to this as $MC_{ES,OC}$—i.e. the marginal opportunity cost of ecosystem conservation. It is formally equivalent to the forgone net benefits of ecosystem conversion, i.e. 'development' as we tend to call it. The sum of $MC_{ES,G}$ and $MC_{ES,OC} = MC_{ES}$ gives us the overall marginal cost of conservation.

Figure 1 is obviously simplistic. For example, it ignores the possibility that ES might be largely maintained while serving some development function. Agro-forestry might be one example of this 'symbiotic' development. But, in general, we know that there is a long-run trend towards ecosystem conversion with the nature of the conversion meaning that many ES are lost. The diagram also ignores the possibility, realistic in practice, that the conversion process may be very inefficient. Ecosystems may be converted only for the development option not be realised because of mismanagement of the conversion process or of the subsequent development. In what follows we ignore these qualifications in order to focus on the basic messages from the analysis.

Finally, Fig. 1 shows us various points of interest.

First, since the true *aggregate* costs of maintaining a given level of ES are given by the area under the overall MC_{ES} curve,[2] and since the true global benefits of ES provision are given by the area under the $D_{ES,MNM}$ curve (assuming the demand curve shows 'true' willingness to pay—see later), the point ES_{OPT} shows the economically optimal level of ES provision.

Second, any point to the left of ES_{OPT} has benefits of ES (area under $D_{ES,MNM}$) greater than the overall costs of their supply. But all such points also have an interesting feature. Unless we arbitrarily confine attention to points between ES_{MIN} and ES_{OPT}, all points to the left of ES_{OPT} have either *infinite* total benefits or *undefined* total benefits, depending on how one wants to interpret the unbounded region.

Third, while $D_{ES,MNM}$ reflects the true global benefits of ES provision, it is not an 'operational' demand curve. This means that unless the WTP is captured by some form of market, or unless the evidence on WTP is used to formulate some quantitative restrictions on ecosystem conversion (bans, restrictions on type of conversion etc.), the demand curve that matters is curve $D_{ES,M}$. Figure 1 shows the real possibility that failure to reflect true WTP in actual markets results in a serious under-provision of ES. Here again we see the importance of the dual process of economic valuation (determining the location of $D_{ES,MNM}$) and capturing those values through forms of market creation.

3 Locating the current trend in ES provision

The next task is to gain some idea of where we might be in terms of Fig. 1. Clearly, without detailed knowledge of the cost and benefit functions, we cannot be sure. But some of the evidence suggests strongly that things are getting worse, not better.

3.1 Historical land conversion

First, we can be reasonably sure that we are moving leftwards in terms of the horizontal axis, provision of ES. The reason for this is that natural ecosystems have been converted to agriculture on a fairly systematic basis over very long periods of time. Indeed, the whole

[2] Total cost is the integral of the marginal cost curve. Total benefits are given by the integral of the MWTP curve, i.e. the demand curve.

history of humankind is a history of land conversion. Richards (1990) estimates that, between 1700 and 1980, the area of world forests and woodlands declined by 1.16×10^9 hectares, the area allocated to grassland and pasture stayed fairly constant, but that allocated to crops rose by 1.24×10^9 hectares. In short, cropland grew at the direct expense of forests and woodlands. One way of thinking about this process in terms of Fig. 1 is to regard the $MC_{ES,OC}$ curve as being shifted upwards over time as population expands and the demand for food increases. This process is consistent with the 'human appropriation' estimates of Vitousek et al. discussed earlier.

3.2 The extinction record

Second, while the evidence is more difficult to evaluate, many ecologists feel that species extinction rates are increasing. Ecologists usually determine whether extinction rates are high or not by comparing them with (a) past trends rates of extinctions over long periods of time and (b) past episodes of mass extinctions. Thus, Pimm et al. (1995) argue that the background trend rate of 'natural' extinctions, based on the geological record, is 0.1–1.0 extinctions per million species years.[3] They suggest that current extinction rates vary from 20 to 200 extinctions per million species-years, orders of magnitude higher than past extinctions. Dirzo and Raven (2003) argue that recent recorded extinctions seriously understate actual extinctions, largely because of sampling errors and concur with the longer-run estimates of Pimm et al. (1995). These estimates of extinctions are very uncertain, not least because there appears to be no consensus on just how many species there are in the first place, which is unsurprising when one considers the scale of the task that would be required to count them or infer their existence. Pimm et al. (1995) suggest a range of 10–100 million species. Stork (1999) suggests a more precise 'working' figure of 13 million species. Dirzo and Raven (2003) opt for ~7 million.

However, whereas the very long run estimates derive from geological records, many of the more dramatic predictions are derived from species-area relationships, themselves controversial and generally thought to exaggerate loss rates (Stork 1999).[4] Analysis of actual recorded extinctions suggests that around 0.24% of species have gone extinct since 1600 (Stork 1999). Such rates appear very low when compared with the more dramatic guesstimates using the other methods and especially the species-area relationship. Other evidence is consistent with the lower extinction rates (see e.g. van Kooten 1998).

The extinction record is thus still debated, but all that matters for current purposes is that a significant number of expert commentators believe that extinctions are increasing.

3.3 'Underfunding'

If we know that the maintenance of existing *managed* natural and semi-natural ecosystems is under-funded, then we might also conclude that there is a leftwards move along the ES axis in Fig. 1, at least as far as those ecosystems are concerned.[5] On the face of it, under-funding must mean that some ecosystems are not being maintained and hence must be being degraded. While there were extensive discussions about 'funding needs' at the Rio Earth Summit in

[3] Thus if on average species survive for one million year, natural extinction rates would be 1.0 extinctions per million species years (E/MSY). If average 'life' is 10 million years, the natural extinction rate would be 0.1 E/MSY.

[4] The species area-relationship takes the form $S = cA^z$ where S is species, A is area, and c and z are parameters. The value of z is usually taken to be around 0.3.

[5] Only a fraction of the world's natural and semi-natural ecosystems is managed.

1992, the first comprehensive efforts to secure some insight into this issue are the important papers by James et al. (1999) and Balmford et al. (2003). James et al. (1999) estimate that there is serious under-funding of existing Protected Areas, primarily, but not exclusively in developing countries. Expressing their estimated shortfall in expenditure as a fraction of the actual expenditure (management costs only) shows shortfalls of 10% in North America, developed Asia and Australasia. But shortfalls in Europe (somewhat surprisingly) are put at 50%, Sub-Saharan Africa and Russia/East Europe at 100%, 140% in Latin America, 450% in North Africa and the Middle East and over 500% in Asia. For the world as a whole, they put under-funding at around 40%. Overall, then, the picture suggests under-funding on a major scale in developing countries, supporting the notion that existing ecosystems—as indicated by Protected Areas—are declining if not in area then in quality. If the world's managed areas are declining in terms of funding requirements, it seems reasonable to suppose that most of the rest where, it will be recalled, human intervention is still dominant, will be in a worse state.

4 Reasons to be cheerful

Now consider some reasons why we might either be moving to the right in Fig. 1 or, if we are losing ES, why it may not matter and could even be net beneficial.

4.1 Optimal ecosystem loss

On the previous arguments, the record of:

- Historical land conversion
- Extinctions
- Under-funding of Protected Areas

suggests that we are moving leftwards along the horizontal axis of Fig. 1. However, such a finding, if correct, does not tell us if we are to the right of ES_{OPT} but moving left, or to the left of ES_{OPT} and moving left. It may be that what we are failing to conserve is what we should not be conserving anyway. Potentially, we have some reason to be optimistic. The parallel argument in terms of global warming has been quite widely advanced. There is indeed, say some of the commentators, increased global warming, but it does not matter very much—e.g. see Lindzen (1994) and the rate of return to alternative uses of the finance needed to combat global warming is higher—e.g. Lomborg (2004).

4.2 The property rights argument

One argument advanced by 'free market' thinkers is that, as the appropriation of Net Primary Product expands, so property rights to scarce ES will be established, markets will emerge, and conversion will take place only if benefits exceed costs. In terms of Figure 1, $\mathrm{D}_{ES,MNM}$ and $\mathrm{D}_{ES,M}$ will gradually converge and the optimal amount of ES will come about. The non-market services will gradually be 'captured' by market creation. There is a range of views within this argument, from extreme free market positions in which the state has no role and biodiversity should be 'privatised' through to those who acknowledge the potentially large role that the private sector already plays in ecosystem management. See Anderson and Hill (1995) and Drake (1995).

Evidence in favour of such a process lies in the same international treaties outlined earlier. The FCCC and the Kyoto Protocol, for example, are ways of converting essentially open access rights to the global atmosphere and its ecological services into common property rights, with access being partially limited by agreements, at least for developed economies. Policy instruments such as carbon and energy taxes, and tradable carbon quotas, are the means by which some of this market creation work. One problem with this view is that we have no guarantee this process will move fast enough to prevent serious loss of ES. One reason for doubt lies in the observation made above: it is not clear that such treaties do much to conserve ES (Pearce 2003; Barrett 2003, 2004). In other words, the common property provisions do not differ significantly in their effects compared to open access. On the other hand, evidence suggests that those communities that escape resource degradation and overcome open access poverty are those who win the 'race' between environmental degradation and institutional adaptation to resource scarcity (Lopez 1998).

4.3 Cornucopians

Every now and again, some commentators suggest that things are certainly getting no worse, and may be getting better. In the past, Simon (1981, 1986, 1995) and Simon and Kahn (1984) have suggested this, and in recent years Lomborg (1998) has echoed this view. Interestingly, Lomborg (1998) has attracted far more controversy than the earlier publications of Simon and Kahn, despite the fact that the messages, and the approach to the evidence, are very similar. Simon (1995), which is essentially an update of Simon and Kahn (1984), pays only limited attention to ecosystem protection and species loss, but did draw attention to the contrast between estimated rates of extinction of species and the recent historical record, which was noted earlier. Lomborg (1998) questions both the more alarming estimates of forest loss, including tropical forest loss, and the species extinction estimates. Part of the problem with all these contributions is that they tend to take the most pessimistic interpretations of ecosystem change and criticise them, a kind of 'straw man' approach. Nonetheless, despite the controversy, any analysis that addresses the data as best they can needs to be taken seriously.

4.4 The growth of protected areas

Perhaps a more substantive piece of 'good news' is the expansion in the world's protected area (PA) system. Yellowstone National Park in the USA was the first PA, established in 1872. Globally, protected areas did not reach 1 million km^2 until just after the Second World War. Since then the growth has been fairly dramatic until, today, they cover some 18.8 million km^2 (Chape et al. 2003). Between 1962 and 2003 the area grew from 2.4 million km^2 to 18.8 million km^2, or around 0.4 million km^2 per annum.[6] Of course, this is not 'new' land but existing land with its use at least nominally proscribed to prevent its conversion to some other use, hopefully with biodiversity preserved if not encouraged.

A comparison can be made with converted land, although data for land conversion remain the subject of controversy. Most converted land is forest. The Food and Agriculture Organisation (FAO) (2001) estimates global net rates of deforestation of around 9 million hectares in the 1990s, or 0.23% of total forest area. The World Resources Institute (Matthews 2001) disputes the figure, noting that FAO data include biodiversity-poor plantations as afforestation, offsetting natural forest loss. Net of plantation growth, annual losses are closer to 16 million hectares per annum, or 0.4% per annum of forest cover, nearly double the FAO figure.

[6] Note that the Greenland national Park, established in the 1970s, covers 97 million ha. and the Great Barrier Reef Marine Park, established in the 1980s, covers 34 million ha.

A direct comparison therefore suggests that the world is designating protected areas at a rate of 40 million hectares per annum, and deforesting land at perhaps 16 million hectares per annum, a net gain of 24 million hectares per annum.[7] If the world is protecting areas at a rate three to four times greater than rates of deforestation this must be good news for biodiversity conservation, and *prima facie* evidence that we care. But there are some caveats.

First, PAs under IUCN Management Categories I-VI account for only 10% of land in developing countries and 12% in developed countries (World Resources Institute 2003).

Second, well over one half of all protected areas occur in nations where governance is weak (World Resources Institute 2003). Weak governance shows up as poor management and neglect and, in many cases, corruption. Protected areas may therefore be 'paper parks', protected in name but not in reality (Whelan 1991). van Schaik et al. (1997) document the poor state of many tropical rain forest reserves, showing how a combination of lack of resources, lack of commitment, lack of knowledge and what they call 'resource theft' places many of them in peril. In many cases this is unsurprising. Government involvement in protected areas arises precisely because market forces do not dictate that protection is the most privately beneficial use of the land. But governments have no comparative advantage in managing land for biodiversity. Where there are conflicts between protection and the conversion uses of the land, therefore, government is likely to lose out, or to become involved in rent capture procedures that involve its surrender of conservation. Furthermore, the available data on expenditures on PAs suggests major under-funding, as we discuss shortly.

Third, there is a real possibility that the better protected areas are bio-diverse not because of protection effort but because the alternative use value of the land is low. The argument here is akin to that used by game theorists to explain why nations sign up to international agreements. Signing up is most likely when the nation in question has little or nothing to lose (Barrett, 2003). Similarly, protection is more likely when the opportunity costs of protection are low. If we surmise that existing PAs will have low opportunity cost relative to any new ones, and the evidence for this is considered shortly, then it may well be that existing PAs would not have been damaged in any event.

What this suggests is that publicly owned and managed protected areas will be at risk wherever there is a high private economic value to the alternative use of the land, e.g. for agriculture or forestry. Governments are then either not able to resist encroachment and conversion, because they lack resources, or will actively connive in the conversion if there are rents to be gained for select groups and individuals. Where there is a low opportunity cost to conservation, the land will appear 'protected' when in reality the gazetting of the area makes little difference to its biodiversity status. Conservation would have occurred anyway. However, private ownership may succeed where government ownership fails (e.g. Langholz et al. 2000; Langholz and Lassoie 2001).

Finally, the protected areas movement is not one that, so far, has been well informed by an explicit balancing of costs and benefits to the nation in question. But as the demand for alternative uses of the land grows, especially for agriculture and human settlement, so a questioning of the national worth of protection will occur. Already, some of the results of this reappraisal suggest that nations may be better off sacrificing their protected areas – see, for example, Norton-Griffiths and Southey (1995).

[7] Actually, since the forest loss figures are specific to the 1990s, the comparison should strictly be between the rate of PA formation and deforestation in that decade. The relevant figures would then be +59 m.ha – 16 m.ha = + 43 m.ha.

5 Costs and benefits

Historical land conversion rates, the extinction record, evidence of 'under-funding' tell us that we are moving leftwards along the horizontal axis of Fig. 1. The argument that things are at least getting no worse comes from the cornucopians, those who believe that property rights regimes react rapidly to changing scarcity, and the expansion of PAs. But none of arguments tell us whether we are at or to one side of the economic optimum level. That is, we do not know if the rates of loss are optimal or not. For that we need evidence on costs and benefits 'at the margin'.

6 The cost–benefit evidence: protected areas

If we had some idea of the likely costs and benefits of expanded protected ecosystems, we would have some evidence to locate us to the right or left of the point ES_{OPT} in Fig. 1. Essentially, if the costs of 'new' protection exceed the benefits then we are to the right of ES_{OPT} and if we have benefits in excess of costs then we are to the left of ES_{OPT}. The first piece of information needed concerns the costs.

James et al. (1999) looks at the world's protected areas and consider expenditures in current PAs and on an hypothetical expansion from the 13.2 million km^2in 1999 to 20.6 million km^2. Converting their estimates to annual per hectare costs the picture appears as follows:

Apart from the apparent reduction in global management costs as the PA area is expanded—they go down instead of up as might be expected—the picture is in keeping with Fig. 1. One would expect opportunity costs to rise significantly as more ecosystems are conserved. The reason for this is that the 'low cost' areas will tend to be protected first and, as the system is expanded, areas that have higher development potential will be converted. This prediction is also borne out by the later paper by Balmford et al. (2003) which shows that protection costs rise with an index of 'development'.

The World Bank (2002) have also costed the setting aside of just over 200 million hectares of *new* land in developing countries as protected forest areas. Their combined management plus opportunity cost estimates are very much higher than those in James et al. (1999) and amount to \$93 per ha.p.a, some eight times the James et al. estimates. Even allowing for the inclusion of some high cost land acquisitions, 70 per cent of the land hypothetically considered in the Bank calculations are acquired at an opportunity cost of less than \$50 per hectare. Whereas James et al. have opportunity costs of just under \$9 per ha. the World Bank has opportunity costs of \$83 and management costs of \$10 per ha. It is not clear why the estimate should diverge so much. However, we make use of both sets of estimates shortly.

The extensive literature on environmental economics as it relates to ecosystem protection tends to focus on individual case studies. Moreover, while we have many studies of willingness to pay to conserve individual species and some habitats, it is hard to come by estimates of willingness to pay to conserve *diversity* as such. Finally, even if the focus of studies is on species and habitats, we have few *meta-studies* on which to base any consensus judgement. There are several 'global' assessments of the value of biodiversity, most of which, unfortunately, rest on serious errors of analysis (for a critique see Pearce, 1988; Bockstael et al. 2000). While a full review of the evidence on willingness to pay has to take its turn for a later date, we briefly review some of the wider surveys of ecosystem values here, and also look at some estimates of global willingness to pay. Efforts to argue that the willingness to pay for ecosystem conservation outweigh the overall costs of conservation, based on individual case studies, can be found in Turner et al. (2003) and Balmford et al. (2002).

However, considerable caution is required in interpreting these reviews. First, published studies are more likely to report cases where benefits exceed costs, rather than vice versa: a 'censoring' effect is likely to be present. Second, some of the case studies utilise data from the illicit literature on ecosystem valuation such as that of Costanza et al. (1997). Another literature reports specific examples where markets have been created in an ecosystem's services, sometimes with the conclusion, stated or implied, that there are higher net benefits to ecosystem conservation than to the alternative 'development' option. This may be true, for example, of Costa Rica's Forest Law (Chomitz et al. 1998), although some authors express doubts about whether the Law passes a cost-benefit test.

Pearce (2003) and Pearce and Pearce (2001) survey the value of forest ecosystems. They conclude that the dominant economic value of forests lies in carbon storage and sequestration. Present values of \$360 to 2200 per hectare would more than compensate for many, though not all, conversion values for tropical forests. Genetic information for pharmaceuticals and agriculture probably has low per hectare value, perhaps a few dollars per hectare, although the debate on the appropriate procedures for valuing this information continues in the literature (for a review see Pearce et al. 2005, Ch.12). Watershed protection, at \$15 to 850 per hectare, and recreational values, from near zero to \$1000 per hectare for unique forest areas, can be significant, but critically dependent on location. Non-timber forest products tend to be modest in terms of economic value relative to conversion values, but can be high relative to local community incomes. Since it is the former that tend to dictate conversion decisions, non-timber product values are unlikely to protect most forest areas, contrary to some of the early euphoria attached to these benefits. The somewhat gloomy finding is that, unless carbon is 'monetised', the economic values of tropical forests do not, at the moment, compete with alternative uses of the land in many cases. Put another way, implicit willingness to pay is not revealing high levels of care. Mobilising carbon values could change this for many areas of forest provided international markets are allowed to develop fully in carbon storage and sequestration. So far, efforts to do this have been very modest, with international negotiators on climate agreements finding too many reasons why stored carbon should not be the subject of the various flexibility mechanisms.

In addition to the use and indirect use values of forests, several authors have attempted to use stated preference techniques to secure some kind of 'global non-use value' for tropical forests. Kramer and Mercer (1997) use contingent valuation surveys to elicit US citizens' willingness to pay to conserve an extra 5 per cent of the world's tropical forests (taking 5 per cent as being already conserved in one form or the other). Their results suggest an annual per hectare valuation of about \$4. Extended hypothetically to households in high income countries, the value would rise to around \$25 ha.p.a[8]. The Kramer and Mercer study uses a 'one-off' payments. Horton et al. (2003) use a parallel approach to UK and Italian willingness to pay to protect the existing 5 per cent of (under-) protected areas in Amazonia. But their results contrast starkly with those of Kramer and Mercer. For 'UK only' willingness to pay, the implied annual per hectare value is \$48, producing a fund of \$912 million *per annum*. If the UK result is extended to all industrialised countries the implied 'fund' amounts to \$26.8 billion, and the per hectare value rises to a staggering \$1400 per annum, well in excess of the capital value, let alone the rental value, of most Amazonian land. Horton et al. (2003) argue that willingness to pay exceeds the costs of protection (calculated here as about \$ 1 billion p.a. from the data in Horton et al.) 'by an enormous amount'. This would be true if the willingness to pay figures were credible. The authors themselves express doubts their

[8] The US has 91 million households. World high income countries have 580 million households. The UK has just under 20 million households.

reliability. As we see shortly, it is also very hard to square these willingness to pay estimates with actual flows of funds for ecosystem protection..

Efforts at some kind of meta-analysis have been made with respect to wetlands. Woodward and Wui (2001) review 39 studies and find values of \$2 to \$20,000 per hectare in 1990 prices, or say \$3 to \$30,000 in current prices. The average is around \$3000 per hectare. Brouwer et al. (1999) analyse contingent valuation studies only, and present results in willingness to pay per household per year, making the study non-comparable and less useful than the Woodward and Wui study. The average willingness to pay in the Brouwer et al. study is around \$40 per household per year in 1990 prices or, say, \$60 per household per year in current prices. In turn, this might suggest \$20 per person per year. If we imagine this sum was typical of all people over the age of 15 years of age and confine attention to the high income countries of the world only, it would translate to about \$14 billion per annum. However, as is well known, many economic valuation studies suffer from problems of aggregation across all goods. We cannot suppose that people would be willing to pay sums of this kind for wetlands conservation, plus another sum for tropical forests, and so on.

Overall, there are unquestionably contexts in which the inferred or stated willingness to pay for ecosystem conservation exceed the combined management costs of conservation plus the opportunity costs of conservation. But how far this is a general truth is open to serious question. Studies finding benefit-cost ratios greater than unity, and sometimes substantially greater than unity, may reflect 'censoring', the process whereby 'good news' is published and 'bad news' is not. Moreover, studies that do cost-benefit evaluations are often directed at ecosystems with fairly unique attributes. It is unwise to extrapolate from those studies to the far greater stock of ecosystems—this is perhaps one of the lessons of the literature on the value of genetic information in tropical forests. Finally, finding a willingness to pay is not the same as finding a value that can be captured and turned into cash flows. As is well known, only a fraction of overall willingness to pay, even when correctly estimated, can be converted to cash flows. Where this coefficient of capture is low, reliance has to be placed on decision-makers measuring and understanding non-market values and using the information to establish regulatory frameworks that prevent ecosystem conversion.

What we do we know about the relevant magnitudes of the costs and benefits of PAs? The answers for tropical forests appear to be that global WTP is anything from \$25 ha.p.a. (Kramer and Mercer) to \$1400 ha.p.a (Horton et al.). Protection costs range from \$7 -12 ha.p.a. (see Table 1) to \$93 ha.p.a. (World Bank). Given the doubts surrounding the extremely high value obtained by Horton et al. in geographically extending their WTP figures, the suspicion has to be that benefits do not automatically exceed costs, contrary to the optimistic findings in contributions such as Turner et al. (2003) and Balmford et al. (2002). This does not mean that those publications are reporting false case study conclusions, simply that their findings may be far from general. But this comparison of costs and benefits is obviously fraught with difficulties. Perhaps the best that can be said is that they neither support nor disprove the notion that more global conservation passes a cost-benefit test.

7 What do we actually spend on ecosystem conservation?

Willingness to pay studies are the only way in which we can secure some idea of the economic 'worth' of marginal or discrete amounts of ecosystem services. Hence studies that seek this magnitude are wholly legitimate. But willingness to pay and actual payments are not the same thing. Whilst not denying the value of estimating areas under discrete ranges of D_{ES} in Fig. 1, finding out what we actually spend on ecosystem conservation should give

Table 1 Estimates of protected area costs (after James et al. 1999)

Costs $ per ha. p.a.	Existing PAs	Extra PAs
Global costs		
Management	6.3	4.5
Opportunity costs	n.a	14.6
Total	n.a	19.1
LDC costs		
Management	2.8	2.8
Opportunity costs	3.8	8.8
Total	6.6	11.6

us a ‘reality check’. What follows is necessarily incomplete and constitutes a first attempt to estimate actual international flows of funds for ecosystem conservation.

7.1 Protected area costs

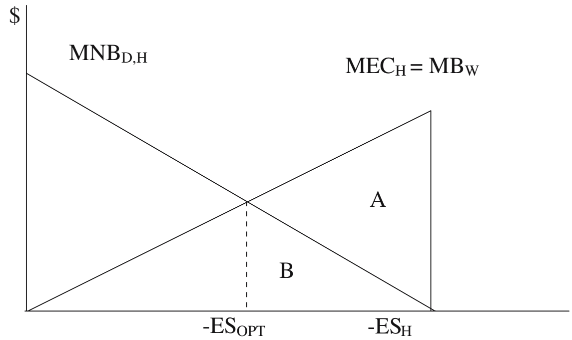
First, we recall the estimates of James et al. (1999) that *actual* protected area expenditure is some $6 billion p.a. James et al. note that this figure excludes compensation that many feel should have been paid to those who have been displaced or deterred from converting or using the PA land. They estimate that at a further $5 billion. However, while there is clearly a strong moral (and economic) case for making such compensation payments, they are not in fact made. Hence the annual sum of relevance is $6 billion.

7.2 Debt-for-nature swaps

Debt-for-nature swaps (DfNSs) are one form of debt-for-development swaps and involve the purchase, usually by an international conservation organisation, but also by governments and even individuals, of developing countries’ or transition countries’ debt in the secondary debt market. Such debt is often quite heavily discounted, i.e. the redemption price is well below the face value, due to the market’s realistic assessment of the prospects of repayment. In a DfNS, the purchaser of the secondary debt offers to give up the debt holding—usually by converting foreign exchange debt to domestic currency debt—in exchange for an undertaking by the debtor country government, usually through a local conservation NGO, to protect an environmentally important area, train conservationists, reduce pollution threats etc. Some of the most celebrated debt swaps involving governments and NGOs are those under the Enterprise for the Americas Initiative (EfAI), established in 1990. Another significant government player in DfNSs is Switzerland which set up a Swiss Debt Reduction Facility in 1991. DfNSs are clearly Coaseian bargains in which the indebted country has the property rights to a natural resource and accepts some attenuation of that right in exchange for payments by the beneficiaries of the resulting conservation. The involvement of at least the host government is necessary because rights are being attenuated and because issues of national sovereignty arise. But government involvement also helps reduce transactions costs. The involvement of lender governments is also clearly necessary where the debt is official debt.

A DfNS is an example of a Coaseian bargain (Coase 1960). Since the property rights to the environmental asset rest with the indebted country, and since the beneficiaries are environmentalists or the world as a whole, the beneficiaries are paying the host country not to convert the land in question or not to let it degrade. The essence of a Coaseian bargain is that the benefits derived from the payments made by the beneficiary must exceed the costs to the host country. It is the very fact that a bargain takes place at all that determines that a

Fig. 2 Beneficiary pays



cost-benefit test is passed. If this is right, then the continuing existence of DfNSs confirms that, for some ecosystems at least, the benefits of conservation exceed the costs of conservation. Fig. 2 shows this. In this case $MB_{D,H}$ shows the marginal benefits of land conversion, i.e. 'development', for the host country. MEC_H shows the global externality imposed by this conversion of the rest of the world, W, due to the loss of biodiversity. This can also be interpreted as the marginal benefit to the world of *not* converting the land to development. Exercising its sovereign property rights, H will go to $–ES_H$ where its profits from converting the land are maximised. But the optimum is at $–ES_S$ (note that the horizontal axis measures the loss in ES, hence –ES). It pays the world to offer H any sum less than MEC_H to prevent ecosystem loss, and it pays the host nation to accept any sum greater than MB_D. Clearly there are gains from trade in moving from $–ES_H$ to $–ES_S$, area (A+B)> areas (B). The move passes a cost-benefit test.

Adapting data in Sudo (2003) Pearce computes the total flows of funds 1987–2003 under DfNSs. These are summarised in Table 2. Sudo (2003) estimates the funds leveraged by DfNSs, i.e. additional sums that 'piggy back' on the actual swap. The leveraging ratio is 1.9 for the overall portfolio of funds. The figures in Table 2 correspond to an annual flow of some $140 million. In order to find the 'implied price' of a hectare of conservation, Pearce and Moran (1994), following on earlier work by Ruitenbeek (1992), analysed some of the early DfNSs where information is available on payments and land area. They suggest that an implicit price of, at most, $5 ha is being paid for the 'average' swap. One can therefore argue that DfNSs have a conservation cost of up to $5 ha.p.a. which is in keeping with the figures in Table 1. (Note that these are management costs—DfNSs appear typically to exclude land purchase).

7.3 The GEF

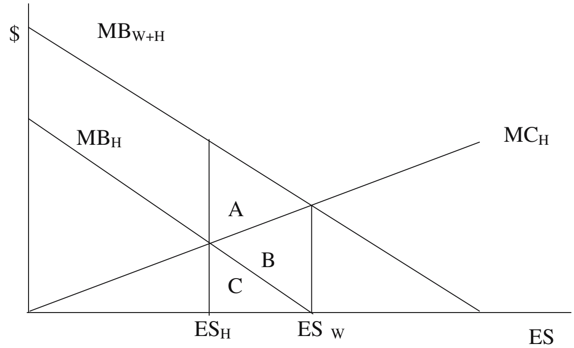
The Global Environment Facility is a United Nations agency charged with meeting the 'incremental cost' of developing countries' provision of global environmental goods. The definition of incremental cost is treated rather broadly but is intended to reflect the additional costs a developing country would face if it switched from an activity that is justified in domestic terms only to one that has both a domestic and global justification. As such, the GEF also fits the Coaseian model. Figure 3 illustrates. MB_{W+H} is the marginal benefit to the world plus the marginal benefit to the host nation of conservation. This time more conservation shows up as a move to the right along the horizontal axis. The marginal cost of conservation, MC_H reflects the management costs of conservation plus the opportunity costs, i.e. the forgone development benefits. If we first ignore the additional global benefits of conservation in the

Table 2 Debt-for-nature swaps—flow of funds 1987–2003

	Total face value of debt($ million, (rounded)	Total discounted value of debt ($ million, rounded)
Total excluding Poland (100 projects)	1,943	582
Total including Poland (102 projects)	4,840	1,153
Total including leverage	–	2,190

Source: Pearce (2004), Sudo (2003)

Fig. 3 Incremental costs and the GEF



host country, then the host nation will, if it optimises, go to ES_H. But the global optimum is at ES_{H+W}. The host nation has no incentive to go to the global optimum but will do so if it is compensated for the lost development benefits = area B+C. On the other hand, by going to the global optimum the host country secures some incremental benefit = area C. Hence there are two notions of incremental cost: gross incremental cost = area B+C and net incremental cost = area B. Whoever pays the residual element, area C, it is clear that gross benefit = area A+B+C exceeds incremental cost. A cost-benefit test is met.

The implementing agencies of the GEF were initially the World Bank, United Nations Environment Programme and the United Nations Development Programme, with various other agencies being given similar powers later on. The GEF was established in 1990 in a 'Pilot Phase', or GEF I, which lasted from 1991 to 1994, and its initial activities were unrelated to any international environmental conventions other than the Montreal Protocol on ozone layer depletion. Its coverage was biodiversity, climate change, ozone layer depletion and, curiously, 'international waters'—seas and lakes shared by two or more nations. The GEF soon took on the role of being the financing mechanism for the Framework Convention on Climate Change (1992), the Convention on Biological Diversity (1992), and the Stockholm Treaty on Persistent Organic Pollutants and the Convention to Combat Desertification.

Table 3 shows how much money the GEF has allocated to its various 'focal areas' between 1991 and 2002. The crucial role of co-financing is revealed. Co-financing refers to the leverage that GEF has on other funds outside the official Trust Fund.

Table 3 suggests that GEF funding has run at approximately $1 billion per annum. This certainly makes it the largest single source of market creation funding in the world. To facilitate comparison with other financial mechanisms, like has to be compared with like. Table 3 shows the comparison for biodiversity, although there are problems of separating

Table 3 GEF allocated funds and co-financing 1991–2002. $ million

	Climate change	Biodiversity	International waters	Ozone depletion	POPs	MFAs	Total
GEF	1409	1486	551	170	21	210	3847
Co-financing	5000	2000	n.a.	67	n.a	n.a	7067
Total	6409	3486	551	237	21	210	10914

Source: GEF allocations from GEF *Annual Reports*. Co-financing estimates from GEF (2002). Notes: MFAs = multi-focal areas such as land degradation. In 2002 land degradation was recognised as a separate focal area. POPs = persistent organic pollutants, approved as a focal area in 2001 and linked to the Stockholm Convention. Co-financing estimates for biodiversity and climate change are approximate and include expected sums. N.a = not available but assumed to be zero or close to zero

out the biodiversity component in GEF expenditures because biodiversity is often the beneficiary of non-biodiversity focal areas such as international waters. Table 3 shows some estimates for two other market creation activities— payments to forest landowners by the Costa Rican government to encourage conservation, and bioprospecting—the payment by drug companies and others for genetic prospecting rights. Overall, GEF expenditures on biodiversity conservation appear to run at about $315 m.p.a. but since a large part of 'international waters' expenditure is also biodiversity oriented, this sum could be raised to $365 million p.a. Finally, global warming control can also been seen in terms of protection of ES, in which case total ecosystem conservation expenditure rises to $950 m.p.a.

7.4 Bilateral assistance

Finally, we look at the available data on overseas development aid targeted at biodiversity. Under the OECD Creditor Reporting System (CRS) individual nations are meant to record biodiversity-related aid expenditures. OECD also records 'Rio Marker' expenditures, i.e. aid directed at achieving the Rio Earth Summit (1992) goals. These categories of expenditure tend to range much more widely than ecosystem conservation, for example including expenditures on water supply and agriculture. It is known that both sets of data are problematic (Lapham and Livermore 2003). Taking expenditures by the six main donors — USA, UK, France, Germany, Japan and the Netherlands, annual expenditures 1998–2000 were $38 million on the narrower definition and $240 million on the broader definition. An arbitrary compromise figure of $100 million is adopted here. Note that, whereas we have a prima facie case for the GEF and DfNS expenditures passing a cost-benefit test we cannot make any such presumption for bilateral assistance unless we can assume that assistance simply is not given without a cost-benefit analysis being carried out. This seems an unlikely assumption.

The bilateral aid figures also cast some light on the estimated willingness to pay figures in Horton et al. (2003). There it was suggested that the willingness to pay of UK citizens for conserving the 5 per cent existing protected areas of Amazonia would generate a fund of over $900 million p.a. Yet the OECD data suggest UK actual expenditure on overseas biodiversity aid at on-thirtieth of this sum, just $30 million p.a.

8 Conclusions of actual flows of funds

Table 4 summarises the previous discussion. It is important to note that the totals cannot be added. This is because bilateral, GEF and DfNS expenditures will overlap with PA expenditures – there will be some double counting.

Table 4 Summary of flows of biodiversity conservation funds $ million p.a.

Debt-for-nature swaps	Protected areas	Costa Rica Forest Law[a]	Bioprospecting[a]	GEF biodiversity	GEF all areas	Bilateral aid
140	6000	20	Small	315	1000	1000

Notes: [a] not discussed in this paper—see Pearce (2004)

The caveats to Table 4 are fairly obvious. First, the coverage is incomplete and excludes, for example, domestic expenditures on 'own' ecosystem conservation. This item will be significant for developed economies. Second, the figures exclude opportunity cost payments or estimates of opportunity costs, whether paid or not. The focus here is on actual flows of funds so the exclusion of opportunity costs is justified in one sense. On another view, it could be argued that the world has implicitly 'paid' the opportunity costs of conservation because it has sacrificed those costs by adopting ecosystem conservation — we saw that the World Bank estimated these at $20 billion for an *expansion* of the PA system.

Despite these caveats, the message that tends to emerge is twofold: (a) actual expenditures on international ecosystem conservation appear to be remarkably small, and (b) they bear no resemblance to the willingness to pay figures obtained in the various stated preference studies. At best, the world spends perhaps 10 billion dollars annually on ecosystem conservation. As others have noted— e.g. James et al. (2001)— these sums are trivial in relation to what the world actually spends on subsidising economic activity, perhaps $1 trillion p.a., 1000 times as much. Unfortunately, while such comparisons demonstrate the 'affordability' of a massively expanded ecosystem conservation programme, they also raise the complex question of why the world prefers to spend its money on subsidies that damage the environment rather than saving the environment in the first place. One might also contrast the ten billion dollars spent on conservation with the suggested 'required' budget for an effective system. James et al. (2001) put this budget at $300 billion p.a.: again, affordable but dramatically at odds with what we do.

9 Conclusions: what have we learned?

The purpose of this chapter has been to explore the extent to which political rhetoric on the importance of biodiversity conservation is backed by action. The rationale for trying to probe this issue is the suspicion that we are doing far too little whereas the rhetoric is suggesting we are doing a lot. If we are being misled— deliberately or inadvertently, in one sense it does not much matter which—then we may have a serious problem of misallocation of resources. We cannot find out unless we track down the flows of funds and relate them to willingness to pay. In short, what are the costs and benefits of global conservation?

The indicators of action we have chosen have been fairly narrowly defined as actual expenditures or willingness to pay. The paper quite deliberately excludes other indicators of the degree of concern for biodiversity. Overall, the contention is that conservation policy is not backed by financial commitments. Game theory teaches that many policies, and especially international policies that appear to transcend national interests, may achieve little when compared with the counterfactual.

The next problem is to define the costs and benefits of ecosystem conservation and to identify, if possible, where we currently stand. The costs are clearly the costs of ecosystem management plus the forgone opportunity costs of the 'development' that might otherwise take place. A simplifying assumption, that all ecosystem conversion is successful in developmental terms, was made. In practice we know that a significant fraction of such conversions result in neither development nor conservation. The benefits are measured by the economic value of ecosystem services as measured by the world's willingness to pay for them. It was noted that this willingness to pay is not a constant, will vary with the quantity and quality of the ecosystem service in question, and cannot be defined when some subset of ecosystem services goes below some threshold. We noted that some ecosystem services are already subject to market forces, but most are not. While the process of conferring property rights on currently 'open access' resources is developing, we have no guarantee that it will move fast enough to prevent serious degradation of ecosystem services. Indeed, we have significant evidence that this process of institutional change is moving all too slowly.

Various indicators were discussed which demonstrate that the dynamic process is very probably one of ecosystem service loss, not gain. Others might suggest a more optimistic story and that, at least, things are not getting any worse. Neither the pessimists nor the optimists, however, can tell us whether we are at, above or below some economic optimum in terms of ecosystem services. Most ecologists are probably convinced that we are well below the optimum and this view is no doubt shared by quite a few economists. But demonstrating this is hard since data that compare costs and benefits for the same thing are difficult to come by. Micro cost-benefit studies may well suffer from censoring—the studies that show net benefits get reported, those that show net costs do not. Moreover, some of the claims about citizen willingness to pay for ecosystem conservation look decidedly dubious when compared to actual expenditures. Needless to say, willingness to pay estimates *should* exceed actual expenditures (by the amount of consumer surplus), but it is hard to believe some of the reported differences. A comparison of the costs of protected areas and their benefits also raises some doubts, not about specific studies, but about the generality of the view that conservation benefits exceed conservation costs.

A partial picture of actual expenditures on conservation was produced by looking at protected area expenditures, bilateral biodiversity assistance, expenditures by the Global Environment Facility, and debt-for-nature swaps. For the GEF and DfNSs we have an a priori reason to assume benefits exceed costs: both are examples of Coaseian bargains and such bargains should not take place unless a cost-benefit test is met. The overall impression is that ecosystem conservation expenditures are to be measured in a few billions of dollars, certainly not hundreds of billions. Several apparent conundrums ensue.

First, why are actual expenditures so far below apparent willingness to pay? This question is not explored in any detail here. It could be that willingness to pay studies exaggerate real intent. It could be that the element of 'surplus' of willingness to pay over actual expenditure is huge. It could be that we are under-recording actual expenditures by significant amounts — certainly we have not tried to measure domestic expenditures on 'own' conservation. May be there are other explanations too.

The next conundrum arises from a comparison of actual expenditures (and, for that matter, willingness to pay estimates) and some of the more celebrated estimates of the economic value of the world's ecosystems. The former seem to be measured in billions of dollars, the latter in trillions of dollars. But studies that attempt to measure the *total* value of all ecosystem services are more than flawed — they are arbitrary. What they do is to take valuation techniques designed to value small (marginal) or discrete changes in ecosystem services and fallaciously apply them to the totality of systems. It is perhaps significant that these estimates are produced (in the main) by non-economists writing in science journals.

There is no pleasure in reporting the *suspicion* that despite all the rhetoric, the world does not care too much about biodiversity conservation. May be the efforts of economists and ecologists will force a change of policy in the future. But the proper place to begin is with an honest appraisal of just how little we do. Hopefully, others will show that we do more than suggested in this paper, or that we can have an expectation that a lot more will be done in the future.

Acknowledgements I am indebted to Cameron Hepburn of St Hugh's College, Oxford University, Stefano Pagiola of the World Bank, and Paul Jefferiss of the Royal Society for the Protection of Birds, Sandy, UK, for comments and suggestions on earlier versions of this chapter. Paul Jefferiss of RSPB, Andrew Balmford of Cambridge University and Kirk Hamilton of the World Bank kindly supplied some of the references used in the paper. Finally, I am indebted to the audiences at several seminars at Cambridge University, Oxford University, University of Gothenburg, Sweden, and University College London for comments.

Editors Note: This edited version is taken from a paper in the forthcoming volume *Biodiversity Economics: Principles, Methods and Application,* edited by Andreas Kontoleon, Unai Pascual, Timothy Swanson and to be published by Cambridge University Press. The Guest Editors are very grateful to Andreas as well as to Chris Harrison and CUP for permission to publish this version of the fuller paper within the present collection.

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