Cost-Benefit Analysis and the Environment
Further Developments and Policy Use

PART III

Chapter 13

Ecosystem services and biodiversity

The valuation of ecosystem services has become a crucial element (perhaps the crucial element) in quantifying the contribution of ecosystems and biodiversity to human well-being. While the evidence base is broad and – at least for some ecosystem services – deep, reflections on this progress indicate a need for greater understanding of ecological production, especially as it relates to spatial variability and complexities in the way that services are produced. This is a truly interdisciplinary given the need for natural science to inform the stages of this analytical process. There is considerable debate remaining also about how to conduct decision analyses in those contexts where valuation and understanding of the natural world is likely to remain relatively uncertain. Such challenges need to be viewed in context. A growing number of large-scale ecosystem assessments has shown how the empirical record can be put to use in an informative and policy-relevant way. Such developments could be crucial in translating valuations into meaningful policy analysis.

13.1. Introduction

The valuation of biodiversity and ecosystem services is increasingly seen as a crucial element of robust decision making. An impetus for this has been large-scale "ecosystem assessments", which have helped clarify the way in which ecosystems contribute to human well-being. This is an antidote then to past practice which has all too often given cursory consideration, or even completely ignored, this link between nature and well-being in policy analyses.

The application of economic valuation techniques to the complexities of the natural environment raises a number of significant challenges. Perhaps most fundamental is the need to ensure that such applications are based upon a sound foundation of natural science.[1] This requirement for interdisciplinarity is given a conceptual framework within the so-called "ecosystem service" approach to decision making. While typically characterised as emanating from the natural sciences, the approach is highly compatible with economic analysis as it emphasises the role of ecosystems in providing services which, in turn, either support production or are direct contributors to well-being. Ecosystem services are therefore defined as contributors to anthropocentric values and while the natural sciences provide an understanding of the former, it is economics which is well placed to assess the latter. Economic valuation, in particular, becomes an essential element of the ecosystems service approach to decision analysis.

While the term "ecosystem services" is relatively recent, at least in the scheme of things having only being popularised in the wake of the Millennium Ecosystem Assessment (MA, 2005), environmental economists have been applying non-market valuation techniques to such services for many years (see, for example, Adamowicz et al., 1994; Ruitenbeek, 1989). Understanding the economic value of ecosystems is important for a number of reasons. One of these is undoubtedly the perceived persuasiveness of economic language. For example, Bateman et al. (2011b) estimate that, in the United Kingdom, ecosystem services help contribute to 3 billion outdoor recreational visits annually with the social value of the output created by these trips likely to be more than GBP 10 billion. Gallai et al. (2009) calculate the global value of the services provided by insect pollinators to be about USD 190 billion (in 2005) just in terms of the benefits arising from pollination of crops for (direct) human consumption. Thus, conveying what it is that the natural world provides us with in monetary terms is a powerful means of communicating the importance of conservation to a wider (and perhaps previously unreceptive) audience.

But beneath the rhetoric there is genuine substance in that these data can also be used to guide policy thinking and decisions. In the case, for example, of the recreational value of UK ecosystems, Bateman et al. (2011b) also show that how location (of these sites) matters. This is also pointed out in Wilson et al. (2014). A specific and moderate sized nature recreation site, for example, might generate values of between GBP 1 000 and GBP 65 000 per annum depending solely on where it is located. The critical determinant of this range is perhaps not surprisingly proximity to significant conurbations. Put another way, woodlands in the "wrong" place (i.e. relatively far from potential visiting populations) are unlikely to give rise to such high social values (other things being equal), an insight of particular importance if policy makers are contemplating new investments in these nature sites.

More generally, the key insight in explicitly placing a value on nature is that it redresses a fundamental imbalance whereby this value is – all too frequently – grossly misjudged or just plainly ignored in private and (much of) social decision-making. Demonstrating that nature has significant value for human livelihoods or human well-being more broadly is a crucial practical step in developing policy actions that address current and projected rates of ecosystem destruction and biodiversity loss. One much cited example, in this respect, is Barbier (2007). That study estimates the ecological value of mangroves in Thailand – in terms of providing fuelwood, a habitat that supplies fisheries and storm water attenuation (which reduces the risks of coastal flooding) – in order to compare those findings with the returns from the competing land use activity of shrimp farming. Thus, private profits under these two different uses are USD 584 and USD 1 220 per hectare respectively, giving, on the face of it, a clear (financial) case for mangrove conversion. However, social cost-benefit analysis reveals another story in that a representative hectare of mangrove is shown to generate a social value of USD 12 392.

Of course, the economic approach may not always provide the answer that ecosystems should be protected (and thus indicates the pitfall for those who see only the rhetorical worth in economic arguments). There is also a concern about the challenge of demonstrating the importance of ecological fundamentals – notably, "biodiversity" – in these assessments of the instrumental value of nature. And debates about the intrinsic value of nature remain relevant too. Nevertheless, and however the question is posed, determining how much of nature "ought to be" conserved is likely to require a significant effort to understand its value in economic terms as well as the (opportunity) costs of its conservation. The challenges are immense. While many of these are not insurmountable (as the growing evidence-base suggests), as is also discussed, in what follows, there is an inevitability to limits on valuing nature. How to do economic appraisal whenever these limits are reached is also an important question as illustrated in Chapter 12.

13.2. Ecosystem services

All life is embedded in various categories of *ecosystems*, where ecosystems are defined as 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. For example, the following indicates some services that have obvious human benefits. 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 the concentration of 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.

One starting point for marrying notions of ecosystem service from the natural sciences to the requirements of economics is classification systems. There are a number of variations on these classifications. Common to almost all is a distinction between: provisioning services; cultural services; and regulating services. The former two services nicely capture some elements of the previous distinction between use and non-use (see Chapter 4). Provisioning services, for example, are typically physical products such as food and natural materials provided by nature. Cultural services, by contrast, describe the experiences that people enjoy as a result of interactions with nature (e.g. recreation), as well as more intangible pleasures arising from knowledge about the existence of nature or its spiritual value. Of course, while these services can be thought as being distinct for the purposes of classification, ecosystems might provide "goods" which fulfil both provisioning and cultural criteria (see, for a detailed discussion, Chan et al., 2011) as well as different types of cultural benefit: i.e. use and non-use. For example, a woodland might be valued both for its provision of recreational opportunities as well as for the knowledge that this natural area continues to be conserved or provided even if the person expressing the value does not observe directly this outcome. Table 13.1 provides an example of this classification, drawn from Markandya (2016) and based on ongoing work to classify ecosystem services for the purposes of ecosystem accounting.

Further classifications of ecosystem services do exist. Kumar (2010), for example, add habitat services in recognition of the role that ecosystems provide in protecting "gene pools" as well as crucial sets of interlinking habitats for migratory species. MA (2005) also emphasised the supporting services of ecosystems as the natural processes that underpin those services of provision, culture and regulation. These services, such as nutrient cycling, thus provide a further intermediate tier to ecological production and, indeed, it has since become more common to see these functions subsumed under the "regulating services" heading (e.g. Kumar, 2010). Other classifications, such as Heal et al. (2005) and de Groot et al. (2002), have focused more specifically on habitat services and regulating services.

While this emphasis is partial, it encapsulates a key distinctive element of the effort to understand the economics of ecosystems. This likens the enjoyment of (final) ecosystem services to a process of (natural) production whereby critical inputs are, for example, regulating services. As an illustration, it is these services – by e.g. regulating water flow (and the quality of that water) and the supply of insect pollinators – that

Table 13.1. **Classification of ecosystem services**

Section	Division	Group	Class
Provisioning	Nutrition	Biomass	Cultivated Crops
			Reared animals and their outputs
			Wild plants, algae and their outputs
			Wild animals and their outputs
			Plants and algae from in-situ aquaculture
			Animals from in-situ aquaculture
		Water	Surface water for drinking
			Groundwater for drinking
	Materials	Biomass	Fibres and other materials from plants
			Plants, algae, animals materials for agricultural
			Genetic materials from all biota
		Water	Surface water for non-drinking purposes
			Groundwater for non-drinking purposes
	Energy	Biomass based energy	Plant-based resources
			Animal-based resources
		Mechanical based	Animal-based energy
Regulation and Maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Bioremediation by micro-organisms etc.,
			Filtration/sequestration/storage/accumulation by micro-organisms etc.,
		Mediation by ecosystems	Filtration/sequestration/storage/accumulation
			Dilution by atmosphere, freshwater, marine ecosystems
			Mediation of smell, noise, visual impacts
	Mediation of flows	Mass flows	Stabilisation and control of erosion rates
			Buffering and attenuation of mass flows
		Liquid flows	Hydrological cycle and water flow maintenance
			Flood protection
		Air Flows	Storm protection, ventilation and transpiration
	Maintenance of physical, chemical, biological conditions	Habitat and gene pool protection	Pollination and seed dispersal
			Maintaining nursery populations and habitats
		Pest and disease control	Pest control
			Disease control
		Soil formation and Composition	Weathering processes
			Decomposition and fixing processes
		Water conditions	Chemical condition of fresh and salt waters
		Atmosphere and Climate regulation	Global climate regulation by reducing GHGs
			Micro and region climate regulation
Cultural	Physical and intellectual interactions with biota/ecosystems	Physical and experiential	Experiential use of plants, animals landscapes
			Physical use of land/seascapes in different ways
		Intellectual and representative interactions	Scientific, educational, heritage/cultural, entertainment and aesthetic interactions
	Spiritual, symbolic interactions with biota/ecosystems	Spiritual and/or emblematic	Symbolic
			Sacred and/or religious
		Other cultural	Existence
			Bequest

Source: Markandya (2016).

contribute ultimately to the production of agricultural provisioning services (Goulder and Kennedy, 2011). Valuing ecosystem services has often focused on the end output, by asking what is the final service that ultimately benefits people. Clearly, knowledge of what ecosystems provide as final goods and services that is being consumed is important. Yet it is equally crucial to understand the way in which intermediate tiers of production contribute to this final output.

In many of these classifications, there appears to be no explicit place for the value of *biodiversity*. Indeed, a significant anxiety about recent ecosystem assessments is that the emphasis upon ecosystem services might ironically lead to the omission of the vital role which biodiversity plays in both the delivery of those services and as a source of value in itself. On the one hand, biodiversity can be thought itself of as a service. For example, pollinator biodiversity directly enhances agricultural production. Certain aspects of biodiversity, such as the continued existence of iconic species such as the polar bear, itself constitutes a good (i.e. a direct source of well-being). On the other hand, Mace et al. (2012) warn that exclusively focusing on that role risks missing something fundamental. As discussed in detail by Elmqvist et al. (2010), biodiversity acts as a supporting service underpinning the delivery of what Fisher et al. (2009) term final ecosystem services. So, for example, soil biodiversity enhances farmland fertility which in turn determines production of a good (here food). In fact, such functions provided by biodiversity have been likened by, for example, Pascual et al. (2010) to a form of insurance (following from earlier contributions such as Gren et al., 1994).

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 producing 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).

Initial clues in the search for practical ecosystem values can be found by reflecting on how ecosystem services ultimately provide benefits to people and businesses. This was discussed in Chapter 2 and is what Freeman et al. (2013) term: “The economic channel through which well-being is affected” (p. 13). These channels are manifold (e.g. Brown et al., 2007; Freeman et al., 2013) but can be summarised in three ways.

- First, there are those ecosystem services which are used as inputs to economic production. Examples include soil fertility which is an input to agricultural production. Water regulation and water purification services are inputs to those economic (producing) units which need a supply of clean water as an input, perhaps alongside e.g. other factors of production.
- Second, ecosystem services can act as joint inputs to household consumption. That is, there is use of ecosystem services in combination with expenditure on produced goods and services in providing a “product” for consumption. In such cases, an ecosystem services and the market goods or services are complementary inputs. Examples include nature services which in combination with travel expenditures are used to produce nature recreational experiences. However, an ecosystem service can be a substitute for a market good. An example is air purification services which can substitute for purchase of a produced good which filters air.

- Third, ecosystem services can be inputs which directly contribute to household well-being. That is, there is no existing economic production or household consumption where these services act as inputs. These services are consumed directly in generating benefits (which themselves are ultimately a source of well-being). Examples here are by their nature rather abstract, but include those services valued for reasons surrounding what is usually termed "non-use" or "passive-use", such as "true wilderness".

13.3. Valuing ecosystem services

Uncovering the true value of goods and using these data to ensure decisions contribute to improving human well-being is a defining rationale for economic analysis. A number of recent comprehensive reviews make clear the proliferation of methods – and applications of those methods – to assess the value of ecosystem services and biodiversity (see, for example, Pascual et al., 2011; US EPA, 2009; Bateman et al., 2011b; Kaveira et al., 2011; as well as Chapters 3 to 7 in this volume). These assessments have been important for revealing, on the one hand, what is known about ecosystem and biodiversity valuation and, on the other hand, in identifying what remains to be learnt. Table 13.2 provides a brief overview of the key approaches. What is important to note here is that *all* of these methods have been used in the ecosystems context. In large part, this breadth of methods reflects, in turn, the diversity of services that practitioners have sought to value rather than variety for its own sake.

The starting point for thinking about the valuation of ecosystem services is that such assessments rely upon standard economic theory but with an underpinning by the natural sciences (Daily, 1997; MA, 2005; Pagiola et al., 2004; Heal et al., 2005; Barbier, 2007; Sukhdev, 2008). Whether this valuation can be based on market prices or whether the analyst must look to evidence from non-market behaviour (be this actual or intended) depends on the characteristics of the ecosystem good or service in question. In some cases, valuation might begin with market prices. For example, provisioning services, such as food and fibre, are frequently market goods or near-market goods with close (market) substitutes. It follows, therefore, that market-based valuation has been prominent in such contexts, although perhaps these observed prices need to be adjusted for distortions (Table 13.2). However, the provisioning service is itself typically determined by some underlying service provided by an ecosystem process. Thus, while the valuation of this final output is relatively straightforward, the analytical heavy-lifting is often done through the specification and estimation of an ecological production function. In other words, ecosystem services frequently are valued as a productive input (see Barbier, 2007; Freeman, 2003; and Hanley and Barbier, 2009). In this approach, an attempt must be made to isolate and uncover the value of ecosystems services from the perspective of their effect on some observed level of output (Table 13.2). This approach can be applied to a range of market (consumption) goods but has also been used for valuing regulating and "protection" goods (where examples of the latter include flooding and extreme weather protection).

In other cases, however, the value that people place on ecosystem services is not adequately reflected in market prices, if at all. In such cases, non-market valuation techniques must be employed and applied to some ecological end-point which itself may have been estimated following some application of a production function. Revealed preference methods value non-market environmental goods by examining the consumption of related market-priced private goods. A number of variants of the revealed preference approach exist, depending on whether the environmental good and the related market good are complements, substitutes or one is an attribute of the other (Table 13.2).

Table 13.2. **Summary of economic valuation methods used in ecosystem service valuation**

Valuation method	Description	Typical applications to ecosystem services
Adjusted market prices	Using market prices adjusted for any distortions (e.g. taxes, subsidies, non-competitive practices)	Crops, livestock, woodland
Production function methods	Estimation of an ecological production function where the ecosystem service is modelled as an input to the production process and is valued through its effect on the output	Maintenance of beneficial species, maintenance of agricultural productivity, flood protection
Revealed preference methods	Examining actual expenditures made on market goods related to ecosystem services. When market goods are substitutes, avertive behaviour or mitigating expenditure approaches can be used (e.g. expenditures to avoid damage, such as buying bottled water or installing double glazing). Travel cost methods can be used when market goods are complements, (e.g. travel costs for recreation). When the ecosystem service is a characteristic of the market good hedonic price methods can be used (e.g. looking at the impact of noise or amount of green space on property prices)	Water quality, peace and quiet, recreation, amenity benefits
Stated preference methods	Using surveys to elicit willingness to pay for an environmental change (contingent valuation) or to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay (choice modelling)	Water quality, species conservation, air quality, non-use values
Subjective well-being (SWB) methods	Uses survey responses of measures of SWB, and investigate the extent to which ecosystem-related metrics are determinants of well-being. Valuation might entail looking at the income/ecosystem trade-off in reaching a given level of SWB	Water quality, species conservation, air quality depending on the availability of suitable metrics

In the first case, economists make use of the "weak complementarity" concept introduced by Mäler (1974) to examine how much individuals are prepared to spend on a private good in order to enjoy the environmental good, thereby revealing the value of the latter. For example, the travel cost method examines the expenditure and time that individuals are prepared to give up to visit natural areas for recreation. In cases of substitutability between goods, approaches such as avertive behaviour or mitigating expenditures to avoid damages can be used, such as buying bottled water to avoid drinking contaminated water. Finally, the hedonic property price method assumes that one can look at the housing market to infer the implicit value of the underlying characteristics of domestic properties be these structural, locational/accessibility, neighbourhood or environmental (Rosen, 1974). It can be used for example to examine the premium which people are prepared to pay in order to purchase houses in areas with greater proximity to green spaces or habitat types (Gibbons et al., 2011).

While revealed preference methods estimate original values by looking at *actual* behaviour, eliciting values by looking at *intended* behaviour is the province of stated preference (SP) methods. This is an umbrella term for a range of survey-based methods that use constructed or hypothetical markets to elicit preferences for specified changes in provision of environmental services (Table 13.2). By far the most widely applied SP technique is the contingent valuation method (see, for example, Alberini and Kahn, 2006).[2] However, in recent years, choice modelling has become increasingly popular. In this variant, respondents are required to choose their most preferred out of a (possibly relatively large) set of alternative policy or provision options offered at different prices and their willingness to pay is revealed indirectly through their choices (see, for example, Hanley et al., 2001; Kanninen, 2007).[3]

In theory, SP approaches should be applicable to a wide range of ecosystem services and can be used to measure future/predicted changes in those goods. Importantly, such methods are thought to be the only option available for estimating those services which are valued for "non-use" purposes. In practice, SP methods are mostly defensible in cases where respondents have clear prior preferences for the goods in question or can discover economically consistent preferences within the course of the survey exercise. Where this is not the case, then elicited values may not provide a sound basis for decision analysis. Such problems are most likely to occur for goods with which individuals have little experience and poor understanding of (Bateman et al., 2008a, 2008b and 2010). Therefore, while stated preferences may provide sound valuations for high-experience, use value goods, the further one moves to consider indirect use and pure non-use values, the more likely one is to encounter problems. Paradoxically then, where SP techniques are most useful is also where they have the potential to be less effective.

A number of solutions have been proposed for the problem of valuing low-experience goods. Christie et al. (2006) have proposed the use of intensive valuation workshops where participants learn about the environmental services being valued. However, the techniques involved are almost inevitably prone to reliance upon small unrepresentative samples which, after such intensive experiences, cannot be taken as reflecting general preferences. So while offering useful insights about overcoming the low-experience problem, it must be asked whether the cure is worse than the disease. Others have proposed and implemented extensions of conventional, individual based SP applications. Bateman et al. (2009), for example, use virtual reality software to convey images of landscape goods. This avoids the difficulties of conveying attributes of goods such as landscape in unfamiliar units, such as hectares. Results show a significant reduction in the rate of preference inconsistencies through the application of such techniques.

While significant strides can be made in filling out the ecosystem valuation matrix without recourse to what might be judged by some to be more "problematic methods", crucial gaps remain in the empirical record. This issue seems particularly acute in the case of many types of cultural ecosystem services. As stated by Chan et al. (2010, p. 206) "…few classes of value have been more difficult to identify and measure than those concerned with the cultural and non-use dimensions of ecosystems". Cultural ecosystem services include use-related values, such as leisure and recreation, aesthetic and inspirational benefits, spiritual and religious benefits, community benefits, education and ecological knowledge, physical and mental health. Difficulties arise as some of these cultural services may be bound up by non-use motivations such as altruistic, bequest and existence values (Krutilla, 1967).[4] Moreover, some of these benefits are also difficult to identify separately. As things stand there appears to be a generalised lack of knowledge and a specific dearth of monetary information about the contribution of cultural ecosystem services to well-being. The following sections therefore discuss some of the challenges with regards to the "health" and "non-use" values of ecosystems in particular.

Box 13.1. **Practical values for ecosystem services**

A range of these ecosystem services are presented in Table 13.3, from Markandya (2016), but based on an earlier synthesis of the empirical record by Groot et al. (2012). The table lists 22 ecosystem services in all classified by biome (very broad habitat types, both terrestrial and aquatic). The data there are presented in terms of the per hectare (ha) monetary value of an ecosystem service (expressed in terms of USD in 2007 prices).

A number of observations about the data are possible. First, the empirical record is incomplete most likely due to a combination of factors. It might be that in some cases a particular ecosystem service is insignificant for the particular biome. But it might be that data simply do not exist or the empirical record for that cell of the table is too thin to synthesise

Box 13.1. **Practical values for ecosystem services** *(cont.)*

in this way. Second, in other respects the table is remarkably full especially for certain biomes (notably for inland and coastal wetlands as well as tropical forests). This is perhaps striking given the novelty of this literature reflecting a lot of progress in a relatively short space of time. Third, the data in the table are suggestive of the importance of particular ecosystem services relative to one another and in the context of particular biome types. Moreover, in principle these per hectare values seem reasonably straightforward to apply to new policy questions. That is, if some amount of hectares of woodland are to be planted in some location what might be the expected change in ecosystem services that results? While not providing the exact answer (for reasons expanded on immediately below), the table gives a sense of how to think about answers to this question as well as indicating a summary of the evidence base that exists.

Just as important as these observations is a proper reflection on the issues that lay below the veneer of Table 12.3 and the subsequent "health warnings" that might be applied in interpreting these sort of data. For example, Markandya (2016) notes that these data are not necessarily always additive. Some regulatory ecosystem services are actually inputs to the generation of other provisioning ecosystem services. So the table while useful in a lot of a respects does not absolve the analyst from a fuller consideration of the stages of the natural and economic production processes whereby these ecosystem services enter. The details underlying these synthesis are important for other reasons too. Standardised per hectares values do not convey substantial spatial variation in ecosystem services especially where location really matters, as it does in this valuation context. There is no reason to believe that these data apply everywhere (and so may need adjustment) or are simply linear in the way that the table (implicitly) suggests. More generally, the table says nothing about the quality of the valuation studies that have been synthesised to arrive at this summary. Many of these issues are discussed in detail in a number of chapters elsewhere in this volume. For present purposes, it is important to note these considerations do not mean that Table 13.3 is of no practical use. Rather it makes clear that such values, while useful, need to be treated and used with care by analysts.

A final point relates once more to what the table misses. The challenge of missing ecosystem service value data has already been mentioned. Yet another issue is the emphasis on ecosystem services says little explicitly about the value of biodiversity, defined as by the Convention on Biodiversity as: "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexities of which they are part; this includes diversity within species, between species and of ecosystems". What this means is that, in the context of Table 12.3, typical ecosystem services at best only implicitly reflect the contribution of this biodiversity (the contribution of the richness, complexity and resilience of species and the ecosystems that they inhabit) , if at all. For example, Mace et al. (2012) caution that ecosystem services and biodiversity should be viewed simply as synonymous terms. Nor is biodiversity just a particular type of (final) ecosystem service (e.g. the provision of the wild species). Biodiversity, as stressed by Mace et al., is also a regulatory of ecosystem processes and so a fundamental building block of the sorts of ecosystem service values summarised in the table.

Table 13.3. **Summary of monetary values for each service by biome**

International dollars per hectare per year, 2007 price level

		Marine	Coral reefs	Coastal systems	Coastal wetlands	Inland wetlands	Rivers and lakes	Tropical forests	Temperate forests	Woodlands	Grasslands
	Provisioning services total	**102**	**55 724**	**2 396**	**2 998**	**1 659**	**1 914**	**1 828**	**671**	**253**	**1 305**
1	Food	93	667	2 384	1 111	614	106	200	299	52	1 192
2	Water				1 217	408	1 808	27	191		60
3	Raw materials	8	21 528	12	358	425		84	181	170	53
4	Genetic resources		33 048		10			13			
5	Medicinal resources				301	99	1 504				1
6	Ornamental resources		472			114				32	
	Regulating services total	**65**	**171 478**	**25 847**	**171 515**	**17 364**	**187**	**2 529**	**491**	**51**	**159**
7	Air quality regulation							12			
8	Climate regulation	65	1 188	479	65	488		2 044	152	7	40
9	Disturbance moderation		16 991		5 351	2 986		66			
10	Water flow regulation					5 606		342			
11	Waste treatment		85		162 125	3 015	187	6	7		75
12	Erosion prevention		153 214	25 368	3 929	2 607		15	5	13	44
13	Nutrient recycling				45	1 713		3	93		
14	Pollination							30		31	
15	Biological control					948		11	235		
	Habitat services total	**5**	**16 210**	**375**	**17 138**	**2 455**		**39**	**862**	**1 277**	**1 214**
16	Nursery services			194	10 648	1 287		16		1 273	
17	Genetic diversity	5	16 210	180	6 490	1 168		23	862	3	1 214
	Cultural services total	**319**	**108 837**	**300**	**2 193**	**4 203**	**2 166**	**867**	**989**	**7**	**26**
18	Aesthetic information		11 390			1 292					167
19	Recreation	319	96 302	256	2 193	2 211	2 166	867	989	7	26
20	Inspiration					700					
21	Spiritual experience			21							
22	Cognitive development		1 145	22					1		
	Total economic value	**491**	**352 249**	**28 918**	**193 844**	**25 681**	**4 267**	**5 263**	**3 014**	**1 588**	**2 871**

Note: Coastal systems include estuaries, continent shelf areas and sea grasses but exclude wetlands like tidal marshes, mangroves and salt water wetlands.
Source: Markandya (2016) – Adapted from De Groot et al., 2012.

13.3.1. Health values

Despite increased recognition that ecosystem services can have substantial effects on human health, both directly and indirectly, (e.g. Myers and Patz, 2009; Bird, 2007; de Vries, et al., 2003; Hartig et al., 2003; Mitchell and Popham, 2008; Osman, 2005; Takano et al., 2002; Ulrich, 1984) the knowledge on the complex relationships linking the biophysical attributes of ecosystems with the many aspects of human health remains limited (Daily et al., 2011)

Environmental quality and proximity to natural amenities is increasingly recognised as having substantial effects on physical and mental health, both directly and indirectly. Broadly this could arise in a number of ways. Ecosystems provide many services that sustain human health (such as nutrition, regulation of vector-borne disease or water purification). Also, natural settings could act as a catalyst for healthy behaviour, leading for example to increases in physical exercise, which affect both physical and mental health (Pretty et al., 2007; Barton and Pretty, 2010). Finally, simple exposure to the natural environment, such as having a view of a tree or grass from a window, can be beneficial, improving mental health status (Pretty et al., 2005) and physical health (Ulrich, 1984). Health outcomes in this respect can be disaggregated into two categories: reductions in mortality and reductions in morbidity (including physical and mental health).

While there is a large literature on health valuation, a crucial gap is in relation to the contribution of ecosystems to these improvements. Moreover, the statistical evidence for the health-ecosystem link is still to be established unequivocally. For example, on the link between physical exercise and availability of green spaces, the suspicion is that even if the physical health link can be more firmly established, the value is possibly likely to be small given the availability of substitutes for this physical exercise. Hence, it is likely to be the mental health benefit that is plausibly the more substantial of these two (bundled) health outcomes. Less is known with regards to valuation of these outcomes. However, it might be that subjective well-being approaches linked to monetary valuation are a promising path to explore further (see Chapter 7). A final but no less important challenge is to know what values are for *changes in* ecosystem provision whereas most work to date has examined the possible health benefits associated with *current* provision.

13.3.2. Non-use values

Environmental non-use values are often thought to be substantial (see, for example, Hanley et al., 1998). Critically, however, when and where these arise remains the subject of some discussion. Due to their intangible nature and disconnect from actual uses, the valuation of non-use benefits is complex. As a result, there appears to be no systematic body of evidence about non-use values and, importantly, little consensus about how the empirical record (such as it is) can be used for practical assessment in the context of project and policy appraisals or broader national-level ecosystem assessments. In the former, a particular concern might relate to whether a (change in a) non-use value relates to a specific and discrete proposal (or the provision of a service more generally). In the latter, a concern might be double-counting or erroneously assuming that the same (per household or individual) non-use value estimate applies to all of the parts rather than something more broadly resembling the whole. Put another way, the physical “unit” to which these non-use values can be applied is, on reflection, not at all obvious. Yet, given the possible importance of non-use value in certain ecosystem contexts, this issue surely merits further investigation.

One significant obstacle to addressing this challenge is that, as noted above, SP methods are often thought to be the only economic valuation techniques capable of measuring non-use values and so any doubts about the application of those methods or the accuracy of such valuations will loom especially large in this context. Challenges in the application of SP methods to non-use values are readily identified. Lack of experience and familiarity is likely to be important when respondents, for example, are asked about their preferences for conserving species which might well be located in distant lands. Related to this is the lack of adequate testing for preference consistency exhibited in many such studies (although, see for an exception, Morse-Jones et al., 2012, discussed in further detail below).

It may be, however, that other avenues for non-use valuation remain to be explored (although none appear to offer a general panacea for the challenges inherent in this endeavour). For example, legacies can be argued to represent a pure non-use value. That is, individuals leaving a charitable bequest to an environmental organisation in a will, for the purposes of supporting conservation activities, clearly will not experience the benefits of this work. Atkinson et al. (2009) estimate that while (in 2007) only 6% of all deaths in Britain resulted in a charitable bequest, their value remained substantial. And while legacies to environmental charities will be a relatively small proportion of this total, Mourato et al. (2010), for example, have estimated that this amounts to more than GBP 200 million in the financial year 2008/09.

Related to the notion of "non-use" is current interest in what has been termed "shared values" (see, for example, Fish et al., 2011). For some this appears to be unfinished business arising from earlier discussions about how people value environmental policy changes, more generally, as individuals or citizens (Sagoff, 1988). However, the concept has also been a way of conveying that there might also something extra to the value of an ecosystem over and above adding up different elements of its total economic value.[5] The emphasis on shared values traces this missing element of value to the way in which ecosystems have collective meaning and significance for communities of people related perhaps to "non-use" or perceptions about ecosystem aesthetics.

There is little obvious evidence to add empirical substance to these insights. However, the handful of studies that have sought to use deliberative monetary valuation approaches provide some practical understanding of the individual or collective value of certain proposed environmental changes in a group context. (e.g. Macmillan et al., 2002; Alvarez-Farizo et al., 2007). Investigating this notion of shared values for ecosystems through wider-scale testing than has been possible thus far is a possibly rich topic for further development. In such contexts, the "deliberative nature" of this valuation process (providing participants with information that they can reflect on) might also help mitigate problems of poorly informed consumers making valuation choices about possibly complex changes. There remains, however, an urgent need for a better understanding about the conceptual basis for "shared values" and, in particular, how it might be integrated within economic appraisal. For example, this might involve the recognition not only that i) the value of goods to an individual may differ radically from the value of the same good from a societal perspective, but also that ii) even these individual values are likely to be the product of social (and other) contexts.[6]

A readiness to understand "value" not just in terms of its economic interpretation can be seen in the work of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystems (IPBES). The IPBES was established in 2012 as an interface for science and policy for biodiversity and ecosystem services, and is administered by UNEP. While the work programme of the IPBES is wide ranging, its work on value is summarised in Pascual et al. (2017) and is firmly grounded in the "value plurality" discussed in Chapter 2. In other words, *instrumental value* – familiar in most economic applications, and the main focus of this chapter – is just one guiding principle that the IPBES considers as the value of nature's contribution to people.

This includes the notion of shared value, which Pascual et al. locate as a type of *relational value* defined as: "… values which do not emanate directly from nature but are derivative of our relationships with it and our responsibilities towards it" (p. 11) and envisaged as comprising cultural identity, social cohesion and shared moral responsibilities associated with ecosystems and biodiversity. The notion of *intrinsic value* is also explicit in this framework too: that is, values which are inherent in nature and independent of human experience and evaluation. In terms of the practical implications for appraisal of accommodating this mix of preferences and belief systems, this entails at the very least a strong degree of participative and deliberative approaches alongside "traditional" valuation techniques.

13.4. Valuation and policy appraisal

Much of the recent attention to the economics of ecosystems can be traced to the MA (2005) which made clear the scale of the challenge at hand in its identification of persistent and growing threats to ecosystems around the world. Importantly, the MA had the effect of broadening the focus of concern from biodiversity loss to cover, in addition, the loss of ecosystem services with the critical emphasis of the latter on "the benefits people obtain from ecosystems" (MA, 2005, p. 53). In addition, the focal valuation message in the Stern Review on Climate Change appears not to have been lost on decision-makers within the domain of conservation policy. Assessments including the G8/EU initiated "TEEB Review" (The Economics of Ecosystems and Biodiversity, TEEB, 2010) and the UK National Ecosystem Assessment (NEA, 2011)[7] can be viewed as attempts to generate a correspondingly increased awareness and strong policy response for biodiversity and ecosystem services as well as a concerted effort to build on the momentum and insights generated by the MA.

One of the largest ecosystem service valuation exercises conducted to date forms the core of the economic analysis underpinning the UK National Ecosystem Assessment (UK-NEA, 2011). This was based upon highly disaggregated, spatially sensitive, large observation databases, and provide decision makers with a rich and more holistic picture of the overall consequences of any given policy option. The advantages of such an approach were quickly realised by UK policy makers and the lessons of the UK-NEA were explicitly incorporated in the UK Natural Environment White Paper (Defra, 2011), published in the immediate aftermath of the former report. Such academic and policy developments suggest that the incorporation of value transfer techniques as tools for official policy formulation show promise. Notwithstanding this interim conclusion, there remains a need for tools capable of translating valuation information into policy action.

In the UK-NEA, value functions were estimated for multiple ecosystem services, including the provisioning value of agricultural food production, the regulating services of the environment as a store for greenhouse gases and the so-called cultural services of both rural and urban recreation (including urban greenspace benefits). Following Bateman et al. (2011a), the functions were simplified to focus upon the main – theoretically expected –

drivers of value, thereby avoiding the transfer of factors which only apply in a given context and are not general. The functions were also built in an integrated manner which linked the levels of each to the other. So, for example, if provisioning values are increased as a result of agricultural intensification, that same intensification feeds into an increase in greenhouse gas emissions and deterioration of rural recreation resources which result in a fall in both of these latter values.

An example of the output obtained from such analyses, Figure 6.5 in Chapter 6, illustrated findings from the UK-NEA analysis of rural recreation benefits arising from a change of land use from conventional farming towards multipurpose, open-access, woodland.[8] The distribution obtained by transferring a recreational value function across the entirety of Wales reflects various factors, including the distribution of population (this being highest in south western Wales and in the areas of England neighbouring the north-east) and the availability and quality of the road network. Such spatially disaggregated outputs allow decision makers to target resources in the most efficient manner; an ability that is clearly of great importance during times of austerity.

One further example of this challenge with respect to spatial variability can be illustrated with reference to those valuation studies that focus on the value of a representative unit (typically a km^2) of an ecosystem's area of extent. A naïve approach to aggregation might simply estimate total value as the product of this unit value and total ecosystem area. Barbier et al. (2008) illustrate the dangers of this where there is a non-linear relationship between ecosystem extent and the functions that it provides. Using the example of Thailand's mangroves in attenuating wave damage from more commonly experienced storm events, spatial heterogeneity arises because proximity (of mangroves) to shorelines is a critical determinant of the degree to which this function is provided: that is, it diminishes the further the ecosystem is (inland) from the shore. Taking explicit account of this heterogeneity is needed as a more defensible basis for aggregation. This is also required for more accurate policy analysis. Put another way, what Barbier et al. show is that the (estimated) marginal value of mangrove area in their study area in Thailand is declining. The total net benefits of protection of this ecosystem are at their maximum at around 8 km^2. Given that current mangrove area was 10 km^2, this means that while mangrove protection is frequently justified, some conversion might also be economically desirable.

While economics can contribute greatly to guiding the valuation of ecosystem services, it can also shape thinking about the implementation of policies aimed at delivering such values. Unfortunately, at present, many of the policies employed to deliver ecosystem services fail to heed either evidence regarding the way in which values can vary over different patches of ecosystems or the lessons of basic economic theory regarding incentives that actors possess to reveal truthfully their valuation of services that they enjoy. An example is provided by the UK Entry Level Stewardship (ELS) scheme (Natural England, 2010) which offers a flat rate payment to all farmers irrespective of their location. Such schemes fail to target payments to those areas which yield the highest values and provide no incentive for farmers to provide anything other than the basic level of land management consistent with the scheme. Similar approaches characterise much of the increasingly substantial payments made under Pillar Two of the EU Common Agricultural Policy.

Thus economic valuation of itself is insufficient to improve the efficient delivery of ecosystem services. A simple example illustrates the problem and how economic intuition can help. Suppose that policy makers seek to reduce diffuse water pollution from farms through a payment for ecosystem services (PES) scheme. A first requirement is to undertake a valuation exercise identifying those river catchments (and areas within those catchments) where reductions of pollution are likely to generate the largest net benefits. This might identify, for example, farms in locations above the inlet to water supply reservoirs as those most important to target. Now the focus ought to switch to the efficient implementation of such policies.

One rather naïve approach might be to simply ask farmers to state the levels of compensation they require to move towards modes of production which avoid diffuse pollution. Of course, farmers have an incentive to strategically overstate their compensation requirements. However, the economic theory of auctions suggests that even relatively simple approaches can significantly improve implementation efficiency (Vickrey, 1961; Clarke, 1971; Groves, 1973; Groves and Ledyard, 1977). For example, switching to a simple sealed bid contracting system might reduce the potential for strategic responses and improves incentive compatibility. This could be the case if farmers are told that contracts will be awarded according to the combination of pollution reduction and cost.

In certain circumstances even greater efficiency gains can be obtained. For example, where the delivery of ecosystem services can be readily measured (for example in policies seeking the protection of certain habitats) then land owners will be those best able to judge whether their land is particularly suitable for providing such goods (or faces the lowest opportunity costs). Such actors can outbid competitors by offering better outputs (or lower costs) than their rivals.[9] To date, practical examples of such agreements are generally confined to the experimental laboratory.

One further point is that valuing ecosystems and biodiversity valuation are complex endeavours and often at the frontier of valuation knowledge. This suggests good reason, in certain contexts, to be circumspect about the role that valuation might play in informing decisions about conservation. Decision-making in such situations where values are unknown – or where values cannot be established to any degree of validity – has generated much debate. In such cases, however, "caution" (given what might be lost) might be a sensible watchword. Possible responses include the adoption of ecological standards sometimes termed "safe minimum standards" to ensure the sustainability of resources which are not amenable to valuation (Farmer and Randall, 1998) or offsetting or compensatory projects validated for their ecological suitability (Federal Register, 1995). In such cases, valuation of benefits is downplayed perhaps for a greater emphasis on cost-effectiveness in meeting specified physical targets (see Chapter 12).

An illustration of this challenge in determining how exactly valuation could guide social decision-making is provided by the example of valuing biodiversity. Weitzman (1993) – using the example of the world's remaining species of cranes – defines biological importance of each species in terms of their taxonomic distinctiveness (e.g. of the whooping crane compared with other crane species)[10] and the likelihood of extinction (of a given species). Assuming that maximising (expected) diversity is the objective, species conservation becomes a problem of cost-effectively distributing the marginal (available) unit of money from conservation funds to where it achieves the highest pay-off. Typically, this will be where there is some combination of high diversity and low survival probabilities.

Ideally, it would be useful to extend such insights with reference to the preferences that people might have for diversity. Somewhat reassuringly, Morse-Jones et al. (2012), for example, find that stated preference responses reveal expected substitution patterns across ecologically similar species, e.g. different small amphibians. However, preferences need not always conform to what is ecologically feasible or sustainable. Thus, in the Morse-Jones et al. study, respondents had a massively stronger preference for iconic, "charismatic" animals which outweighs concerns regarding ecologically crucial issues such as extinction threat. So, for example, willingness to pay to conserve lions, even where these animals are not threatened by extinction, hugely outweighs stated values for say a species of frog, even when it is on the brink of extinction.

Another example is provided by Bateman et al. (2009). That study observes that while respondents had strongly positive preferences for enlarging an area of freshwater marshland suitable for visiting and viewing bird populations, they had negative values for an adjoining area of tidal mudflats, even though these were a major source of food attracting those birds to the area. In many respects, these findings are not surprising. However, what it does raise is a deeper question about whether the extent to which economic values can be a guide for decision-making or whether ecological constraints need to be considered. Clearly, the claim that human preferences are (almost always) "right" or "wrong" is overly simplistic at either extreme. However, where to draw the line is far from obvious and – given changing knowledge – is anyhow likely to be a shifting target. Nevertheless, while recognising the importance of economic values for thinking about the importance of ecosystems and guiding policy thinking, one needs to be mindful of the complexities and uncertainties involved.

13.5. Concluding remarks

The valuation of ecosystem services has become a crucial element (perhaps *the* crucial element) in quantifying the contribution of ecosystems and biodiversity to human well-being. A significant body of research has already begun to emerge and a number of recent national and international ecosystem assessments have helped provide further impetus to such efforts. Needless to say, significant challenges remain. Hence, while the evidence-base is broad and – at least for some ecosystem services – deep, reflections on this progress indicate a need for greater understanding of ecological production, especially as it relates to spatial variability and complexities in the way that services are produced. The size and significance of inevitable gaps in the empirical record as well as the ability to fill these gaps by judiciously transferring values; and, the scope and limits in using this evidence-base to inform practical decision-making both generally and, in relation, to concerns about whether the valuations that one can find in this literature genuinely tell much about the importance of ecosystem assets and biodiversity.

In this current chapter, the focus has been on valuation methods and particularly the challenges inherent in seeking to value non-market costs and benefits. Some of these challenges involve general considerations although other issues are specific to valuing ecosystems or at least seem particularly acute in that context. In some cases, for example, there might be good knowledge about the value of a particular (ecosystem) service endpoint. So, while valuing the benefits to physical and mental health (of proximity to greenspaces) might be feasible, establishing the causal link between experiences of nature and these health outcomes is the greater relative challenge. For other types of cultural value, particularly those related to "non-use", the suspicion is that these might be substantive in some contexts. Less is known in this instance to confirm systematically these suspicions. A natural response in the past might be to look to stated preference methods to provide this evidence. But there is increasing recognition that this method may not be well suited to eliciting values where those people asked to do the valuing lack familiarity with or experience of the (ecosystem) good. Ways of resolving this contradiction are in their infancy.

Such challenges need to be viewed in context. A growing number of large-scale ecosystem assessments has shown how the empirical record can be put to use in an informative and policy-relevant way. Such developments could be crucial in translating valuations into meaningful policy analysis. It may also offer some hope for shedding light on the value of what is lost when and if ecosystems and biodiversity are degraded and destroyed in more highly aggregated assessments. While such questions are commonplace elsewhere, in the ecosystem context these have only begun to be asked although related issues of valuing ecosystem complexity have a longer standing (see also the Annex to this Chapter). Progress on these matters, both in theory and practice, is surely only a matter of time. Nevertheless, it seems unavoidable that uncertainties will remain. That is, while one can conclude positively on the rapidly evolving scope for ecosystem and biodiversity valuation to contribute to a profound understanding of suitable policy responses, there remains room for debate about whether valuation is in itself enough to ensure effective policies. There is considerable debate remaining also about how to conduct decision analyses in those contexts where valuation and understanding of the natural world is likely to remain relatively uncertain.

Notes

1. Of course, such a comment does not apply only to this ecosystem context. A great many applications necessarily require interdisciplinary collaboration between, at a minimum, the natural sciences and economics (arguably extending to a much wider fusion of disciplines).
2. See, most for a summary, Carson's (2011) bibliography of published and unpublished CV studies from around the world.
3. A number of studies combine RP and SP approaches in order to enhance the respective strengths of these data and minimising limitations (see, for example, Adamowicz et al., 1994).
4. An existence value can be derived from the simple knowledge of the existence of the good or the service. In the context of the environment, individuals may place a value on the mere existence of species, natural environments and other ecosystem. If an individual derives well-being from the knowledge that other people are benefiting from a particular environmental good or service, this can be termed altruistic value. Such values accrue during an individual's lifetime, but vicarious valuation can also occur inter-generationally. The effect on well-being of knowing that one's offspring, or other future generations, may enjoy an environmental good or service into the future, such as a biodiversity-rich forest being conserved, is termed bequest value.
5. Arrow et al. (2000) have made an analogous point in the context of the physical processes that the value of some system as a whole may be more than the value of the sum of its parts perhaps because of complex ecological interactions.
6. In much in the same way, that is, as a move across locations, and consequent environments, will alter the value of any given resource: e.g. water in the desert has a much higher marginal value than in areas of high rainfall.
7. The UK NEA involved a team of over 160 natural scientists assembled to quantify the status of ecosystem processes and the final ecosystem services they generate across the UK, looking at individual habitats classifications (e.g. wetlands and woodlands) as well as ecosystems services across these classifications. In addition, an economics team complemented this work and its structure with the added emphasis on the value of habitats and ecosystem services under investigation.
8. This in turn builds on Bateman et al., 2003.
9. Such markets can also be designed to benefit private sector purchasers of ecosystem services. For example in countries where this institutional regime occurs, private water companies may be able to reduce their costs of providing potable water by avoiding costly treatment options by engaging

with land owners to reduce pollution inputs to rivers. Indeed, economic theory identifies the potential for multiple private sector bodies to combine to purchase such services provided that markets are created so as to avoid free-riding by ensuring that PES trades only go ahead if all parties contribute to their purchase (Guth et al., 2007; Potters et al., 2007; Ekel and Grossman, 2007; Bracht et al., 2008).

10. Genetic distinctiveness is defined, by Weitzman (1993), as the evolutionary distance each existing species is from a common ancestor species.

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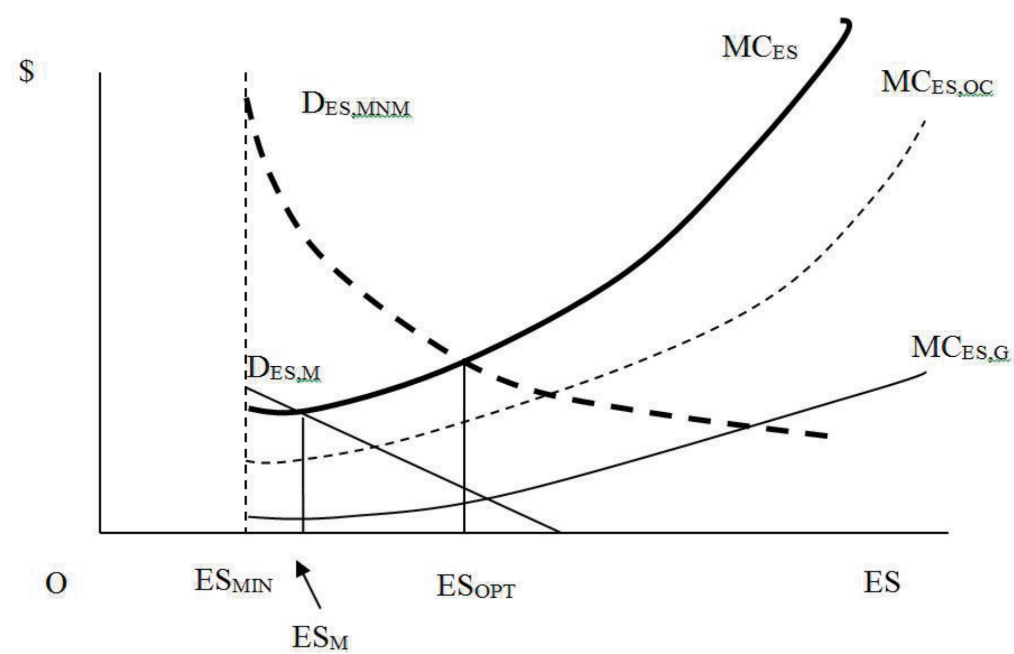
ANNEX 13.A1

Marginal vs total valuation

Whether it is sensible to speak of the "total" value of a type of ecosystem and even more ambitiously the total value of *all* ecosystems has been the subject of considerable debate following a handful of studies that claims to do just this (e.g. Costanza et al., 1997; Sutton and Costanza, 2002). On the one hand, as Costanza et al. (2017) point, these studies have been highly effective both terms of raising the profile of the ecosystem service approach (judging from academic citations and beyond to public forums) and making the broad that ecosystems command considerable economic value. On the other, such highly aggregated studies are beset with challenges. To see some of the issues, consider Figure 13.A1. On the vertical axis is economic value in dollars. On the horizontal axis is a measure of the flow of ecosystem services (ES) which is assumed can be conflated into a single measure for purposes of exposition.

Figure 13.A1. **Stylised costs and benefits of ecosystem service provision**



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 are associated with already established markets in which formal exchange takes place using the medium of money. Thus, if there is 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, this is a risky interpretation (see below). For the moment it is best to think of ES in Figure 13.A1 as covering a single ecosystem, say tropical forests.

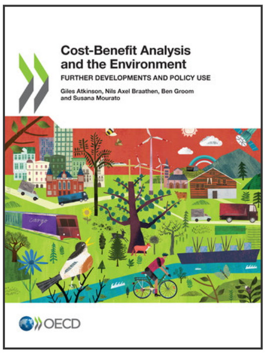
The second construct is another demand curve but this time for all services from the given ecosystem, regardless of whether they currently have markets or not. This is $D_{ES,MNM}$ which is the demand curve for marketed (M) and non-marketed (NM) ecosystem services. As noted above, there are various non-market services such as watershed protection, carbon sequestration and storage, scientific knowledge, the aesthetics of natural ecosystems, and so on. It is known already that $D_{ES,MNM}$ lies above $D_{ES,M}$ everywhere. 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 (Vitousek et al., 1987).

The two demand curves shown in Figure 13.A1 are downward sloping, as one would expect. The more ES there are the less humans are likely to value an *additional unit* of ES. There is 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 there is a very low level of ES. Imagine a world with very few forests, very little unpolluted oceans, a much reduced stock of coral reefs, an atmosphere with a very much higher concentration of carbon dioxide and other greenhouse gases. In the limit, if there were no unpolluted oceans, no forests, extremely high concentrations of greenhouse gases, then the willingness to pay for one more unit of ES would be extremely high, perhaps on the way to infinity. For this reason, $D_{ES,MNM}$ bends sharply upwards as points closer to the origin on the horizontal axis are approached. Essentially, $D_{ES,MNM}$ is unbounded: there is some irreducible minimum ES below which marginal WTP would rise dramatically such that there is no meaning to the notion of economic value in this unbounded area.

Left alone, ecosystems might continue to provide the same ES year after year. But in order to maintain ES of value to humans it will be the case that certain costs are incurred. Figure 13.A1 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, $MC_{ES,G}$ is shown as a gently rising line. The second category of costs is of considerable importance and comprises the opportunity costs of providing ES. The assumption is that ES are best secured by 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. This is referred to in the figure 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". The sum of $MC_{ES,G}$ and $MC_{ES,OC}$ = MC_{ES} gives the overall marginal cost of conservation.

Figure 13.A1 is simplistic but it 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 attention is confined arbitrarily 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 $D_{ES,M}$. Figure 13.A1 shows the real possibility that failure to reflect true WTP in actual markets results in a serious under-provision of ES.

Figure 13.A1 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 unbounded. This explains, perhaps, why one economist referred to Costanza et al.'s (1997) 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 (1998) and Bockstael et al. (2000). Accounting for the value of (actual) changes in ecosystems may be a better focus as in Costanza et al. (2014). However, Chapter 12 noted that the empirical challenge of measuring such changes remains huge especially in the aggregate.



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