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Cost-Benefit Analysis and the Environment
Recent Developments

Chapter 12

The Value of Ecosystem Services

Ecosystems function like other capital assets – they generate a flow of services through time, and the capital can be held intact if the services are consumed in a sustainable fashion. Moreover, any ecosystem tends to generate many such services. This chapter analyses ecosystems from this multi-functional perspective, making a clear distinction between the total value of the ecosystem as an asset and the value of small or discrete changes in its service flow. The valuation issues are illustrated with reference to the debated benefits of ecosystems as providers of genetic value for pharmaceutical research.

12.1. Ecosystem services

All life is embedded in various categories of *ecosystems*, where ecosystems are defined at life forms ("biota") and their abiotic environments. Thus, a forest or a wetland is an ecosystem, as are coral reefs, deserts, estuaries and rivers. All ecosystems generate *services* which are extensive and pervasive. Those services essentially maintain life on Earth so, in one sense, all ecosystem services are economic services – they have an economic value based on the benefits human beings receive from those ecosystems. An ecologist might select the following services as being of considerable importance, but would probably define them without necessarily having the focus on how humans benefit, which tends to be the economist's perspective. Here we select some services that have obvious human benefit. Ecosystems provide:

- Purification services: for example, wetlands filter water and forests filter air pollution.
- Ecological cycling: for example, growing vegetation takes in ("fixes" or "sequesters") carbon dioxide, and stores it in the biomass until the death of the vegetation, the carbon then being transferred to soil. Since carbon dioxide is a greenhouse gas, growing biomass reduces those gases in the atmosphere.
- Regulation: natural systems have interacting species such that pests are controlled through natural processes, reducing the need for artificial controls. Ecosystems may regulate watershed and weather behaviour, reducing risk of floods.
- Habitat provision: habitats are stores of biological diversity which in turn may be linked to processes that reduce the risks of ecosystem collapse ("resilience"), even apart from providing sources of food, scientific information, recreational and aesthetic value.
- Regeneration and production: ecosystems "grow" biomass by converting light, energy and nutrients. This biomass provides food, raw materials and energy. Ecosystems ensure pollination and seed dispersal take place, ensuring that the systems are themselves renewed. It is estimated that some 30% of the world's food crops are dependent on natural pollination.
- Information and life support. Ecosystems are the products of evolution and hence embody millions of years of information. This information has scientific value but is also a source of wonder and life support.

While much of the focus is on "natural" ecosystems, the reality is that few ecosystems are unmodified by human behaviour, either deliberately – as with conversion of forest land to agriculture, or inadvertently through pollution or the introduction of non-endemic biological species ("biological invasions"). The challenge for cost-benefit analysis is to secure some kind of measure of these various ecological-cum-economic values for both natural and semi-natural ecosystems. If such an exercise were possible, and was reasonably accurate, we would have a far better idea of what is being sacrificed when ecosystems degrade in face of constant threats to convert them to simpler, less diverse systems (*e.g.* homogeneous agriculture). In terms of Chapter 6, we would know more about the total economic value (TEV) of ecosystems.

In recent years considerable efforts have been made to identify these ecosystem service values (*e.g.* Daily 1997). There are also increasing efforts to gather some idea of the sum of the values of the individual services – for forests see Pearce and Pearce (2001), for wetlands see Brouwer *et al.* (1999) and Woodward and Wui (2001). In the case of forests, for example, progress has been made on measuring the economic values associated with timber and non-timber products, carbon sequestration and storage, recreation, and watershed regulation. Some limited progress has been made in estimating the non-use values of forests. Far more elusive are the informational values – although a lively debate exists on the value of genetic material in forests for pharmaceutical research (for a survey, see Pearce 2004c) – and the wider ecological values, especially "resilience" to shocks and stresses.

But there are major methodological issues to be addressed. First, it is not clear that the "bottom-up" approaches whereby each type of service is valued separately and then the values are added to get some idea of the TEV of the ecosystem, are capturing the "whole" value of the ecosystem. Put another way, the value of the system as a whole may be more than the value of the sum of its parts. Ecosystems have interactive processes, a variable potential to adapt to exogenous change, and the relevant changes are often non-linear (Arrow *et al.* 2000). The bottom-up valuation procedure could therefore be misleading. A small economic value for one service might suggest it could be dispensed with, yet its removal could reverberate on the other services through complex changes within the ecosystem. The second problem arises from non-linearity. A cost-benefit analysis that fails to account for thresholds, for example, might dictate the conversion of part of an ecosystem for more direct human use. The assumption would be that conversion of this part of the ecosystem would not affect the remaining ecological services. Non-linearity means that this assumption is suspect. The third problem is that there is both uncertainty about the nature of the services themselves and, even more so, about their interactions. Converting a natural system may therefore produce unanticipated effects. And those effects may be irreversible. Chapter 10 looked at one way of approaching this problem in terms of (quasi) option value. We return to this approach shortly.

It follows that "ecosystem valuation" is not a straightforward exercise and it seems fair to say that the literature has progressed only a limited distance in tackling these issues.

12.2. Marginal *vs.* total valuation

Economists are clear that when they value an environmental asset they are valuing a very small ("marginal") change in the asset, or a discrete change. In the former case, consumer and producer surplus (see Chapter 2) are negligible. In the latter case they need to be estimated using the valuation techniques described in Chapter 7-9. A moment's reflection shows that it is not sensible to speak of the "total" value of a type of ecosystem and even less sensible to speak of the total value of all ecosystems. Unfortunately, some of the recent literature on ecosystem valuation claims to do just this (*e.g.* Costanza *et al.* 1997; Sutton and Costanza 2002). To see the issues, consider Figure 12.1. 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.

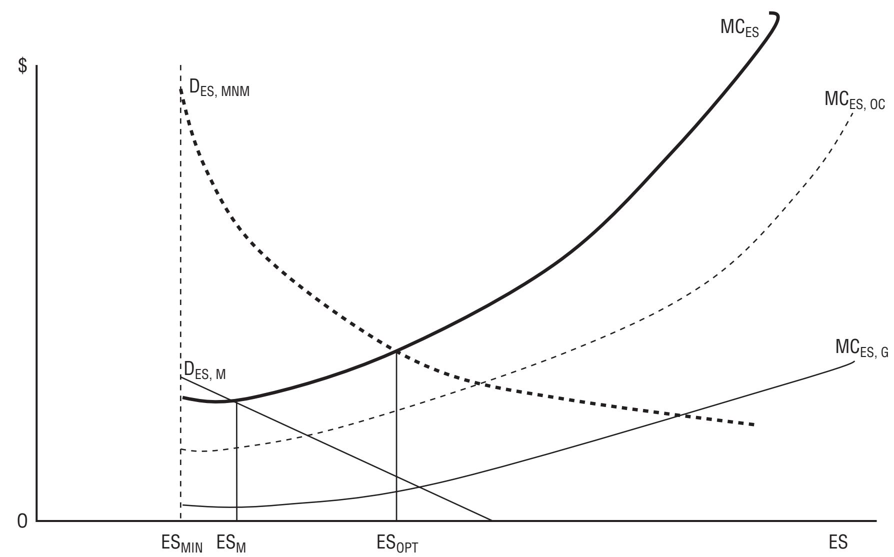
The first construct is a demand curve for ecosystem services $D_{ES,\,M}$. This is a demand curve for the *commercial*, or *marketed*, services of ecosystems, *i.e.* those services that have associated with already established markets in which formal exchange takes place using the medium of money. Thus, 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 (mWTP) because the curve shows how much individuals are willing to pay for incremental amounts of the good in question, ES. While it is tempting to think of D_{ES} as a demand curve for *all* services of *all* ecosystems, we see later than this is a risky interpretation. For the moment it is best to think of ES in Figure 12.1 as covering a single ecosystem, say tropical forests.

The second construct is another demand curve but this time for all services from the given ecosystem, 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. As noted above, 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. A widely quoted indicator of this appropriation is that of Vitousek *et al.* (1997) who estimate that humans already appropriate around 30-40% of the net primary product (NPP) on land. 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).

The two demand curves shown in Figure 12.1 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

Figure 12.1. **Stylised costs and benefits of ecosystem service provision**



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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, perhaps on the way to infinity. 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. Some suggest that at ES_{MIN} the demand curve would become infinitely elastic (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 Figure 12.1.

Left alone, ecosystems might continue to provide the same ES year after year. After all, they have been doing this for millennia. But in order to maintain ES of value to humans we know that certain costs are incurred. Figure 12.1 shows the first category of these costs as $MC_{ES,\,G}$ – the marginal costs of managing ES. In the absence of any 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 ESs. The assumption is that ESs are best secured by 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 12.1 is 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. It 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 development option. Thus some converted land becomes sterile, serving neither development nor ES purposes. In what follows we ignore these qualifications in order to focus on the basic messages from the analysis.

Figure 12.1 shows 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, and since the true global benefits of ES provision are given by the area under the $D_{ES,\,MNM}$ curve, 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 apparently *infinite* total benefits and this arises from the fact that the demand curve for ES is unbounded. As noted above, others may prefer to reformulate the issue and say that the idea of cost and benefit comparison for going below ES_{MIN} has no meaning.

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 12.1 shows the real possibility that failure to reflect true WTP in actual markets results in a serious under-provision of ES. Here 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.

Figure 12.1 can be used to explain why it is not possible to measure the *total* economic value of all ecosystems. This value would be the area under $D_{ES, MNM}$), but, as noted above, this area cannot be defined. If the view is taken that $D_{ES, MNM}$ becomes infinitely elastic at ES_{MIN}, then, the relevant area measuring total value would be infinite. This explains, perhaps, why one economist referred to Costanza *et al.*'s estimate of the total value as "a serious underestimate of infinity" (Toman 1998). Similar critiques of efforts to estimate the total value of all ecosystems, or even the value of a single global ecosystem, can be found in Pearce (1998b) and Bockstael *et al.* (2000).

The focus of ecosystem valuation, therefore, must be on small changes in the size or functioning of the ecosystems. However, if non-linearity is a serious issue, one should not rule out the possibility that small changes might lead to much larger levels of damage.

12.3. Finding ecosystem values

It is clear that ecosystems are "multi-functional" or "multi-product" – they generate an array of ecological-economic services. Unlike a multi-product firm, however, it was noted above that the "products" of ecosystems are usually not known with the level of certainty that would apply to a firm produces an array of market products. The products in question will also range from being purely private goods (*e.g.* fuelwood, clean water) through to being localised public goods (watershed protection) and finally to being global public goods (carbon sequestration and the non-use value of the ecosystem). Table 12.1 provides a simple typology to remind us of the array of products and services, and their probable associated property rights.

If the examples of products and services given in Table 12.1 were independent of each other, then, while the last of valuing changes in their provision would be huge, it is in

Table 12.1. **Economic characteristics of ecosystem products and services**

Examples	Private goods	Public goods		
		Local	Regional	Global
Forest	Fuelwood, water, rattan, food	Watershed protection	Air pollution reduction	Carbon storage and sequestration; Non-use values
Wetland	Fish	Soil erosion control	Storm protection	Carbon storage and sequestration; Non-use values
Corresponding property regime	Open access or common property	Open access or common property	Open access	Open access[1]

1. International agreements such as the Convention on Biological Diversity and the Framework Convention on Climate Change and its first Protocol (Kyoto) can be thought of as partial measures to turn global open access assets into global common property assets.

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principle something that can be done using the array of valuation techniques available. In the cases of forests and wetlands, for example, numerous studies exist on the individual services provided. The real difficulty, however, arises from the interdependencies between the services. In terms of valuation this means that the economic value of any one service may depend on its relationship to the other services. Recall that what valuation is doing is to value *changes* in the ecosystem, so the valuations are themselves dependent on how everything changes, not just the services we might want to focus on. (This is, incidentally, another reason why estimating "total" value is not feasible – as one, say, decreases the ecosystem dramatically, everything will change). As Arrow *et al.* (2000) note, this makes the task of valuing ecosystem services extremely complex, and it underlines the necessity to simplify simply to make valuation tractable. But simplification comes at a cost.

To summarise, the following issues arise in ecosystem valuation:

- Identifying ecosystem service and products in a context where we are usually uncertain about how ecosystems behave and what they "do".
- Focusing on marginal or discrete changes, not the value of the "total ecosystem".
- Determining the degree of irreversibility in ecosystem change.
- Establishing the geographical scope of the benefits generated, from local to global.
- Establishing the property rights regime for the resource in question.
- Valuing the products and services as if they are independent of each other.
- Analysing, in simplified form, the interactions between services to see, as far as possible, how this might modify the "sum of independent values" approach outlined above.

12.4. Valuing an ecosystem product: genetic information for pharmaceuticals

This section briefly reviews one set of attempts to place an economic value on one ecosystem product – the information that resides in tropical forests and which might be used to produce new drugs. Early excitement about the economic values embodied in forests arose primarily from the view that, since pharmaceutical companies have huge billion dollar sales of drugs based on natural materials, the value of those materials must similarly be huge. For example, world markets in products derived from genetic resources are estimated to be valued at USD 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.

But this approach does not conform to the relevant economic magnitude being sought: the willingness to pay to search for and utilise the relevant information. There are various factors that determine this willingness to pay. 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. Second, technological change is increasing the ability to 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. 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, especially Simpson *et al.* (1996), Craft and Simpson (2001), Rausser and Small (2000) and Costello and Ward (2003).

These studies correctly try to estimate the economic value of the *marginal species*, *i.e.* 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.

$$\max WTP = \frac{\lambda.(R-c)e^{\frac{-R}{R-K}}}{r(n+1)} \qquad [12.1]$$

where

λ = expected number of potential products to be identified = 10.52

n = number of species that could be sampled = 250 000

c = cost of determining whether a species will yield a successful product = USD 3 600

r = discount rate = 10% = 0.1

e = natural logarithm = 2.718

K = expected Research and Development cost per new product successfully produced = USD 300 million.

R = revenues from new product net of costs of sales but gross of R and D costs = USD 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 [12.1] gives a maximum willingness to pay (WTP) of USD 9410 for the marginal species. However, WTP for the marginal species is not a concept with which it is easy to identify. Accordingly, the literature tends to translate these values into WTP for land that is subject to the risk of conversion. This is done as follows. First, the "species-area" relationship is given by

$$N = \alpha A^Z \qquad [12.2]$$

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

$$V[n(A)] \qquad [12.3]$$

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Equation [12.3] refers to the value 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 [12.3]:

$$\frac{\partial V}{\partial A} = \frac{\partial V.\partial n}{\partial n.\partial A} \qquad [12.4]$$

The expression $\frac{\partial V}{\partial n}$ is the marginal value of the species, for example the USD 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 [12.2] gives:

$$\frac{\partial n}{\partial A} = Z\alpha A^{Z-1} = \frac{Z.n}{A} = Z.D \qquad [12.5]$$

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

$$\frac{\partial V}{\partial A} = \frac{\partial V}{\partial n}.Z.\frac{n}{A} \qquad [12.6]$$

or, simply, the value of the marginal species multiplied by 0.25 multiplied by the density of species.

The resulting values derived by Simpson *et al.* are given in the second column of Table 12.2. The overwhelming impression is of the very small values that emerge. The essential reasons for the low values are *a)* that biodiversity is abundant and hence one extra species has low economic value; *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.

Table 12.2. **Estimates of the pharmaceutical value of "hot spot" land areas**

Max WTP USD per hectare

Area	Simpson *et al.* (1994) WTP of pharmaceutical companies per ha	Simpson and Craft (1996) "Social value" of genetic material per ha	Rausser and Small (1998a) WTP of pharmaceutical companies per ha
Western Ecuador	20.6	2 888	9 177
South-western 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
South-western 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

Source: Simpson *et al.*, 1996; Simpson and Craft, 1996; Rausser and Small, 2000.

The third column of Table 12.2 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 USD 111 billion in net present value terms. The policy implications of the earlier work by Simpson *et al.* are modified to some extent by the Simpson and Craft work. Whereas economic values of (effectively) zero to USD 20 per hectare are extremely unlikely to affect land conversion decisions, the larger "social" values could be relevant to changing land use in some areas. 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 that allow 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.* 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 is challenged by Rausser and Small (2000). The fourth column of Table 12.2 shows Rausser and Small's 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, argue Rausser and Small, is that an equation like [12.1] 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 USD 9000/hectare in our simulation – can be large enough to motivate conservation activities". The basic difference, it appears, is that the Rausser-Small has "informed search" while the Simpson *et al.* models have "random search".

Costello and Ward (2003) test for the likely differences in value from informed search as compared to random search by conducting a numerical experiment. Their finding is that the Rausser-Small values 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. This suggests that the differences in the estimates have very little to do with the search assumption. Rather, it is assumptions about parameter values that mainly explain the differences. For example, Equation [12.2] above set $Z = 0.25$ where Z is the exponent in the species-area relationship. But Rauuser-Small have an implicit assumption that $Z = 1$. Similarly, the value of n (the number of species) is far higher in Simpson *et al.* than in Rausser and Small, lowering the value in the former case and raising it in the latter.

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-Small 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.*, 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 now appears to be highly dependent on models and parameters. 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. But the analyses to date also show just how difficult it is to estimate ecosystem values, even without allowing for the various kinds of ecological interdependence discuss earlier.

12.5. Actual and potential economic value

Ecosystems are self-evidently important, so important that without them human and other life would not exist. The economic issue is one of measuring what is being lost when parts of given global ecosystems are lost or degraded. The central problem is one of uncertainty – the basic fact is that we do not know what these losses are likely to be. Efforts at valuation are therefore important but are unlikely to inform us of the scale of "tolerable" change. Moreover, if decisions are made and they turn out to be extremely costly, little can be done to reverse them. Finally, if ecologists are right and the systems have thresholds and other non-linearities, maybe the consequences of losing even modest ecosystem areas could be large. Ecosystem loss thus combines several features:

- A potential large "scale" effect;
- irreversibility;
- uncertainty.

Economists have long known that this combination dictates a "precautionary" approach (*e.g.* Dasgupta 1982). To these features we need to add another:

- Few ecosystems undisturbed by human activity exist.

The relevance of this last point is that the world no longer has a "reserve" of ecosystems subject only to natural variation and to which it could turn for genetic and other information. In effect, the information stored over millions of years of evolution is at risk. Moreover, the impacts of human intervention in these systems are not known. One reason for this is that intervention may appear to leave the ecosystem "intact", say in terms of geographical coverage, but may change the species composition of the system. In particular, interventions frequently reduce the *diversity* of the system. It is widely argued that ecosystem productivity – the amount of biomass generated within an ecosystem – depends on that diversity, and that the resilience of the ecosystem to shocks and stresses also depends on diversity (Tilman and Polasky, n.d.). The implications for ecosystem valuation are that the goal of maximising the economic value of ecosystems might be served by not just "conserving" ecosystems but by managing them for their diversity. Rather like an economic "production function" ecosystems as they are may not be producing maximum economic value. Arguably, if undisturbed by humans and left only to the forces of natural variation, "ecosystem worth" would be maximised. But since nearly all ecosystems are not undisturbed, there is likely to be a potentially large degree of "inefficiency" in the services they do provide. In terms of valuation, care needs to be taken to value the potential rather than what is actually generated.

12.6. Cost-benefit analysis and precaution

Chapter 10 observed that there are two ways in which to conduct CBA. The first approach – the one that is most commonly used – operates either in a world of low uncertainty or in a context of uncertainty where the appropriate decision might be made in terms of expected values. The second takes more account of uncertainty and also takes explicit account of irreversibility either because funds committed cannot be "uncommitted" or because other effects of the policy cannot be reversed (or both). This was described as the "real options" approach to CBA. On the real options approach considerable attention would be paid to the opportunities for learning, and thus reducing uncertainty, by delaying irreversible decisions. It seems clear that the entire issue of ecosystem change fits the real options approach: there is uncertainty, irreversibility and a major chance to learn through scientific progress in understanding better what ecosystems do and how they behave. It is in this sense that real options give rigorous content to a notion like "the precautionary principle". Note that, on this interpretation of the precautionary principle, there would be far more caution about losing ecosystems, but benefits and costs would still be traded off.

Another contender for a precautionary approach would be the "safe minimum standard" (Ciriacy-Wantrup 1968; Bishop 1978). On this approach ecosystem conversion or loss would not be countenanced unless the opportunity costs – *i.e.* the value of the forgone "development" – were intolerably high. What the safe minimum standards approach does is to reverse the onus of proof, away from assuming that development is justified unless the costs to the environment are shown to be very high, to a presumption that conservation is the right option unless its opportunity costs are very high. But determining what is meant by "intolerable costs" is not easy. The level of "tolerance" might be determined by the political process, by reference to some notional benchmark – perhaps a percentage of GNP, or by a more extreme indicator – *e.g.* the forgone development causes severe hardship or poverty.

Finally, others argue that the precautionary principle acts like the strong sustainability principle discussed in Chapter 16. To anticipate the concept, it argues that no further degradation or loss of ecosystems would be tolerated. In a very extreme form it would argue that no existing ecosystem should be degraded. In less extreme form it would argue that any loss has to be offset by the creation of a like asset.

Thus "precaution" could enter into decision-making in several ways:

- As a strong sustainability constraint. In this case CBA remains valid but it operates within this sustainability constraint – see Chapter 16.
- As a safe minimum standard. In this case the trade-off between costs and benefits still exists but, in effect, a substantial premium is added to the benefits of conservation of ecosystems. Put another way, the benefit-cost ratio for deciding to degrade or lose the ecosystem is much higher than unity.
- As an option value approach. In this case the development option must be debited with the potential forgone costs of not waiting to learn more about the conservation benefits.

12.7. Summary and guidance for decision-makers

Research into the value of ecosystem services has evolved to the point where efforts are being made to estimate the total economic value of ecosystem change. This needs to be distinguished from misconceived efforts to value "all" ecosystems. The problems with valuing changes in ecosystem services arise from the interaction of ecosystem products and services, and from the often extensive uncertainty about how ecosystems function internally, and what they do in terms of life support functions. Considerable efforts have been made to value specific services, such as the provision of genetic information for pharmaceutical purposes. But even that literature is still developing, and it does not address the interactive nature of ecosystem products and services.

Once it is acknowledged that ecosystem functioning may be characterised by extensive uncertainty, by irreversibility and by non-linearities that generate potentially large negative effects from ecosystem loss or degradation, the focus shifts to how to behave in the face of this combination of features. The short answer is that decision-making favours precaution. But just what precaution means is itself a further debate. The suggestions here are that real options (Chapter 10), safe minimum standards and strong sustainability (Chapter 16) are all contenders.

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