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Environmental market creation: saviour or oversell?*

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Abstract. In recent years considerable attention has been paid to the notion of 'market creation' for the conservation of environmental assets. Market creation establishes a market in the external benefit or cost in question (e.g. biodiversity or pollution reduction) and leaves the relevant parties to adjust their behaviour accordingly. While most attention has been paid to market creation through tradable permits and taxes (the 'polluter-pays'), it is less easy to secure a perspective on 'beneficiary-pays' initiatives. Both polluter-pays and beneficiary-pays initiatives are examples of modified Coaseian bargains in which governments intervene in the bargains to lower transactions costs, establish property rights, deal with public goods issues, or act on behalf of disadvantaged groups. This paper reviews four major initiatives in this respect – debt-for-nature swaps, bioprospecting and the Global Environment Facility at the global level, and the Costa Rican Forest Law at the local level. It finds that while there is much to applaud in initiatives in these new markets, serious questions remain about the modest flows of funds associated with such 'global bargains', and the extent to which they secure environmental improvements relative to the baseline of business-as-usual.

Keywords: Environmental market creation – Debt-for-nature swaps – Costa Rica – Forest Law – Bioprospecting – Global Environment Facility

JEL Classification: D49, D62, H41, O19, Q57, Q23

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1 Introduction

At the risk of some caricature of the arguments, environmental conservationists can be divided into two broad camps: (a) those who believe that conservation is best achieved through outright protection of habitats and species, sometimes characterised as ‘fence and forget’, or ‘command and control’; and (b) those who believe that, because conservation nearly always conflicts with some alternative use of the resource in question, or the base resource (land or water) in which it is located, conservation has to ‘pay its way’ by attracting money payments in excess of the cash flows associated with that alternative use. These payments need not come from a single source, or be for a single conservation benefit. Instead, whole packages of payments may be made for multiple conservation benefits. This is important, because a significant part of the literature has focused on specified benefits – e.g. carbon storage in forests – rather than on packages of payments for ‘bundles’ of benefits. The emphasis on single benefits is understandable – there are clearly transactions costs in bringing beneficiaries together, each with their own specific conservation goals.

In between the two extremes is any number of combinations of command-and-control conservation and market creation. Guyana’s Iwokrama forest enterprise, for example, combines traditional protected area controls with the sale of forest services such as carbon sequestration, fees for research ventures, wildlife viewing and so on (www.iwokrama.org). Other projects involve encouragement of sustainable use activities, such as agro-forestry, and payments to compensate for the net returns deficit agro-forestry incurs when compared to traditional clearance agriculture (Pearce and Mourato, 2004).

By and large, conservation philosophy up to the last two decades or so has been based on the command-and-control view, for example through calls for the establishment of national parks, ex-situ and in-situ breeding programmes and outright bans on trade in species or species’ products. The exceptions to this rule have certainly been notable, as with paying fees to private landowners for the right to hunt selected species, especially in some states of the USA (Davis, 1995) and privately managed commercial wildlife in Africa (Norton-Griffiths, 2003).

In the last decade or so, however, the second view has gained considerable attention. Sometimes dubbed the ‘market creation’ view, the idea is that those environmental resources that contribute to human wellbeing need to be bought and sold in a market place. Since, in a great many cases, markets currently do not exist in those resources, the market needs to be created through some institutional initiative usually involving one of: (a) the establishment of property rights where none previously existed; (b) the attenuation of prevailing property rights such that restrictions are placed on the use of land or species; and (c) the facilitation of bargains between resource right owners and beneficiaries where rights are already clearly defined. Approach (c) reduces to approach (b) in so far as the bargain imposes such conditionality on the resource owner. The early 1980s probably mark the emergence of this market creation view with the International Union for the Conservation of Nature (IUCN)’s *World Conservation Strategy* (IUCN et al., 1980), a significant document because it was life scientists who embraced the economist’s notion of sustainable use of a resource. Later documents from IUCN more openly espoused economic ideas about paying for conservation (e.g. McNeely, 1988; McNeely et al., 1990).

Market creation involves the marketing of currently non-market goods. The markets in question may be small and highly localised, or national, regional or global in nature. At the very local level, for example, downstream farmers may pay upstream forest owners to conserve the forests in order to conserve the ecosystem benefits generated by the forest – flood control, avoided sedimentation, windbreaks etc. In this case, property rights are historically determined and some form of government involvement is needed to overcome the transactions costs of bargaining. At the other extreme, open access global resources such as the atmosphere become the subject of a global common property regime which attenuates the rights of individual nations to treat the atmosphere as a limitless waste sink for greenhouse gases. Critical to the market creation notion is that this attenuated rights regime generates market exchange. In this case, the rights to emit greenhouse gases are traded through tradable gas emission permits or some form of emissions offset. The 1997 Kyoto Protocol to the 1992 Framework Convention on Climate Change is an example of this global market creation, enabling market creation via the 'Flexibility Mechanisms', which govern the rules for trading emission rights and credits.

The question is: what do we know about these market creation initiatives and are they more, or less, effective than the traditional conservation model? While experience in some of these new markets is still limited, enough exists for a reflective assessment on the role that they can play in overall environmental conservation. This is the purpose of this paper. Such an assessment is far from easy, mainly because there is no central database in which information about created markets is stored. In some cases there is detailed information, e.g. on carbon emission trades. In other cases, primarily those relating to habitat and species conservation, sources are widely scattered and vary substantially in the quality of information they provide. Moreover, there is some likelihood of information censoring. For example, failed experiments tend not to be reported at all and neither do adverse ex post evaluations. More likely, as noted later on, ex post evaluations of many market creation initiatives do not exist. The result is a probable biased sample of activities from which any evaluation can derive.

This paper is fairly selective in its coverage. The main guiding principle in selecting information has been to analyse what is fairly readily available information. Even then, to make the paper tractable (since the literature is huge) an important distinction is made between 'polluter pays' and 'beneficiary pays' markets. Emission trading would be an example of the former, since the emitter pays for emission rights. Downstream farmers paying upstream forest owners not to deforest, or paying countries to adopt more environment-friendly technologies or conservation, are examples of the latter. This paper deals only with 'beneficiary-pays' markets.[1] Sur-

[1] The distinction between polluter-pays and beneficiary-pays is not a hard and fast one. Quite a few corporations, for example, voluntarily engage in carbon-offset schemes whereby they purchase reductions in emissions elsewhere, equal to their own emissions. While the polluter – the corporation – is paying, it is also the probable beneficiary because of the resultant corporate 'green image', which may well raise corporate market value.

veys of some polluter-pays markets already exist – see, for example, Ellerman et al. (2003) and Tietenberg (2003) on US sulphur emissions trading and the implications for carbon trading; and the journal *Joint Implementation Quarterly* for experience with carbon trading linked to the Kyoto Protocol and earlier 'Joint Implementation' schemes. While polluter-pays market creation requires some regulation to be in place – an emissions target, for example – beneficiary-pays market creation may or may not be accompanied by a regulation. Paying for environmental services received may simply be an act of self-interest on the part of the beneficiary. As we shall see, some forms of beneficiary-pays market creation are accompanied by forms of regulation, precisely because beneficiaries and asset owners find bargains difficult to execute.

2 Sources of optimism about market solutions

2.1 Valuation studies

A separate problem in analysing beneficiary-pays market creation is one of distinguishing advocacy from analysis. The early literature tended to engage in optimistic oversell, albeit unwittingly. The prime example relates to biodiversity conservation through 'bioprospecting' – the process whereby firms collecting genetic material from the wilds pay directly for the material, which is subsequently screened for pharmaceutical, agricultural, cosmetic or industrial use. Works such as Norman Myers' *A Wealth of Wild Species* (Myers, 1983) had argued that there was a very large store of economic value to be found in genetic material in the wilds and this could be capitalised in the market place to increase the prospects for conservation. Later studies cast considerable doubt on the magnitudes of economic value involved, although the issue remains debated to this day (see Sect. 6 below). It is true, however, that Myers was one of the very first life scientists to extol the virtues of marketing sustainable wildlife utilisation as a means of capturing these allegedly high values (e.g. Myers, 1981), the forerunner of the more general idea that there is 'wealth in the wilds'.

Two issues arise: how large is the economic value that resides in nature? And how far can that economic value be captured and marketed? If the economic value is 'small' then created markets are likely to do little for conservation. If it is 'large', but the realistic prospects for market creation are small, conservation will still be limited. It may be, however, that capturable values are small but large enough to tip the balance towards conservation, or better conservation. There may be a predisposition to conserve but domestic resources may be limited: the addition of relatively modest sums through value capture could make the difference between a well-conserved habitat and a poorly conserved one.

The notion that huge economic values reside in nature was popularised in a widely quoted paper by Costanza and colleagues (Costanza et al., 1997). This estimated the economic value of world-wide ecosystems and their functions to be \$33 trillion on the basis of extrapolating willingness to pay measures found in the economics literature, a figure actually in excess of world GNP at the time the article was written, which itself should have cast serious doubt on the worth of the figure. Unfortunately, as several critiques pointed out (e.g. Pearce, 1998), this 'value of everything' figure is meaningless since it is applied to the total stock of environmental assets, ignoring the fact that removal of any one of the major ecosystems studied (e.g. oceans) would render all life-forms extinct. The mistake lies in the use of an essentially marginal concept – willingness to pay for small or discrete changes in something – to value total global stocks. Despite the critiques, the Costanza et al. (1997) paper is still widely quoted and the error has even been compounded in other work (e.g. Balmford et al., 2002). It is perhaps significant that these papers tend to be multi-authored with a minority of economists involved and that they appear in science journals rather than environmental economics journals. More sober reviews of ecosystem economic values have begun to emerge. For example, Pearce (2001), Pearce and Pearce (2001) and Pearce et al. (2003) survey the estimates of economic value residing in forests and the differences in economic values in sustainably managed forests compared to conventionally managed forests. While non-market values for some forest areas can be substantial, these surveys show that carbon storage and sequestration values tend to dominate overall non-market values, with the latter tending to be small and non-competitive with alternative use values. Similar reviews can be found for coral reefs (Cesar, 2000) and wetlands (Woodward and Wui, 2001; Brouwer et al., 1999). As yet, only a few of these studies attempt direct comparison of conservation values with 'development' alternatives (some attempt is made in Pearce and Pearce, 2001, for forests), so that little can be said about whether or not conservation values would dominate alternative use values if they were captured in markets. As noted above, out-bidding alternative use values may not be so important if there are additional motives for conservation, backed by real resources. Much therefore depends on context.

Despite the emergence of attempts to review what we know about economic values of ecosystem services, analysis of some of the broader functions remains in its infancy. Two deficiencies exist. First, valuation studies have tended to focus on selected use values (recreation, carbon storage, watershed protection, etc.) and only a few studies have attempted to ask about non-use values, values associated with the ecosystems by people who neither use the systems nor ever intend to use them. Pearce (2001) reviews the available studies for non-use values in forests and suggests that they imply per hectare valuations of about \$4, with a notable outlier in respect of protection of the habitat of the Mexican spotted owl at \$4000 ha! Leaving aside much publicised and exotic issues, non-use values again appear to be low when expressed in dollars per hectare terms. Nonetheless, one could wish for a bigger sample of studies on which to base any conclusion. Second, valuation studies to date do not appear to be addressing some of the broad functions of ecosystems. For example, diverse ecosystems generate insurance. Homogenous systems are more at risk from sudden shocks and chronic stresses, as the perpetual race against crop diseases has shown. There is therefore an insurance value in maintaining natural diversity. It is unclear what the economic value of this diversity is, but certainly some consider it to be very large (e.g. Heal, 2000). In the same way, studies have only just begun to consider the value of knowledge embodied in natural systems. This issue is addressed in at least one respect later when we look at the value of 'bioprospecting' (Sect. 6 below). The suspicion remains that the knowledge resulting from millions of years of evolution and embodied in many different ecosystems has yet to be understood and measured.

2.2 Capture studies

Just as the initial studies claimed high embodied economic values in natural systems, there has perhaps been excessive optimism about the ease with which markets can be created. Claims that markets and environmental conservation are complementary bedfellows have often ignored the institutional requirements for creating markets, not least the difficulties of changing or conferring property rights and of getting the central players to cooperate for the common good. Even when property rights are fairly clear, setting up agreements to trade in environmental services can be extremely complex and fragile. Even if trade is established, there is no guarantee that it will persist. Examples of this optimism are Hawken et al. (2000) and Daily (1997).

In yet other cases, claims, sometimes coming from surprising sources, have been made for market creation which appear not to withstand scrutiny. One of the more curious cases relates to the Catskills mountains watershed, which serves New York City. Historically, the watershed both purified and regulated the flow of water for New York. According to a widely quoted paper (Chichilnisky and Heal, 1998), over the years the water supply became increasingly polluted and failed to meet US Environmental Protection Agency standards. New York faced a stark choice: either to manage the entire watershed so it could revert to its original function of cleansing the water supply, or to build a new purification plant. Chichilnisky and Heal (1998) indicate that the former option would cost around \$1 billion and the latter over \$10 billion (in present value terms). New York decided in favour of the option to restore the integrity of the Catskills watershed and, it is claimed, floated an environmental bond to raise the necessary finance. If true, the example would be an instance of market creation because the ecosystem services of the watershed were effectively being marketed as a commercial product. But, somewhat mysteriously, it is unclear if this 'bargain with nature' exists at all. Sagoff (2002) casts doubt on the entire edifice of the Catskills story. He notes, first of all, that there is no record of water from the Catskills watershed deteriorating. Nor is there any evidence that pollution pressures in the watershed have increased. Indeed, the population there has hardly changed in a century or more. No evidence could be found of New York City issuing an environmental bond. New York did enter into an agreement with the US Environmental Protection Agency to comply with a new nation-wide requirement covering protection against the microbe *Cryptosporidium parvum*, nothing to do with locally declining water quality. Part of that agreement involved some ill-defined amount of habitat restoration, but New York also began to invest in more conventional means of water treatment. Actual land purchases in the watershed appear to have been minimal. If Sagoff is right, the 'Catskills parable' is more myth than fact, a caution against over-optimism in selling the market creation message.

Because claims and counter-claims about values and the ease of value capture are pervasive, it seems appropriate to try and get some quantitative perspective on beneficiary-pays market creation. In doing so, it should not be forgotten that such markets are only one of the two general types of market creation. Polluter-pays markets may well be vastly greater in terms of transactions value than beneficiary-pays markets. For example, the size of the greenhouse-gas trading market could run into billions of dollars, even without the USA as signatory to the Kyoto Protocol (Grubb et al., 1999; Springer, 2003), although, even here, there seems to be evidence of the same unintended exaggeration as that which has characterised the early discussions of beneficiary-pays markets.

3 The importance of property rights

On their own, markets will not guarantee conservation. A clear example is the 'bushmeat' trade where there is ample evidence that the taking of wild meat is contributing to the rapid demise of wildlife in many high-diversity areas and may even be more important as a cause of localised (and even global) extinction than habitat destruction (Robinson et al., 1999; Wilkie and Godoy, 2000). The reason such markets fail the goal of sustainable use and conservation is that the ultimate property right – in the land or marine areas occupied by the wildlife, the 'base resource' as Swanson (1994) calls it – is undefined or, if defined on paper, is unenforceable in practice. As is well known in bioeconomics, the combination of open-access and competing resource users produces what Hardin (1968) called the 'tragedy of the commons', which, less prosaically, should really be the 'tragedy of open access'. Economists had long shown that the economic rents in the resource could quickly be dissipated and that the resulting equilibrium in which zero rents prevailed could be close to a biological depensation point with a high risk of extinction. In short, for market creation to work, there must be well-defined property rights and feasible trade. Conferral of rights on its own will not guarantee conservation, for reasons to be explored, but neither will creating a market without property rights in the habitat. Finally, the dominant force in securing a feasible trade has to be the size of the mutual gains obtained by trading, i.e. the net surplus. If polluter profits are high relative to external costs, then the polluter has a strong incentive to trade if he does not have the property rights, but a low incentive if he does have the rights. Similarly, if external costs are high relative to polluter profits, then the sufferer has a strong incentive to bargain if he does not have the property rights, but a low incentive to trade if he does have the rights.

The *locus classicus* for the theory of beneficiary-pays market creation is the Coase theorem (Coase, 1960). What Coase showed was that, under very restrictive assumptions, an economically optimal level of conservation (pollution, etc.) would emerge from any initial allocation of property rights. Thus, it does not matter if a resource destroyer or polluter owns the resource, or whether the beneficiary of conservation (i.e. the sufferer if the resource is degraded or destroyed) owns the resource. In the former case, the sufferer will pay the polluter not to pollute, but only up to the point where the avoided marginal suffering (the marginal external cost, or the marginal benefit to the sufferer from having the pollution reduced) is just equal to the marginal gains secured by the polluter from the polluting activity (the marginal profit or marginal net benefit to the polluter). Thus, this form of bargaining secures the condition that marginal external costs and marginal net private benefits are equal, the condition for optimality.[2] This result holds in the case of perfect competition but requires tripartite bargaining between polluter, sufferer and consumer once imperfect competition is introduced (Buchanan, 1969). If the property rights reside with the sufferer, then the polluter can pay compensation to the sufferer so long as marginal net benefits from pollution exceed marginal external cost. Once again, an equilibrium is reached where price and marginal social cost are equated (see footnote 2). Note that bargaining makes sense if the net gains from trade are significant. Expositions of the theorem usually assume that they are significant. As discussed earlier, the likelihood of trade will be dependent on the gains from trade. If only limited trades are observed, this may be because the relevant economic values are not large enough to justify trade.

Limited trades may also be due to transactions costs. Indeed, many regard the most restrictive condition in the Coase theorem to be that bargaining is costless. In reality, we know that transactions costs are very important in actual bargains. This immediately suggests a role for government, provided that intervention costs do not outweigh the gains from trade, something that cannot be guaranteed. Intervention here would typically mean 'facilitating' the bargain by actions which directly reduce transactions costs (e.g. government may have more access to information about polluters or sufferers than do the parties themselves, an obverse of the usual assumption about asymmetric information), or by the government taking over the bargain on behalf of one of the parties. The second rationale for intervention is that Coaseian bargains are indifferent to equity concerns – the theorem is about efficiency alone. But governments are highly likely to have equity concerns. Interestingly, these may arise especially where either the sufferer is poor or the polluter is poor. In the former case, government may take on the role of acting for the poor sufferer. This is the case with the Costa Rican ecosystem service payments discussed later (see Sect. 5). The government effectively acts for downstream beneficiaries of upstream forest conservation and the presumption is that many of these beneficiaries are relatively poor and could not pay for beneficial conservation. The case where the polluter is poor is less obvious, but a striking example is the technical and financial assistance given by Scandinavian countries to Baltic countries to switch energy generating technologies away from high polluting to less polluting ones. The benefit to Scandinavia is the reduced transboundary acid rain deposition that results. As long as Scandinavian payments are less than the value of the avoided damage, Scandinavia is better off. As long as the incremental cost of the cleaner technology is zero or negative to the Baltic States, they are better off.[3]

[2] The bargaining function of the polluter is essentially price – marginal cost, P-MC. The bargaining curve of the sufferer is his/her marginal external cost, MEC. Hence, P – MC = MEC is simply rearranged as P = MSC, where MSC is marginal social cost, the condition for optimality.

[3] The cost to the polluting nations can be negative if the incremental cost of the technology is zero and the polluting nations also make gains, e.g. from reduced local pollution.

While the Coase theorem sets the theoretical backdrop for market creation, it is really its deficiency in assuming costless bargaining that produces the real-world cases of beneficiary-pays market creation. Other features that usually necessitate some form of government intervention include the need to create binding contracts between the bargaining parties and some system of monitoring and enforcement to ensure contracts are not broken. However, the resulting intervention tends to be minimised so that market creation becomes a 'soft' form of regulation in which the parameters for bargains are set and groups are then generally left to bargain with each other to varying degrees. The greater the degree of government intervention the less the bargain approximates a Coasian bargain – indeed, one of the attractions of the Coase theorem is the potential it holds out for avoiding heavy-handed command and control.

With the Coase theorem in mind, we can now look at some examples of market creation.

4 Debt-for-nature swaps

Resource-rich developing countries usually have a high degree of foreign indebtedness. Since the burden of debt repayment is often judged to impair the prospects of future economic development, various 'debt forgiveness' schemes operate. While these retain some element of conditionality – debt is forgiven for some undertaking about, say, anti-poverty policies – some swaps involve direct exchanges. These might be education or health investments in exchange for debt forgiveness or, of relevance to this paper, environmental conservation for debt forgiveness. The 'forgiveness' in this case, however, involves conversion of the debt denominated in foreign exchange to domestic interest-yielding bonds. These 'debt-for-nature swaps' (DfNSs) began in the late 1980s and continue to this day, although the parties involved tend to have changed over the years. All swaps are confined to commercial debt (i.e. debts owed to private lenders such as commercial banks) and official bilateral debt (i.e. debt owed to foreign governments). No multilateral debt (e.g. World Bank loans) is involved in the swaps, which has limited the prospects for developing this instrument. Bilateral debt deals tend to operate through the Paris Club, a group of bilateral lenders dedicated to reducing and converting debt that threatens poor country development. In 1990, the Paris Club agreed to allow a considerable portion of international debt to be dealt with via debt-for-development swaps. In the event, only a limited number of creditor countries have operated such schemes.

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' secondary 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 non-governmental organisation (NGO), to protect an environmentally important area, train conservationists, reduce pollution threats, etc. One of the most celebrated debt swaps involving governments and NGOs are those under the Enterprise for the Americas Initiative (EfAI), established in 1990. The debt in question is owed by Latin American and Caribbean countries to the USA. The US Tropical Forest Conservation Act (TFCA) of 1998 enabled further expansions of the EfAI, permitting debt reductions against forest conservation. From 1991 to 1993, EfAI conversions amounted to $875 million face value, creating local trust funds in seven Latin American/Caribbean countries of $154 million. The TFCA has provision for $325 million of funding. Another significant government player in DFNSs is Switzerland, which set up a Swiss Debt Reduction Facility in 1991. The Swiss programme involves several forms of conditionality: there must be economic reform in the indebted country, rule of law and a general debt reduction programme. The Swiss deals have involved some $460 million face value debt or over $160 million of redemption value and investment funds (leverage appears to be zero on the Swiss deals).

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. Table 1 summarises DfNSs up to 2003.

Table 1 reveals some interesting features. First, one deal, involving Poland and the Paris Club members, accounts for 60 per cent of the face value of the total debt, and 50 per cent of the aggregate purchase price. Another group of deals in North Africa and the Middle East accounts for a further 25 per cent of the aggregate purchase price. Second, the total sum realised has been at least $1.1 billion, the purchase price of the debt. This is substantial. Third, Sudo (2003) records the value of the conservation funds generated by the various deals, i.e. allowing for the leverage of the funds generated by the debt purchase price. Total investment funds are some $1600 million without Poland and $2170 million with Poland, suggesting an average leverage factor of 2.7 excluding the Polish deal and 1.9 with the deal. All in all, then, some six deals out of over 100 account for 75 per cent of the aggregate purchase price. Some $2.2 billion has been generated for investment in conservation since 1987. Fourth, there has been an understandable focus in terms of the number of deals on ‘mega-diverse’ or ‘biodiversity hot-spots’ countries. How far it is sensible to focus on such countries is, however, debatable. Hot spot locations tend to be classified as such because they face the greatest threat of biodiversity loss. But those very threats, e.g. rapid population growth, may mean that investments in conservation are very high risk. Suggestions have been made for ranking criteria to reflect the degree of threat and the chances of successful investment. These do not necessarily coincide with the rankings secured by considering biodiversity scarcity or threats to biodiversity. Fifth, Latin America and Asia dominate the numbers of swaps. Only fourteen swaps have taken place in Africa, of which six are in

Table 1. Debt-for-nature swaps (1987–2003)

Host country (number of swaps)	Donors/ purchasers	Years	Total face value of debt ($ million, rounded)	Total discounted value of debt ($ million, rounded)
Bolivia (6)	Germany, TNC/WWF Switzerland, CI, EAI	1987, 1991, 1992, 1993, 1996, 1997	115	>10
Peru (6)	WWF, CI, Finland, Switzerland, TNC, Germany, Canada	1993, 1995, 1998, 1999, 2002	232	36
Mexico (13)	CI, USAID (share of one swap only)	1991, 1992, 1993, 1994, 1995, 1996, 1997	5	3
Costa Rica (8)	WWF, Canada, Sweden, Spain, TNC, Netherlands, NPF, Rainforest Alliance	1988, 1989, 1990, 1991, 1995, 1999	100	25
Ecuador (6)	Switzerland, Japan, Belgium, WWF, TNC, MBG	1987, 1989, 1992, 1994	>62	>19
Nicaragua (3), Belize (1),	Germany, TFCA,	1991, 1992, 1993,	465	>68
El Salvador (2), Chile (1),	Canada, Switzerland,	1994, 2000, 2001	(includes 271 in one	

Table 1 (continued)

Host country (number of swaps)	Donors/ purchasers	Years	Total face value of debt ($ million, rounded)	Total discounted value of debt ($ million, rounded)
Colombia(1), Panama (2),	TNC, CI, USAID EAI swap with Jamaica)			
Guatemala (2), Brazil (1), Honduras (2), Jamaica(2), Dominican Republic (1)				
Tunisia (5)	Netherlands 1996, 1997	1992, 1994, 1995,	26	n.a.
Jordan (6)	Germany, Italy, France, Switzerland	1994, 1999, 2000, 2001	203	>50
Egypt (2), Syria (1), Morocco (1), Algeria (1)	Switzerland, Italy, Germany	1995, 2000, 2001, 2002	528	>281
Philippines (7)	WWF, USAID, Germany, TFCA	1989, 1990, 1991, 1992, 1993, 1996, 2002	64	39
Vietnam (3), Bangladesh (1)	Germany, TFCA	1996, 1999, 2000, 2001	56	24
Madagascar (6)	WWF, CI, UNDP, Netherlands	1989, 1990, 1991, 1993, 1996	16	>7

Table 1 (continued)

Host country (number of swaps)	Donors/ purchasers	Years	Total face value of debt ($ million, rounded)	Total discounted value of debt ($ million, rounded)
Tanzania (2), Nigeria (1),	Switzerland,	1989, 1991, 1993,	51	5
Guinea Biss (2) Zambia(1),	Belgium, NCF, DDC,	1995, 1997		
Cote d'Ivoire (1), Ghana(1)	CI, SI, WWF			
Poland (2)	Polish Ecofund via Paris Club	1992	2,897	571
Bulgaria (1)	Switzerland	1996	20	20
TOTAL excluding Poland (100)			1,943	582
TOTAL including Poland (102)			4,840	1,153

Source: adapted from Sudo (2003)
Notes:
CI: Conservation International
DDC: Debt for Development Coalition
EAI: Enterprise for the Americas Initiative
MBG: Missouri Botanical Garden
NCF: Nigeria Conservation Foundation
NPF: National Parks Foundation
SI: Smithsonian Institution
TFCA: US Tropical Forest Conservation Act
TNC: The Nature Conservancy
UNDP: United Nations Development Programme
USAID: US Agency for International Development
WWF: Worldwide Fund for Nature

Madagascar. This suggests that the swaps tend to be concentrated in the more stable countries, i.e. that they follow the risk profile of conventional portfolios (although Madagscar has been the subject of a political coup). But many mega-diverse countries are those with the least investment stability. This may account for the low fraction of funds going to Africa. Sixth, many of the swaps are very small in financial terms, with some 50 per cent having purchase prices less than \$2 million. Finally, the role of international NGOs is clear. Roughly half the swaps involve a major NGO, with Worldwide Fund for Nature (WWF) and Conservation International (CI) being the major ones.

While such facts are interesting, the real question is how effective DfNSs are in environmental terms. Since there appear to be few ex post evaluations of DfNSs, the approach taken here is an indirect one. The payment made by the initial purchaser of the secondary debt reflects at least part of the rich nations' willingness to pay for conservation in developing countries. Since conservation in such countries tends to have global public good characteristics, there is likely to be substantial free riding. In other words, there will be many other beneficiaries who are paying nothing for the resulting protection, so that the actual payment does not measure the world's willingness to pay for the conservation. Rather, as Ruitenbeek (1992) notes, the figure is a supply price. Different DfNSs can be expected to come up with different implicit prices since the nature of the 'good' being bought will vary (e.g. the quality of the area protected will vary and different packages of measures will be involved). The implicit 'price' of a hectare (ha) of protected land (P_{ha}) is given by:

$$P_{ha} = \frac{PV[RP + L]}{PV(ha)} \tag{1}$$

where *PV(ha)* is the present value of the land expressed in hectares, *RP* is the redemption price and L is any other revenue from leverage. Table 1 does not show land areas affected by the DfNSs, but Pearce and Moran (1994), following on earlier work by Ruitenbeek (1992), analyse some of the early DfNSs where this information is available. They suggest that an implicit price of, at most, \$5 ha is being paid for the 'average' swap.

There are several ways of viewing this \$5 ha figure. First, as noted above, it is not an aggregate willingness to pay figure because of free riding, i.e. it does not reflect the economic value of conservation. Second, the payment must be sufficient to induce a change in management practice in the host country, or meet a shortfall in domestic conservation expenditures compared to the minimum needed for conservation, or be sufficient to offset the opportunity cost of conservation. In the absence of detailed figures showing host country expenditures and the value of alternative uses it is difficult to be precise. Some idea of the cost of conservation can be derived from the work of Abramovitz (1991) who estimated that USA funding for biological diversity in various regions only exceeded \$5 per *1000 hectares* (i.e. \$0.005 per ha) in 23 out of 127 countries for which survey data were available. Even allowing for different year prices and co-financing, the implicit value of \$5 ha in the DfNS analysis therefore greatly exceeds US aid by orders of magnitude. Only two countries – Jamaica and Costa Rica – received more than \$1 per hectare of US funding. The problem, of course, is that this kind of indirect analysis shifts the burden of proof of effectiveness from the DfNS to conventional forms of funding, for e.g. protected areas. On balance, while there is considerable uncertainty, it looks as if DfNS may be quite generous in terms of supplementing traditional means of conservation finance.

Far less persuasive is the idea that $5 ha offsets the value of alternative uses. It is true that substantial areas of land in the tropics have close-to-zero commercial values because of poor soil, distance to market, absence of roads, etc. Areas closer to markets and where some infrastructure exists – even logging roads – command much higher land prices than $5 ha. Again, the $5 is on top of any unknown domestic expenditures, but the suspicion must be that at least some DfNSs are 'protecting' areas where there is no real risk of conversion. Put another way, they may not be protecting the status quo of protected areas so much as preventing the more gradual decay of some of those areas through lack of management funds. Finally, while we have noted the leverage secured by DfNSs, there remains an unknown issue there too. We do not know how far the leveraged funds are truly 'additional' to the funds that would have been available for conservation generally. There is a 'counterfactual' or 'baseline' problem, which is common to much conservation expenditure. Until we have proper ex post appraisals of conversion threats and the environmental impacts of DfNSs, it is hard to reach a conclusion.

5 Costa Rica's Forest Law

Costa Rica has given official recognition to the role that forests play in: (a) biodiversity conservation; (b) carbon storage; (c) watershed protection; and (d) scenic beauty for ecotourism. As part of a wider system of 'Pago por Servicios Ambientales' (PSA), in 1996 Costa Rica adopted a new Forest Law whereby forest land owners can be compensated for the provision of these services. The Forest Law builds on an earlier and fairly elaborate system of payments for supporting the timber industry. The change to paying for environmental services was nonetheless dramatic. Financing for payments comes from a gasoline tax, a tax on wood products, the issue of 'forest bonds' and from some other beneficiary payments. Finance is directed via the National Forestry Fund (FONAFIFO). Landholders can secure payment for reforestation, sustainable management practices, forest regeneration and forest conservation. The payment schedule is outlined in Table 2.

In 1997, some $14 million was disbursed for the conservation of 79,000 ha, a cost of under $200 per ha. This average sum and the sums in Table 2 are indicative of the very much higher 'protection' costs under the Forest Law than under DfNSs

Table 2. Payment schedules under Costa Rica's Forest Law

Activity	Payment per hectare over 5 years	Annual schedule, % payments per year
Reforestation	$480	50,20,15,10,5
Natural forest management	$320	50,20,10.10,10
Forest regeneration	$200	20,20,20,20,20
Forest protection	$200	20,20,20,20,20

Source: Chomitz et al. (1998)

from which Costa Rica has also benefited. The disparity perhaps underlines the fact that DFNSs may be 'protecting' land not under grave threat of conversion, whereas the Forest Law clearly is protecting convertible land, i.e. the Costa Rican figures are more indicative of the opportunity cost of conserved land. Essentially, the $200 must compensate for the opportunity cost of the higher profitability of unsustainable forestry. The reforestation payments appear to be more generous and probably would act as a strong incentive to reforest. The regeneration incentive is thought to be about equal to the rental price for pasture, i.e. giving an incentive to regenerate rather than lease the land for cattle. There has been excess demand for the programme, the supply capacity being determined by the availability of finance.

Is Costa Rica's experiment a Coaseian bargain? There are some basic differences. So long as the forest conservation is a public good, the beneficiaries are the whole of Costa Rica's population. Moreover, given the role that forests play in carbon sequestration and in generating existence value, the beneficiaries are actually global because of the reduced climate change effect and the global willingness to pay for habitat and species conservation independent of any use the beneficiary may make of the resource. As it happens, foreign nationals are not providing finance for the Forest Fund, but nationals are, via the various taxes in question. The original plan did encompass the idea of foreigners paying part of the compensation because it was intended that carbon emission credits would be sold and the revenues from these would finance a significant part of the payments made.

Second, the government is a main player in the compensation mechanism. The Coaseian character of the policy is retained so long as the government can be seen as acting as an agent for the Costa Rican population in general and poor downstream farmers in particular, overcoming the insensitivity of Coaseian solutions to equity concerns.

Third, similar to a DfNS, the Forest Law effectively changes the rights regime so long as payments are being made. During that time, farmers cede their environmental service rights to the National Forestry Fund whilst retaining the rights of land ownership. While this may seem inconsistent with a Coaseian bargain, property rights are in fact attenuated in a similar fashion in a 'pure' bargain. Once the sufferer pays the polluter, the polluter has to abide by an agreement not to pollute.

Earlier, it was noted that transactions costs can quickly render Coaseian-style bargains unworkable. In the Costa Rican case, intermediaries or 'brokers' have emerged to assist with the deals, thus lowering transactions costs as the intermediaries learn to standardise contracts and learn from experience.

The PSA also recognises the value of water provided by natural ecosystems and the biodiversity associated with those systems. Landowners in hillside areas can claim payments from water consumers. One study found that a charge of 2.7 colones (about 1 US cent) per cubic metre of water could produce a payment of some $90 ha to upstream farmers and this sum reflects the opportunity cost of forest conservation for the supply of water. Consumers' actual willingness to pay appears to be higher than this required sum (Castro, 2000). Additional benefits (e.g. carbon sequestration) from forest conservation make up the rest of the sum needed to compensate farmers for not converting forest to cattle production. In this case, the Coaseian nature of the bargain is more direct. Property rights belong to landowners and hence their conservation of forests amounts to generating an external benefit. To date, the only water users making payments under the PSA programme have been hydro-electricity utilities. The gain to them of reduced reservoir sedimentation and hence higher electricity output is fairly obvious. Other users, including domestic users, have shown less willingness to enter into the trades. Pagiola (2002) states that the hydroelectric producers have paid some \$100,000 up to 2002 for agreements covering 2400 ha, i.e. around \$40 ha of 'conserved' land. In 2000, Costa Rica established a trial 'adjusted water tariff' programme in an effort to maintain watershed areas near Heredia. The principle is the same as in the PSA programme.

How important is the Costa Rican experience in helping to reduce deforestation? The experience is very much a test case. If it works, other countries can be expected to follow and, indeed, some already have done so. Hence, the results should be treated in terms of what is learned from the experience rather than whether there are identifiable and significant changes in forest cover compared to a baseline of what would happen without the policy. Furthermore, while government is very much involved, the mainspring for the bargains is the fact that there are mutual gains to be made. Market forces alone might eventually have produced similar forms of bargain, but the presence of transactions costs has prompted government (and NGO) involvement. It seems clear that equity considerations have also been important. A number of authors have argued that the value of ecosystem services of forests implied by the payment schedules is exaggerated (for a discussion see Pagiola et al., 2002). If that is true, then the Costa Rican system is surviving because of the artificial prices attached to the services by the government. Those prices would then owe more to the ability to raise revenues from other taxes (e.g. fuel taxes) than to genuine downstream gains. Payments may approximate the opportunity costs of forest conservation, but not the economic values of forest protection. Whether there is room to raise the prices further depends on the maximum willingness to pay for the services by the beneficiaries, which in turn means identifying far more clearly just what those beneficiaries are. The fact that there have been so few players in the water payments provision of the PSA could suggest that the willingness to pay is limited. Or it may simply be a feature of the novelty of the process. Finally, payments to landowners are annual and over a limited period. The success of the scheme will depend on the extent to which the resulting easements secure far more permanent postponement of forest conversion beyond the agreement period.

The Costa Rican experiment also acknowledges that forest owners do not have an incentive to undertake sustainable forestry for its own sake, otherwise there would be no need for the law. In other words, sustainable forestry pays less than unsustainable forms of land use. This picture is confirmed by Pearce et al. (2003) in their survey of unsustainable and sustainable forest practices. Hence, unless efforts are made to 'capture' the ecosystem service values, deforestation will continue. Pagiola et al. (2002) suggests that market creation stands more chance of success than other measures, most of which have already been tried with little success, including forms of outright protection or regulated land uses, which can often lower financial returns to poor people, and subsidies to 'sustainable' activities. Viewed as a learning process, it is to the credit of Costa Rica that it has pioneered experiments of this kind.

6 Bioprospecting

Early excitement about the economic values embodied in nature arose primarily from the view that, since pharmaceutical companies had huge billion dollar sales of drugs based on natural materials, the value of those materials must similarly be huge. The data are seductive. For example, world markets in products derived from genetic resources are estimated to be valued at $500-800 billion (ten Kate and Laird, 1999). Hence, it appears that, provided 'bioprospectors' could be induced to pay for access to genetic material, the subsequent cash flows should be substantial. The most widely cited contract of this kind was negotiated in 1991 between Merck, a major pharmaceutical company, and Costa Rica's 'INBio' (Institute Nacional de la Biodiversidad). In return for a payment of over $1 million, Merck secured bioprospecting rights in Costa Rica. The money paid was to be used partly for forest conservation but mainly for INBio's own training and equipment. In addition, Merck offered to share any profits from the successful development of any drug from the genetic material obtained. It is important to understand that the success rate of developing commercial drugs from genetic material is very small, so the profit share, whatever it might be, needs to be multiplied by the probability of commercial success to obtain an expected value of the return to Costa Rica. An average probability is some 1 in 250,000. The Merck-INBio deal involved just 2000 samples. The royalty share was never disclosed but is thought to have been around 1-3 per cent of eventual profits (Simpson, 2001). The deal expired in the late 1990s and no commercial drug was developed. INBio has entered into other contracts of a similar kind and appears to have gained financially from these exercises. How far forest conservation has been advanced remains unknown. In all likelihood, the effect has been small. This is not surprising. The Merck-INBio deal was the first of its kind and hence its value lies more in learning about how to develop such contracts to see if they are viable and replicable. Certainly, Costa Rica, which we have already seen has been in the vanguard of market creation, secured good publicity, as, in general, did Merck.[4] To give perspective, however, Merck's expenditure on this contract was less than 0.1 per cent of the company's research and development budget for that year (Firn, 2003). Firn also reports Macilwain (1998) to the effect that no major pharmaceutical company has found bioprospecting especially rewarding.

How far bioprospecting can contribute to sustainable development depends on the extent to which: (a) the Merck-INBio type contracts can be multiplied; (b) the leverage such contracts give to conservation efforts; and (c) the willingness to pay of bioprospectors for access to natural organisms, i.e. the price paid for access to genetic resources. Relevant factors, some favourable to bioprospecting, some not, are as follows. First, there are technological developments that are likely to reduce the need of bioprospectors to have access to natural organisms, notably the ability to use synthetic and combinatorial chemistry and biotechnology using human genes. Put another way, natural sources are not the only sources of new material, nor even the most likely to succeed. A contrary view is that scientific developments in the study of plants and development of genetic materials will enhance the demand. Second, technological change is increasing the ability to

[4] Some critics accused Merck of 'bio-piracy' and INBio as collaborators.

exploit further existing collections of seeds, reducing the need for access to new genetic resources. Third, search processes are becoming very selective, favouring particular areas with known prior information, and thus reducing the demand for access to new areas as a whole. Fourth, paralleling the demand for organic foods, there is a growing demand for 'natural' products that require direct access to genetic material. Fifth, legal and institutional difficulties in securing access may well deter bioprospectors. This partly reflects the limited institutional structure in many host countries, bureaucracy and even corruption. Sixth, the supply of genetic material is vast. At best, bioprospectors can be expected to 'demand' only a tiny fraction of what is available, so that most natural areas will be very unlikely to benefit from bioprospecting. Seventh, international patent law still discriminates against worldwide protection for natural materials.

These variable forces affecting supply and demand should show up in the price received for genetic material. No consistent tabulation of contract prices appears to be available (for limited information see ten Kate and Laird, 1999), but various efforts have been made to estimate what a bioprospector would be willing to pay for forest genetic material. The most sophisticated studies have been those carried out by David Simpson and others at Resources for the Future (RFF), Washington DC (Simpson et al., 1996; Craft and Simpson, 2001) and at the University of California at Berkeley (Rausser and Small, 2000), although work by Barbier and Aylward (1996) which analyses the MERCK-INBio deal reached similar conclusions. The Simpson et al. studies tend to paint a gloomy picture for those who believe that pharmaceutical values will 'save' the world's forested areas where biodiversity is high. The California studies criticise the Simpson et al. studies and offer a more optimistic picture, but these studies have themselves been criticised (e.g. Costello and Ward, 2003). Others have stressed the simple intuition that bioprospecting values will be low because: (a) the genetic material is simply part of a very much larger process of drug development; (b) relative to reliance on developing synthetic materials, bioprospecting has all kinds of disadvantages; and (c) the supply of genetic material remains huge (Firn, 2003).

In contrast to previous estimates of the economic value of genetic material for pharmaceuticals, which tended to estimate either average values or which attributed all drug value to the genetic resource, the recent studies correctly try to estimate the economic value of the *marginal species*. What matters are the costs and benefits of the change in the total stock, not the value of the stock over all (even if it made sense to speak of the 'total' value at all). In the pharmaceutical context, the relevant economic value is the contribution that one more species makes to the development of new pharmaceutical products.

The fundamental equation elicited by Simpson et al. (1996) is given below:

$$\max WTP = \frac{\lambda.(R-c)e^{\frac{-R}{R-K}}}{r(n+1)} \tag{2}$$

where *WTP* is willingness to pay; λ is the expected number of potential products to be identified (10.52); n is the number of species that could be sampled (250,000); c is the cost of determining whether a species will yield a successful product ($3,600); r is the discount rate (10% = 0.1); e is the natural logarithm (2.718); K is the expected research and development cost per new product successfully produced ($300 million); and R are the revenues from new product net of costs of sales but gross of research and development costs ($450 million).

Note the very large sums for K and R: developing new drugs is extremely expensive and the revenues from successful ones are potentially extremely large. One implication is that pharmaceutical companies may find paying for prospecting rights easy so long as such rights are small fractions of the very large development costs. But, as noted above, if there are alternative routes to finding the genetic material, making prospecting difficult through bureaucratic procedures and high transactions costs, the prospecting companies may well take them.

Substituting the estimates above into equation (2) gives a maximum willingness to pay of $9410 for the marginal species. Willingness to pay for the marginal species is not a concept with which it is easy to identify. Accordingly, the literature tends to translate these values into willingness to pay for land that is subject to the risk of conversion. This is done as follows. First, the 'species-area' relationship is given by:

$$n = \alpha A^{Z} \tag{3}$$

where n is the number of species; A is area; α is a constant reflecting the species richness potential of the area; and Z is a constant equal to 0.25. Species-area equations of this kind are widely used to estimate the number of species likely to be present on a given area of land. Second, the economic value V of land area A is given by:

$$V[n(A)] \tag{4}$$

Expression (4) refers to the value (V) of a collection of species (n) likely to be found in area A. Third, the value of a change in land area A is given by differentiating (4):

$$\frac{\partial V}{\partial A} = \frac{\partial V . \partial n}{\partial n . \partial A} \tag{5}$$

Equation (5) is what we need to estimate. The expression $\frac{\partial V}{\partial n}$ is the marginal value of the species, for example the $9410 derived above. The expression $\frac{\partial n}{\partial A}$ is the change in the number of species brought about by a small change in the land area. Differentiating (3) gives:

$$\frac{\partial n}{\partial A} = Z\alpha A^{Z-1} = \frac{Z.n}{A} = Z.D \tag{6}$$

where $D = n/A$ is the density of species. Hence, the bioprospecting value of marginal land is given by:

$$\frac{\partial V}{\partial A} = \frac{\partial V}{\partial n}.Z.\frac{n}{A} \tag{7}$$

or, simply, the value of the marginal species $\times 0.25 \times$ density of species.

Table 3. Estimates of the pharmaceutical value of 'hot spot' land areas

Area	Simpson et al. (1996) WTP of pharmaceutical companies per ha	Simpson and Craft (1996) 'Social value' of genetic material per ha	Rausser and Small (2000) WTP of pharmaceutical companies per ha
Western Ecuador	20.6	2,888	9,177
Southwestern Sri Lanka	16.8	2,357	7,463
New Caledonia	12.4	1,739	5,473
Madagascar	6.9	961	2,961
Western Ghats of India	4.8	668	2,026
Philippines	4.7	652	1,973
Atlantic Coast Brazil	4.4	619	1,867
Uplands of western Amazonia	2.6	363	1,043
Tanzania	2.1	290	811
Cape Floristic Province, S. Africa	1.7	233	632
Peninsular Malaysia	1.5	206	539
Southwestern Australia	1.2	171	435
Ivory Coast	1.1	160	394
Northern Borneo	1.0	138	332
Eastern Himalayas	1.0	137	332
Colombian Choco	0.8	106	231
Central Chile	0.7	104	231
California Floristic Province	0.2	29	0

(maximum WTP $ per hectare)
Sources: Simpson et al. (1996); Simpson and Craft (1996); Rausser and Small (2000)
Notes:
ha: hectare; WTP: willingness to pay

The resulting values derived by Simpson et al. (1996) are given in the second column of Table 3.[5] The overwhelming impression is of the very small values that emerge. While 'hot spot' land often exchanges for low prices, those prices are almost universally well in excess of even the highest value found by Simpson et al. (1996), i.e. around $20 ha. The essential reasons for the low values are: (a) that biodiversity is abundant and hence one extra species has low economic value; and (b) that there is extensive 'redundancy' in that, once a discovery is made, finding the compound again has no value. Each additional 'lead' is likely to be non-useful or, if useful, redundant. Either way, low values result. Simpson and Sedjo (1996) offer some further 'scenarios' to show the sensitivity of the value of the marginal species to the assumed abundance of species (n). With n = 250,000 the value of the marginal species might be $2500, but with n =1 million the value is effectively zero.

[5] In Simpson (1998a) the values per unit land-area are in fact even smaller than those shown here, by about an order of magnitude. The difference arises from the fact that the original estimates, shown here, are 'static' whereas the smaller estimates come from a 'dynamic' form of the Simpson et al. (1996) model. In the dynamic form of the model, testing of genetic material takes place until the marginal contribution (benefit) of the last species is equal to the marginal cost of waiting until the next period in which tests are conducted. Essentially, then, the dynamic model makes the marginal value of species for pharmaceutical use even smaller. See also Simpson (1998b).

The third column of Table 3 also shows later estimates by Simpson and Craft (1996). The basic difference between the Simpson et al. (1996) estimates and the Simpson and Craft (1996) estimates is that the former assume either perfect substitutability between species or no relationship between species, whereas the latter estimates assume that species are 'differentiated' such that one is not a perfect substitute for the other. The result is that the new estimates relate to 'social surplus', i.e. the sum of profits and consumer surplus and this is higher than the original estimate of the marginal value of a species. Simpson and Craft (1996) illustrate the outcome of their estimation procedure by assuming a 25% loss in the number of species. The result is a social loss of some \$111 billion in net present value terms, or around 0.01% of the world's gross national product when the former is expressed as an annuity. The policy implications of the earlier work by Simpson et al. (1996) are modified to some extent by the Simpson and Craft (1996) work. Whereas economic values of (effectively) zero to \$20 ha are extremely unlikely to affect land conversion decisions, the larger 'social' values could be relevant to changing land use in some areas. They conclude that 'modest incentives might be sufficient to motivate conservation in some areas' (Simpson and Craft, 1996, p. 4). The Simpson and Craft paper of 1996 is modified by a later paper (Craft and Simpson, 2001) which shows that 'social' values could be very different to private values, depending on the degree of complementarity presumed among new products. On one model, the social value could actually be negative due to excessive entry into the market for differentiated products. On another model, social values always exceed private values. The essential feature of these later models is allowing for competition between derived products as well as for the scarcity or otherwise of the natural resource. Social values become 'model-dependent and parameter specific' (Craft and Simpson, 2001, p.13).

The general import of the Simpson et al. (1996) work remains that private prospecting values are very small, whilst social values may or may not be significantly different. But the result that private values are very small has been challenged by Rausser and Small (2000). The fourth column of Table 3 shows Rausser and Small's (2000) estimates. Rausser and Small argue that the Simpson studies characterise the pharmaceutical companies' search programme as one of randomly selecting from large numbers of samples. Each sample is then as good as any other since each is assumed to contribute equally to the chances of success. This random sequential testing does not in fact describe a cost-minimising approach to selection. Rather, samples are selected on a structured basis according to various 'clues' about their likely productivity. 'Leads' showing high promise are therefore of significant value because they help to reduce the costs of search overall. Such leads are said to command 'information rents', i.e. an economic value that derives from their role in imparting information. In effect, samples cease to be of equal 'quality' with some samples having much higher demand because of their information value. Clues to that value may come from experience, knowledge of particular attributes, even indigenous use of existing materials. Rausser and Small (2000) argue that the information value attached to a lead arises from the costs of search and the probability of a success, with the value of the successful drug being relatively unimportant. The effect of having different probabilities of success is that an equation like (2) no longer applies. The Rausser-Small estimates confer greater value on biodiversity than do the Simpson-Craft estimates and substantially more than the Simpson et al. values. Rausser and Small (2000) conclude that 'The values associated with the highest quality sites – on the order of $9000/hectare in our simulation – can be large enough to motivate conservation activities'. The basic difference, it appears, is that the Rausser-Small model has 'informed search' while the Simpson et al. models have 'random search'.

Simpson (2003) argues that the Rausser and Small (2000) results are too optimistic. First, the Simpson et al. (1996) estimates were deliberately set up to be upper bound estimates. Actual marginal values would be considerably smaller. The Rausser-Small upper bounds also come from an unlikely context in which it is known that one species is a very good 'lead' and all others are very bad. Second, the real-world context of search is that we have no idea about more than 90 per cent of resources – they have not even been identified let along screened. If so, the assumption of random sampling does not seem incorrect. Moreover, if search is concentrated on species about which something is known, as the Rausser-Small model requires, it implies little or nothing for the vast number of species about which nothing is known. Costello and Ward (2003) test for the likely differences in value from informed search as compared to random search by conducting a numerical experiment. The startling finding is that the Rausser-Small values in Table 3 above hardly change if random search is substituted for optimal search. Indeed, the values are not very different if search is conducted perversely, i.e. by taking the lowest probabilities of success first. Costello and Ward show that the differences between Rausser-Small and Simpson et al. have nothing (or very little) to do with the search assumption. Rather, it is assumptions about parameter values that mainly explain the differences. For example, Equation (3) above set $Z = 0.25$ in the Simpson et al. (1996) model, where Z is the exponent in the species-area relationship. But Rausser-Small have an implicit assumption that $Z = 1$. Similarly, the value of n (the number of species) is far higher in Simpson et al. (1996) than in Rausser and Small (2000), lowering the values in the former case and raising them in the latter. Allowing for yet more differences in assumptions, Costello and Ward (2003) show that the Rausser-Small values multiply the Simpson et al. values by a factor of 344.

By shifting the focus to parameter estimates, the Costello-Ward analysis changes the debate. Previously, the search model seemed to explain the difference between optimism and pessimism about bioprospecting. In that case, it is comparatively easy to argue about which search model is the more realistic. Now that the difference seems to be explained mainly by parameter values, the issue becomes one of choosing the 'right' values. The problem is that the plausibility of these values has not been tested. Just as Craft and Simpson (2001) showed that *social* values are model and parameter dependent, the situation now appears to be that *private* values are also parameter dependent.

How far does the bioprospecting literature illuminate the policy dimension? If *private* prospecting values are high, as Rausser and Small (2000) would suggest, then there appears to be no role for social policy, i.e. there is no need for a policy instrument to encourage prospecting. However, social policy might be focused on ensuring that prospectors pay what they are alleged to be willing to pay, rather than treating genetic material as a de facto open access resources. To this end, the Convention on Biological Diversity would be right in its urging of host countries to extract their share of the rent through binding contracts. If values are small, as suggested by Simpson et al (1996), then we would not expect to see significant prospecting activity, nor would there be a rationale for encouraging it since the values to be captured would be small. Again, however, there would be case for encouraging host countries to extract their 'share' of the benefits, small as they may be. The more positive role for instruments to encourage prospecting comes if social and private values diverge significantly. The problem at the moment is that we have no real idea what this divergence is. What appeared to be significant differences in some cases (Simpson and Craft, 1996) now appears to be highly dependent on models and parameters (Craft and Simpson, 2001). Perhaps the best that can be said is that the early, largely unqualified optimism for bioprospecting, cannot be sustained, at least until the assumptions about models and parameter values are better developed.

7 The Global Environment Facility

One of the major innovations in global financing of environmental conservation is the United Nations Global Environment Facility (GEF). 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), the Stockholm Treaty on Persistent Organic Pollutants and the Convention to Combat Desertification. The implementing agencies were initially the World Bank, the United Nations Environment Programme (UNEP) and the United Nations Development Programme (UNDP), with various other agencies being given similar powers later on.

The basic idea of the GEF is that it should assist in financing activities in developing countries and economies in transition that would be of benefit to the global community but which the relevant countries would not undertake as part of their normal development activities. Put another way, the GEF seeks to internalise the 'global externality' arising from development activity. An example might be a coal-fired power plant that a developing country considers the cheapest option for meeting extra power demand. Coal has a high carbon content so contributes significantly to global warming. The role of the GEF would be to investigate alternatives to coal – e.g. natural gas, energy efficiency or even renewable energy. Since, *ex hypothesi*, coal is the cheapest option, the developing country needs an inducement to take on the additional or 'incremental' cost. By paying this incremental cost, the GEF secures the global benefit it was set up to secure. The parallel with a Coaseian bargain is obvious. Developing countries have sovereign rights to use their natural resources as they see fit, but the world as a whole has an interest in, and would

Table 4. 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 (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.

Table 5. Annual expenditures by selected market creation initiatives (US$ million p.a.)

Debt-for-Nature Swaps	Costa Rica Forest Law	Bio-prospecting	GEF Biodiversity	GEF All areas
140	20	Small	315	1000

benefit from, their conservation. The ‘polluter-pays’ principle fails because of the global pervasiveness of the externalities, sovereign rights and the poverty of the polluters. Hence, the ‘beneficiary-pays’ principle is invoked.

Table 4 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. However, one of the central issues is the extent to which both GEF funds and the co-financing sums are truly ‘additional’ as is required by the Rio Conventions on climate and biodiversity. Since official development assistance from rich to poor countries has declined significantly in recent years, from around $60 billion in 1990 to $54 billion in 2000, one argument is that GEF replenishments and co-financing have been paid for by reductions in other forms of development assistance.

Table 4 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 5 shows the comparison for biodiversity, although there are problems of separating out the biodiversity component in GEF expenditures because biodiversity is often the beneficiary of non-biodiversity focal areas such as international waters.

The final question regarding the GEF is: does it work? The GEF’s second Overall Performance Study (GEF, 2002) concluded that ‘GEF-supported projects have been able to produce significant results aimed at improving global environmental problems’. There have been ‘significant reductions of ozone depleting substances’, GEF has been ‘very effective in promoting energy efficiency,’ has achieved ‘some success in promoting grid-connected renewable energy’ and ‘has steadily improved the management standards for protected areas’. What is not known is the extent to which GEF funds have been truly additional rather than diversions from financial flows part of which might have gone to these focal areas anyway. Nor are there any cost-benefit appraisals of GEF interventions, perhaps because of the difficulties of measuring benefits outside of the climate change area. The GEF has largely eschewed monetary valuation but it should be very feasible to conduct cost-benefit studies on climate projects because of the availability of shadow prices for carbon dioxide (Pearce, 2003). Moreover, the GEF is 'finance driven' rather than 'goal driven'. That is, for an institution with fixed core funds, flexible leveraging of other funds, but no environmental targets, the subjection of investments to cost-benefit appraisal would appear to be essential. Otherwise it is hard to see how the efficient use of its funds can be tested.

The other observation about Table 5 is that debt-for-nature swaps are more significant than might at first appear, with finance running at a little under one-half of GEF annual biodiversity flows. Again, however, cost-benefit appraisals of swap initiatives appear not to be available, raising the issue of their efficiency. How do these market creation expenditures compare to global biodiversity conservation expenditure under the traditional model? The latter come to perhaps \$10 billion per annum (Simpson, 2003), suggesting that traditional expenditures are some 6-7 times market creation expenditures. Rather than see this as confirmation of the prevalence of the 'old' conservation model, it is perhaps more significant that the multiple is this low. Market creation has, as noted, only been up and running for a decade or so.

8 Conclusions

How far can beneficiary-pays market creation save the world's environment? In this paper we have suggested a number of avenues.

First, unlike polluter-pays market creation, which relies upon some local, national, regional or global regulatory target being met, beneficiary-pays market creation may or not be associated some kind of regulation. In its purest 'Coaseian' form, beneficiaries and asset owners simply bargain with each other to produce an optimal outcome. Indeed, this is one of the theoretical attractions of this form of market creation – it minimises the 'heavy hand' of government intervention. In practice, beneficiary-pays systems often do involve government if only because bargains entail institutional requirements such as contract, monitoring and enforcement. Nonetheless, the extent of government involvement tends to be less than with alternative forms of conservation regulation.

Second, whereas polluter-pays market creation creates the market through regulation, the beneficiary-pays approach assumes that there are mutual gains to be had from facilitating beneficiary payments. In turn, this assumes that there are some significant values to be captured. We have seen that, historically, the notion that high economic values reside in the environment may have been exaggerated through over-optimistic claims about 'wealth in the wilds', largely because of a lack of appreciation of the underlying economics. All too often, non-economists have borrowed economic ideas and have misapplied them. Moreover, if the expenditures involved in market creation give some idea of the economic values in question, those values appear to be relatively trivial – a few dollars per hectare in debt-for-nature swaps, for example. Even the largest expenditures of all beneficiary-pays schemes, by the GEF, amount to only around $1 billion per year across all its focal areas. The few studies we have of willingness to pay for global environmental resources are not, however, out of line with these sums.

Third, offsetting these gloomy observations, it was noted that the market creation schemes in question almost certainly are subject to substantial free riding. The thought remains that more and better studies of global willingness to pay might unearth the relevant values. Perhaps more importantly, it is far from clear that what we know about these economic values captures the wider general services of insurance and knowledge, although we saw that bioprospecting has hotly debated pharmaceutical knowledge values. The suspicion, therefore, is that market creation has so far tried to capture the more tangible ecosystem services. This is a natural way for markets to develop. How far they can develop further to capture the other values remains to be seen.

Fourth, 'saving the world' does not involve opting for one model of conservation. If nothing else, market creation experiments are showing that there is a potentially wide menu of initiatives, which are not necessarily mutually exclusive. Perhaps the fault of the 'old' conservation model is that it assumed only one approach would work – the command and control paradigm. Moreover, market creation is still relatively new and its true value to date probably lies in what we have learned about how to devise mutually compatible incentive systems for cooperative behaviour. To this end, asking if market creation benefits exceed costs could be an unfair question. Nonetheless, it is hard to see how we will ever be able to determine if these initiatives are relatively good or bad without proper ex post economic appraisal. It is unnerving that even the climate-oriented interventions of the GEF have not been subjected to such analyses.

Finally, there are serious questions about the effectiveness of market creation initiatives. In the case of DfNSs, implicit per hectare valuations appear remarkably low at just a few dollars per hectare. Conservation appears to be 'too cheap'. This raised the suspicion that what is being conserved under these initiatives is land that is not under threat of conversion. Rather, the payments appear as 'top-up' revenues to make domestic management costs of protection more effective. This does not mean that DfNSs are ineffective; if they are addressing a risk of underfunding of management, rather than avoiding outright conversion of land, then they may still be doing a good job, assuming that domestic conservation expenditures do not contract because of expectations about DfNS funds. Moreover, we saw that conventional protected area subsidies from the USA rarely amount to more than a few dollars per *thousand* hectares. The same problem arose with the GEF and with the Costa Rica Forest Law. In the case of the GEF, the question is not so much about the effectiveness of the individual investments, although the point about ex post appraisal remains. It is more an issue of how the GEF funds are truly additional to other forms of aid. In the case of Costa Rica, payments per hectare are far more realistic in terms of preventing conversion. The doubt there is more subtle: how far are the payments consistent with the economic value of the ecosystem services provided?

That more questions have been raised than answered is perhaps not surprising for a subject matter that is relatively new. What can be said is that market creation initiatives of both the polluter-pays and beneficiary-pays kind have arisen precisely because of growing disillusion with the 'old' model of conservation. The initiatives therefore deserve to be expanded, varied and validated. It is perhaps too early to decide if they are 'saviour or oversell.'

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