COST-BENEFIT ANALYSIS AND THE ENVIRONMENT: HOW TO BEST COVER IMPACTS ON BIODIVERSITY AND ECOSYSTEM SERVICES

1. Introduction

1. Valuation studies on the benefits of biodiversity and on the services provided by different ecosystems (ESS) have been growing at an exponential rate. While in 1990 there were only a handful of papers on these topics; in 2012, around 2 500 papers were published (Markandya and Pascual, 2014). There are now estimated ranges for the services from most habitats, by type of ecosystem service, usually expressed in USD per hectare per year. This impressive effort, however, is matched by a much smaller literature that values changes in service flows as a result either of policy inaction, or of a policy or measure designed to modify the habitat that provides the services. To be sure, a number of studies have valued such changes, and some of them have then compared the changes in benefits resulting from a given action to its costs – i.e. carried out a cost-benefit assessment. Coverage varies by habitat and by region, as does the quality of the assessment, but it would be fair to say that it is possible now to include the changes in a number of ecosystem services or of biodiversity resulting from public sector programmes at different levels of aggregation.

2. This paper reviews the literature on the valuation of such services and their use in cost-benefit analysis, under the subheadings of biodiversity and ecosystems. It begins by laying out the distinction between these two sources of environmental benefits and the links there are between them. A survey of the main values of ecosystem services (ESS) in money terms is presented, which reveals a wide range, depending on location, method of estimation, etc. This suggests that there are major limitations on the use of existing estimates from the literature to value services in a specific cost-benefit context. The literature on "benefit transfer" as such a procedure is called, notes that while it is possible to make such transfers, they should be done with care and they introduce additional errors in the measurement of the benefits or costs. All these issues are discussed in Section 2.

3. Section 3 reviews the use of ecosystem and biodiversity values in making cost-benefit assessments at different geographical levels. The first is the global or national level, where the figures are used to inform policies regarding protected area targets, targets for reforestation, etc. The second is for interventions involving specific habitats or types of biodiversity. The ones examined are agricultural systems, coastal restoration, forests, river basins and marine areas. In each case, selected studies are reviewed to illustrate the approaches adopted and the problems associated with them. This is done in the context of the study and the purpose for which it was undertaken, which is relevant to its evaluation.

4. Section 4 looks at how successful such studies have been in influencing policies that have a potential impact on ESS and on the trade-offs between such services and other social and economic goals. Previous surveys that have looked at this question are reviewed (Adamowicz, 2004; Silva and Pagiola, 2003; Atkinson et al., 2012) and along with the results from this evaluation, some overall conclusions on the state of the subject are offered. Finally, areas where further work is needed are identified.

2. Estimates of biodiversity and ecosystem services

2.1 Biodiversity and ecosystem services

5. The term biodiversity is defined by the Convention on Biological Diversity as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems". The same reference source defines an ecosystem as "a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit"[1] and ecosystem services as the benefits people obtain from ecosystems. The focus of much of the literature has been on the nature of these services and their value. Before doing so, however, it is important to consider both concepts and the link between them.

6. The topic of biodiversity loss has been the subject of a vast and growing scientific and economic literature. Species are estimated to be going extinct at rates 100 to 1000 times faster than in geological times (Pimm *et al.*, 1995; Chivian and Bernstein 2008). Moreover the rate of extinction is accelerating as habitats get smaller and smaller (Pimm and Raven, 2000). Globally, terrestrial biodiversity (measured as mean species abundance – or MSA – an indicator of the intactness of a natural ecosystem) is projected to decrease by a further 10% by 2050 (OECD, 2012a).

7. There is evidence that many of these extinctions are associated with economic and social losses. For example, between 1981 and 2006, 47% of cancer drugs and 34% of all "small molecule new chemical entities" (NCE) for all disease categories were natural products or derived directly from them (Newman and Cragg, 2007). In some countries in Asia and Africa, 80% of the population relies on traditional medicine (including herbal medicine) for primary health care.[2] As extinctions continue, the availability of some of these medicines is likely to be reduced and new drug developments could be curtailed (Nunes and van Den Berg, 2001).

8. Yet, while there are a number of pieces of evidence of this nature available, and there are numerous studies that look at the value of biodiversity in specific geographical contexts, it has proven challenging to link loss of biodiversity to the very wide range of benefits that humans derive from natural systems within which such biodiversity is embedded.[3] This is because the links between biodiversity and biological systems and the economic and social values that they support are extremely complex. Even the measurement of biodiversity is challenging, with a multi-dimensional metric being regarded as appropriate (Purvis and Hector, 2000; Mace et al., 2003) but with further work being considered necessary to define the appropriate combination.[4]

9. For this reason the operational focus, initiated by the Millennium Ecosystem Assessment (MEA, 2005) has shifted to measuring *ecosystem services (ESS)*, which are derived from the complex biophysical systems. The MEA defines ecosystem services under four headings: provisioning, regulating, cultural and supporting and under each there are a number of sub-categories. Table 1 provides an update of the original classification prepared by UN System of Environmental-Economic Accounting and the EEA.[5] In this reclassification, regulating and supporting are merged into 'regulation and maintenance' and for the new

1 . *www.cbd.int/convention/articles/default.shtml?a=cbd-02*.

2 . "Traditional Medicine". World Health Organization web site.

3 . For a brief review see, ten Brink (ed.) 2011, Chapter 5.4.

4 . Some researchers take the view that the community has moved too much in the direction of ESS, making the approach to nature too instrumental. It is after all not the only framework for describing the man/nature relationship, although it is now the one with the greatest quantification and with the most attempts to use it in a cost-benefit context, as well as in the construction of a natural capital accounting system.

5 . Common International Classification of Ecosystem Services. Revised version, (CICES v4.3), 17 January 2013, *http://cices.eu/*.

three categories, each heading is broken down into division, group and class. The reclassification is a key part of the attempt to construct capital accounts at the national level, where estimates of natural capital, based on the ESS it provides is a central component.

Table 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 & control of erosion rates
			Buffering & attenuation of mass flows
		Liquid flows	Hydrological cycle & water flow maintenance
			Flood protection
		Air Flows	Storm protection , ventilation and transpiration
	Maintenance of physical, chemical, biological conditions	Habitat and gene pool protection	Pollination & seed dispersal
			Maintaining nursery populations & habitats
		Pest & disease control	Pest control Disease control
		Soil formation & Composition	Weathering processes Decomposition and fixing processes
		Water conditions	Chemical condition of fresh & salt waters
		Atmosphere & Climate regulation	Global climate regulation by reducing GHGs
			Micro & region climate regulation
Cultural	Physical & intellectual interactions with biota/ ecosystems	Physical & experiential	Experiential use of plants, animals landscapes
			Physical use of land/ seascapes in different ways
		Intellectual & 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

10. A link between the concepts of biodiversity and ESS is made in a French government review of the topic, in which the authors distinguish between "remarkable" biodiversity and "ordinary" or "general" biodiversity (Chevassus-au-Louis et al., 2009). The former corresponds to entities (genes, species, habitats, landscapes) that society has identified as having an intrinsic value, based mainly on values other than economic. Such biodiversity can nevertheless be measured in money terms, but such values are only to be

used in a subsidiary manner in the discussions about the conservation of those entities. For the other category of ordinary biodiversity, however, the workgroup opted not to try to evaluate it directly, but to do it on the basis of the ecosystem services that benefit society. The report argues for an underlying hypothesis concerning a relationship hypothesis based on a proportionality between the fluctuations in biodiversity and the extent of these services. Using national data, the study derives reference values for France for categories of ecosystems taking account of normal biodiversity.[6] It makes the further point that there is a danger, in the present methods of elicitation, of both undervaluing and overvaluing the contribution of biodiversity in terms of the benefits it provides. Partly this can arise from double-counting the services provided (see the discussion later around Table 3) and partly from the difficulty to link the precise role of biodiversity in the functioning of ecosystems and the services they provide. These are real issues that need to be addressed in the cost-benefit assessment of public investments where biodiversity and ecosystems services are impacted.

11. The services listed in Table 1 are provided by a range of different ecosystems within which different habitats can be found. An ecosystem where several habitats are present is referred to as a biome. The literature contains ten broad categories (listed in Table 2), for which values of several of the services described in Table 1 have been estimated. It should be noted that not all studies work with these categories of biomes and some services (e.g. pollination) cut across the biome classification.

12. Before proceeding to look at the values of services provided by the ecosystems or biomes, two general points are worth noting. First one finds, as with biodiversity, that the planet has experienced major losses in the services derived from these ecosystems. During the last century for example, the planet has lost 50% of its wetlands, 40% of its forests and 35% of its mangroves. Around 60% of global ecosystem services have been degraded in just 50 years (ten Brink, 2011).[7]

Table 2. Major biomes used in the ecosystem valuation literature

Biome (marine/aquatic)	Biome (terrestrial)
Marine (Open Oceans)	Freshwater (Rivers/Lakes)
Coral Reefs	Tropical Forests
Coastal Systems	Temperate Forests
Coastal Wetlands	Woodlands
Inland Wetlands	Grasslands

Note: Coastal systems include estuaries, continental shelf areas and sea grasses but not wetlands such as tidal marshes, mangroves and salt water wetlands.
Source: De Groot et al. (2012).

13. Second, while working at the ecosystem level makes things somewhat easier it is important to understand the causes of the loss of these services and the links between biodiversity and ecosystem services. Indeed this is a major field of research for ecologists and one thesis developed over a long period is that more diverse ecosystems are more stable and less subject to malfunction and thus the services they provide are more stable over time (Haines-Young and Potschin, 2010; McCann, 2000; Tilman and Downing, 1994). Evidence in support of the thesis has been provided from a range of natural and managed ecosystems, but the evidence also points to more complex relationships, in particular to the fact that the functions of ecosystems are determined more by the functional characteristics of the component organisms rather than the number of species (Grime, 1997). Overall, most ecologists would agree with the statement that "diversity can be expected, on average, to give rise to ecosystem stability" (McCann, 2000, p.232).

6 . In France estimates of the impacts of various fiscal incentives on biodiversity have been analysed using mainly indicators of biodiversity and making little use of monetary estimates of losses of biodiversity as such. This has fed into the debate on which subsidies should be reduced or eliminated (Sainteny, 2011).

7 . The categories in Table 2 may appear not to cover urban landscapes but in fact such areas are included under terrestrial biomes as well as (in some cases) coastal systems. Also it is important to note that urban zones can be significantly affected by changes in biomes outside the physical boundaries, such as freshwater and grassland biomes.

14. In short, the current state of knowledge on the links between biodiversity and ecosystem services is still a topic of research and while some clear lines are emerging, they are not strong enough to allow formal modelling to be carried out at a level that would produce credible estimates of the total economic value of biodiversity.[8] The latter therefore remains a topic for research.[9] At the same time, some efforts have been made to recognise that at least a part of the difference in the productivity of an ecosystem may depend on how much biodiversity it contains and, furthermore, this link between productivity and ecosystem diversity can be captured quantitatively using the concept of mean species abundance (MSA). The use of MSA in deriving the stock value of a biome is discussed below. Lastly, the quality of biodiversity is not relevant in all ecosystems, notably ones where cultural values are at stake. Here one can proceed to conduct the valuation of changes independently of biodiversity considerations.

2.2 Average values for ecosystem services

15. A large number of studies have been undertaken to value the flow of services listed in Table 1, in the context of the biomes listed in Table 2.[10] Much of this work has been summarised in The Economics of Ecosystems and Biodiversity (TEEB) study (TEEB Synthesis report, 2010), which was launched by the G8+5 Ministers of the Environment in 2007 to draw attention to the global economic benefits of biodiversity and the costs of biodiversity and ecosystem loss. A more comprehensive set of background papers and sectoral and country studies have been undertaken since (Russi et al., 2013; McVittie and Hussain, 2013; TEEB, 2013).[11] A good reference to the economic valuation of ecosystems that was undertaken as part of the TEEB exercise is ten Brink (2011). One study undertaken as part of TEEB that is particularly relevant to this exercise is discussed further below (Hussain et al., 2011, 2013).

16. A recent survey that collects and summarises the findings of many of these studies is De Groot et al., 2012.[12] They identified more than 1,600 studies over the period 1960 to 2008 and extracted 655 data points that could be used to calculate the flow of services in terms of international dollars per hectare per year.[13] Studies in different currencies were converted into US dollars using purchasing power parity (PPP) exchange rates and account was taken of inflation between the year of study and the standardised year, which was 2007.

17. Table 3 provides the main results that emerge from this literature review. The results show a significant value from the different biomes, ranging from a high of USD 352 249 per ha per year for coral reefs to a low of USD 491 per ha per year for marine areas. In terms of services, the main categories used

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8 . The term total economic value is used here to reflect the valuation measured at the margin, multiplied by the quantities. In one sense of course the total value, in the sense of value as a whole, is not measureable, as with zero biodiversity there would be no life.

9 . Theoretical models of the economic values attached to biodiversity have been developed. See for example, Brock and Xepapadeas, 2003. Such models draw simple links between harvesting rates, system biodiversity and overall system value. As yet, however, they are not supported by empirical estimates that can be applied to derive these system values.

10 . Studies are available from a number of databases. Sources of monetary values for ecological entities include the Environmental Valuation Reference Inventory (*www.evri.ca/Global/Splash.aspx*). Other databases include COPI (Braat and ten Brink, 2008), EVRI (1997), ENValue (2004), EcoValue (Wilson et al., 2004), Consvalmap (Conservation International, 2006), CaseBase (FSD, 2007), ValueBaseSwe (Sundberg and Söderqvist, 2004), ESD-ARIES (UVM, 2008) and FEEM (Ojea et al., 2009) (see *www.es-partnership.org* for access to most of these databases).

11 . The full list of studies is available from *www.teebweb.org/our-publications/all-publications/*.

12 . A related study is Costanza, 2014. It draws on the same literature but takes it a stage further to attempt a global value of the services of these ecosystems.

13 . International or Geary-Khamis dollar is a unit of currency constructed to standardise money values by correcting money values across countries to the same purchasing power that the US dollar has at a point in time. This involved using purchasing power parity exchange rates as has been done in the Table cited.

in the literature are regulating, followed by cultural, provisioning and habitat. Within the regulating services, ecosystems provide an important source of waste treatment and erosion prevention. There is, however, a concern that one cannot add the service categories to obtain a total economic value. As Chevassus et al., 2009 and others have noted, some of the service categories are the source of the other values and to include both would amount to double-counting. While this is true to some extent, it is also the case that in Table 3 the regulating/service functions are combined and many of these are not captured in the provisioning or cultural categories. Climate regulation is clearly a case in point – it has benefits but the standard valuation of food, water and raw material provision does not include such benefits. By contrast, the benefits of water flow regulation and waste treatment are fully captured in the food and water provision estimates. Thus, while there is an element of double-counting that needs to be avoided (e.g. pollination benefits are picked up in food provision through agro-ecosystems represented here by grasslands), not all categories of regulating benefits constitute double-counting. Care is needed when assembling total values and it should be noted that the total figures in Table 3 contain some double-counting.

18. The methods used to elicit the estimates presented in Table 3 cover the whole range of valuation techniques used in environmental economics. Perhaps more than most environmental valuation studies, the main method used in this set is direct market valuations, notably direct market pricing. Table 4 summarises the share of different techniques in the set of valuation studies. Direct market valuation methods include market pricing, market based payments for environmental services and factor income/production function methods. They are most deployed for provisioning service valuation but are also frequently used for habitat services and cultural services. Cost-based methods include: avoided cost, restoration cost, and replacement cost. They are most used for the valuation of regulating services. The revealed preference methods consist of hedonic pricing and travel cost methods, and are used exclusively for valuing cultural services. Finally, stated preference methods consist of contingent valuation, conjoint choice and group valuation and are used for valuing habitat and cultural services and are also frequently used in the literature. For more details regarding different valuation methods, see Pearce et al., 2006.

Table 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 & 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**	**25847**	**171 515**	**17364**	**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: Adapted from De Groot et al., 2012.

Table 4. Methods used to value ecosystem services
Per cent

Ecosystem Services	Direct Market Values	Cost Based Methods	Revealed Preference	Stated Preference
Provisioning	84%	8%	0%	3%
Regulating	18%	66%	0%	5%
Habitat	32%	6%	0%	47%
Cultural	39%	0%	19%	36%

Note: Percentages sum across the rows.
Source: Adapted from De Groot et al., 2012.

Table 5. Range of values in studies of ecosystem services
International dollars per hectare per year, 2007 Price level

Ecosystem	Mean	Median	Minimum/Mean (%)	Maximum/Mean (%)
Marine	491	135	17%	339%
Coral Reefs	352,915	197,900	10%	603%
Coastal Systems	28,917	26,760	90%	145%
Coastal Wetlands	193,845	12,163	0.2%	458%
Inland Wetlands	25,682	16,534	12%	409%
Rivers and Lakes	4,267	3,938	34%	182%
Tropical Forests	5,264	2,355	30%	396%
Temperate Forests	3,013	1,127	9%	545%
Woodlands	1,588	1,522	86%	138%
Grasslands	2,871	2,698	4%	207%

Source: Adapted from De Groot et al., 2012.

19. The categories given in Table 3 cover a wide range of services with different methods of elicitation of values. Some people might question whether the services valued using stated preferences or indirect valuation methods of revealed preferences are as "real" (i.e. do they represent some coherent underlying preferences) as those obtained using market methods. While this cannot be established absolutely, the predominant evidence is that non-market methods for valuation, when used with care and following the best techniques available, do provide credible numbers that can be compared to those obtained from market methods.[14] Indeed, where both methods have been used to value the same services, the numbers generally turn out to be comparable.[15]

20. While the work summarised in Tables 3 and 4 is impressive, there are a number of aspects that need further consideration. First, it is not really possible to take the average values given in the table and use them as single figures that apply to all services provided by a given biome. The average values per hectare cannot be applied to other areas or regions without some consideration of local factors, such as population density (an ESS may be more valuable when more people are living nearby), the degree of development of the region, etc., because these factors can lead to major differences in values. One can see this by looking at the extent of variation around the mean in the studies summarised in Table 3. Table 5 provides this information and shows that the mean and median values are very different and there are many studies with values much lower than the mean (in some cases by an order of magnitude) and a number with values well above the mean. Thus, local factors need to be taken into account when incorporating ecosystem values into cost-benefit analysis. Such differences can be accounted for at various levels through a procedure referred to as benefit transfer. The simplest would be to take an individual estimate from a similar area in the country or from a similar country. A more sophisticated method would involve carrying out a meta-analysis in which the individual values are used to estimate a relationship, giving the value per hectare as a function to environmental and socioeconomic variables. Such an approach has been widely evaluated (Lindhjem and Navrud, 2008; Brander et al., 2008) and guidelines for the practice are available (Johnston et al., 2015.). While there can be significant errors of

14 . There is also an issue with all methods of determining the boundaries of the area of study. The wider the potential area over which people are affected the greater will be the value but these boundaries are not easy to determine and add an element of arbitrariness to the estimates.

15 . Unfortunately, market methods cannot be used for some services (e.g. non-use aspects of cultural services), where such comparisons cannot be made and where other methods of verification (e.g. contributions to charities or other voluntary payments) have been made (Pearce et al., 2006).

transfer, the method does allow one to get figures that are broadly of the right order of magnitude. Meta-analysis is discussed further in Section 2.3.

21. Second, the coverage of ecosystem services is far from complete. Although Table 3 provides numbers under most categories, the items included do not pick up all the linkages between the service and the state of the biome. For example, the role of oceans in climate regulation is still being investigated and the studies from which current values have been derived are based only on a partial understanding of the underlying physical phenomena. The same applies to the value of genetic resources and genetic diversity in different biomes and to a number of other categories of services.

22. Third, for some categories of services, the studies from which the average values have been derived are disproportionately from developed countries. While this is not true for provisioning services or for services from biomes such as ocean systems, coral reefs and coastal systems, it is true for recreational and other cultural services. The consequence is that for these categories of services the transferability of the numbers to developing countries may be problematic (see below).

2.3 Taking local factors into account

23. Locally applicable values can be derived from studies through the use of a procedure called benefit transfer, in which meta-analysis is a key method. This consists of estimating unit values of ecosystem services for a given site as a function of site characteristics as well as the socio-economic and geographical characteristics of the region or country and the method of estimation used. The estimation uses data from individual studies to construct the meta-level database from which a statistical estimation of the function is carried out.

24. Examples of meta-analytical functions that have been estimated in the literature include inland European wetlands (Brander et al., 2011); grasslands, global wetlands, mangroves and coral reefs (Hussain et al., 2011); recreational and passive forests services (Chiabai et al., 2009). In all cases but one (coral reefs), one of the explanatory variables is the level of per capita income, measured in PPP terms, whereby the value of a given class of ecosystem services rises with per capita income. The elasticity of unit ecosystem service values with respect to GDP varies considerably: from around 0.4 for European wetlands, to 0.6 for wetlands globally, to 0.7 for recreational services of forests, 0.9 for grasslands, 1.5 for mangroves and 3.5 for non-use or passive values of forests.[16] Other variables that emerge as significant include: the presence of similar ecosystems nearby, population concentrations within a local range of the biome (e.g. 50 km), concentration of economic activity in the same range, and method of estimation used. While it is not a problem that estimated values vary with such factors, it is important that thee estimation methods take them into account.

25. Where meta-analytical functions are available they can be used to provide some guidance values for services for other local areas, although a significant level of uncertainty remains with the estimates. Indeed some research shows that the extent of the error in making a transfer of values using sophisticated meta-analysis can be quite large and the method is not always more accurate than a simpler transfer based on adjusting for differences in per capita GDP (Lindhjem and Navrud, 2006). Nevertheless it is generally more reliable to use estimated functions used on as wide a data set as possible.

26. As far as incorporating these values into cost-benefit assessments is concerned, the margins of error should not prevent them from being included (there are after all similar errors in other aspects of socio-economic modelling). Ideally some spatial differentiation is desirable within a country, but even a single value for some of the service could provide a useful complement to the economic components of the appraisal. The more difficult question is how to link the changes in the values of services to *changes* in economic activity. Most of the studies provide an estimate of the service currently provided and the metric used is frequently either the total value of a system or the value per hectare. Policy actions that are of interest in a cost-benefit assessment will often change the functioning of the system. If they reduce the size, for example, of a wetland or a forest one could use the value per hectare and apply it to the area lost. The assumption in doing that is that

16 . In valuing mortality risks in different countries, the OECD has recommended an income elasticity of 0.8 (OECD, 2012b) but that applies for that specific public good and cannot be transferred to other such goods.

the marginal and average values are equal: it is not known if this is the case and there is little evidence to guide us.

27. Another difficulty arises when the policy or measure does not consist of a loss of an area but a change in its quality. Forests, woodlands, grasslands or water bodies many not disappear but may get degraded when roads are built or overexploited for pasture. The valuation then has to focus on how to value changes in quality, for which the literature is much thinner, although there are studies looking at specific impacts and these are reviewed in the following section. Some standardised values of ecosystem services at the local level, taking account of quality or biodiversity changes is the way forward and some work on these lines has been undertaken in France, especially in relation to infrastructure projects (Tardieu et al., 2015).

2.4 Incorporating biodiversity into the ecosystem valuation framework

28. As noted an attempt has been made to incorporate biodiversity changes within the ecosystem approach. To do this biodiversity is measured as "the remaining mean species abundance (MSA) of original species, relative to their abundance in pristine or primary vegetation, which are assumed to be not disturbed by human activities for a prolonged period" (Alkemade et al., 2009, p. 375). The peculiarity of MSA is related to the fact that it is not built on actual observations in the study area, but on the relations between pressures or drivers and impacts on species abundance. For each pressure under consideration, a meta-analysis is first carried out to put in relation the MSA quantities with a number of drivers. The MSA quantities used in the meta-analysis are constructed from indicators taken from the literature, and specifically the abundance of different species (number of individuals per species, density or cover) registered in primary vegetation areas (natural or relatively untouched) and the abundance of species in disturbed environments. The MSA indicator as dependent variable in the meta-analysis is constructed by dividing the species numbers by the area.

29. MSA quantities are calculated for each of the above mentioned drivers taking into account the cause-effect relationships for each driver as estimated in the meta-analysis. As ecosystem approaches use the area of land as the basis of calculating the value of services obtained the MSA component of a geographical region is taken into account by multiplying the area by an MSA quantity normalised on a scale of zero to one with different normalization for different ecosystem services. So for example if an area is pristine and there has been no loss of biodiversity its MSA value will be one, while if it managed in some way the value will range from 0.5 to 0.7 depending on which service one is considering. An area that has become totally artificial and depleted of all species will have a value of zero (Braat and ten Brink, 2008: Chapter 5). Such "MSA adjusted areas" have been estimated for different biomes across the world and over time, going back to 1900 and even earlier by the biodiversity modelling work undertaken by the GLOBIO3 team in the Netherlands (Alkemade et al., 2009). Clearly there is the implicit assumption here that one can make a linear trade-off between the measure of biodiversity in terms of MSA and land area. The authors provide some justification for this but also recognise that there is a major element of expert judgment involved in obtaining the MSA adjusted scales.

30. The MSA adjusted areas have been used to estimate the value of services from the worlds' biomes at different points of time in the past and estimates have also been made of the likely loss of services by 2050 if no action is taken. In the Costs of Policy Inaction study, Braat and ten Brink (2008) estimate that monetary losses in 2050 will run at around one per cent of GDP and cumulative losses from 2000 to 2050 will be around 7% of 2050 consumption. In a reworking of the data as well as a back-casting analysis Markandya and Chiabai (2013) find that losses from 1900 to 2000 were equal to between 1.4 and 3.8% of GDP in 2000.[17] For the period from 2000 to 2050, they get a range of losses equal to between 0.2 to 0.6% of 2050 GDP. If, however, account is taken of the net value of agricultural output from the land conversions then the net losses are much smaller in global terms although they can still be significant in some regions, particularly Africa. The use of the GLOBIO model is discussed further in the next section.

3. Application of estimates of biodiversity and ecosystem services in a cost-benefit analysis

31. Studies that have used estimates of biodiversity and/or ESS in a cost-benefit context can be divided into those that seek to inform policy at the global, national or regional level; and those that look at options for action at a more local level.

17 . These losses are not impacts *on* GDP, but GDP is used as a point of comparison.

3.1 Studies at the global, national and regional level

Global studies of different environmental-economic policies

32. A number of studies have used data on biodiversity and ecosystem services at these levels. Perhaps one of the most comprehensive is that by Hussain et al., 2011, 2013, which uses the GLOBIO model to estimate the change in MSA and services from selected biomes to 2050 under a baseline of no action against an alternative of a policy action, where the policy can take a number of forms.

33. The policy actions are summarised in Table 6, which also indicates key features of the baseline and how each scenario differs from that baseline.[18]

34. The analysis proceeds to evaluate the impacts of the different scenarios on biomes across the globe. The GLOBIO-IMAGE model has a spatial disaggregation at 0.5x0.5º grid cells. Almost all terrestrial cells are classified in one of the biomes resulting in 2.3 million "patches". The underlying Computable General Equilibrium (CGE) model with scenarios gives changes in MSA adjusted areas of grassland, tropical forest and temperate forests, wetlands, mangroves and coral reefs. It does not provide changes for other biomes, such as marine areas.

35. The changes in MSA areas in each cell were valued in terms of the loss or gain of the biomes, using a meta-analysis based approach that determined the values as a function of GDP per capita in the country, size of the biome, Net Primary Product in the areas and other local characteristics. The functions are not all well determined and not all coefficients used in the valuation are statistically significant. In addition to the benefits estimated using the meta-analysis, the study also values carbon sequestration benefits directly, based on changes in biomass. In spite of some shortcomings on the benefit transfer approach, the study gives a useful broad brush estimate of the benefits of going from the baseline to one of the scenarios. The authors compare these benefits with the costs of making the shift. They conclude that programmes to increase agricultural productivity or to reduce deforestation have a high benefit to cost ratios. On the other hand, the programme for increasing protected areas comes out as marginal, depending on what values are taken for the increased carbon savings. They also conclude that policies such as measures to stimulate a change of diet, change in agricultural trade and stricter climate regimes could not be assessed using this approach.

Table 6. Scenarios analysed using the GLOBIO-IMAGE model

Scenario	Baseline	Change compared to baseline
1. Agricultural Productivity	0.64% growth p.a. in yields	Investment leads to 40% increase in crop and 20% in livestock productivity
2. Reducing Post-Harvest Losses in forestry	Current losses around 30%	Losses decline by half to 15%
3.Better forest management	Current rates of logging continue	Reduced impact logging and increase in forest plantations
4. Protected Areas	Level of PAs = 14% of land area maintained	Increase of PAs to (a) 20% and (b) 50% of each region
5. Reduced Deforestation	Current trends continue	All dense forests protected from agricultural expansion
6. Stricter Climate Regime	Biofuel in 2050 modest (=0.5 Mn km^2)	GHG Concentration limited to 450 ppm with 4 Mn km^2 bioenergy area
7. Dietary Changes	Livestock doubles with population	(a) Willett diet with low meat or (b) no meat at all by 2050
8. Agricultural Trade	Current trade regime in force	Non-tariff barriers and subsidies removed so full trade liberalization by 2020

18 . Some of the scenarios could be compared with respect to the baseline in terms of costs and benefits. These were scenarios 1-4. The results show that both the agricultural productivity and the forestry scenarios have very high benefit cost ratios. The ratio for protected areas is much lower.

National ecosystem assessments

36. The MEA approach has given rise to assessments of the services provided by ecosystems at the national scale. The basic idea behind these national ecosystem assessments (NEAs) is to make available to policy makers "the findings of science concerning the causes of ecosystem change, their consequences for human well-being, and management and policy options" (MA, 2005). Ecosystem assessments "provide the connection between environmental issues and people, considering both the ecosystems from which services are derived and the people who depend on and are affected by changes in the supply of services" (Ash et al., 2010). In the review of NEAs actually carried out in the UK, Spain, Portugal and Japan, Wilson et al. (2014) note the diversity of approaches and degree of coverage. The UK made a major attempt to value the services currently provided in monetary terms, as well as making an estimate for some services of what is likely to be provided in the future under different policy scenarios. The services valued in the study are agricultural output, GHG emissions (the service being climate regulation) and recreational values of different landscapes, including urban greenspace. This was done at a spatially disaggregated level and was complemented by a lot of information on trends in ecosystem services in the form of physical indicators of habitat quality and on the causes of changes in these trends (Bateman et al., 2013).

37. In the case of the other three NEAs, virtually no monetary data are provided, although there are plans to carry out further work to make estimates of the values of ecosystem services in monetary terms.

38. While it would be an overstatement to say that the UK NEA carried out a full cost-benefit analysis of different development options (costs and wider economic impacts of the different alternatives are not fully accounted and a number of ESS are not valued, especially biodiversity and marine services), the exercise showed the importance of including ecosystem service values when evaluating different development plans. The ranking of the development options based on market values alone was very different from that based on a combination of market values and the values of ecosystem services. Furthermore the UK NEA had some impact by allowing for the immediate uptake of the evidence base that it provided. This also enabled the work of the UK NEA to be exposed to policy makers directly, thereby creating the opportunity to influence policy and has led to support for further strengthening ESS valuation in the UK (Wilson et al., 2014).

39. In other countries the impact has not been as great but awareness of the importance of ESS in economic terms is increasing and work to mainstream ESS values in development policy has started. Yet one is still a long way from having comprehensive coverage at the national level of the values of ESS, particularly marginal values associated with changes in policy (Wilson et al., 2014).

Other regional assessments

40. A number of regional assessments have been made, looking at one type of ecosystem service for a country or region and comparing the costs of different actions to enhance or protect these services with the benefits.

41. For biodiversity, Naidoo and Adamowiz (2005) compared different ways of conserving avian diversity in Uganda.[19] They compared the costs of doing so through increased fees for forest visits against converting agricultural land to forests and found that while a feasible increase in fees (based on surveys of willingness to pay to protect 80% of forest bird species) could indeed provide protection for that percentage, the same level of protection would be prohibitively expensive by purchasing agricultural land for ecological rehabilitation. The study thus estimated the benefits of the increased level of conservation through a contingent valuation survey and compared it with different methods of achieving a given level.

42. A study covering England and Wales estimated the willingness to pay of the public for sites of special scientific interest (SSSI) for current levels of services and benefits that these sites provide and for levels they would provide if they were maintained in a favourable condition (Christie and Rayment, 2012). The aggregate increase in willingness-to-pay (WTP) for the improvement was GBP 769 million per annum

19 . This study is discussed further under the biodiversity section.

compared to GBP 111 million annual costs of managing SSSIs to achieve the improved level. The study involved building up the WTP for individual ESS at each of the sites and then estimating the costs of conservation management to attain the improvements in services at the sites. As such it linked the benefits and costs of marginal improvements in a bottom-up manner in a way that allows for a detailed cost-benefit assessment.

43. An important assessment at the national level for the US was that of Pimental et al. (1995), which compared the costs of soil erosion with the benefits of preventing it through conservation methods. Although not couched in ESS terms (the study was conducted well before the term was invented) it is essentially a valuation of the agricultural services provided by erosion control (see Table 1).[20] It notes that the world has lost a quarter of its arable land over the period 1955-1995 on account of soil erosion and continues to lose it at the rate of 10 million hectares per year. Damages caused by erosion are valued in terms of the additional energy and nutrients and water needed to maintain a given level of production, as well as the costs of siltation and damage caused by soil particles entering streams and rivers and harming habitat quality. Total damages amount to about USD 100 ha^{-1} yr^{-1}, which can be compared to the costs of conservation through methods such as ridge planting, no-till cultivation, contour planting, cover crops and windbreaks. They estimate the costs of reducing erosion rates from 17 tons ha^{-1} yr^{-1} to about 1 tons ha^{-1} yr^{-1} on cropland and pasture land would be around USD 45 ha^{-1} yr^{-1}, thus providing a healthy net benefit in overall terms.

44. Such studies are examples of what has been achieved at the national or regional level in comparing the costs and benefits of different policies and measures to conserve ESS or to improve or maintain the services we have. The valuations of the benefits have considerable margins of error and there are, to be sure, important gaps in the valuation literature. Furthermore, in terms of actions to be implemented, governments would need more detailed information in which locations and for what technologies or measures the benefits exceed the costs. In spite of these limitations, they are a useful guide to policy making and it is unfortunate that more have not been carried out.

3.2 Local cost-benefit studies using ecosystem valuations

45. This section looks at cost-benefit analyses of ESS carried out at the local level for specific services in specific habitats. The coverage of studies is derived from a review of the main journals and databases that hold information on ESS valuation. It cannot be guaranteed to be comprehensive given the huge amount of literature there is on ESS valuation. It has sought to extract those studies in which "benefit cost" or "cost-benefit" was explicitly mentioned either in the key words or in the abstract and which had something interesting or new to contribute. In addition, only those that gave some quantitative information on both benefits and costs were retained.

45. The studies are divided into biodiversity; agriculture and grasslands; forests; freshwater habitats (including inland wetlands); and marine and coastal habitats. These categories are not watertight – for example a study may be classified under freshwater but also include aspects relating to agriculture or biodiversity. At the end of the section some general findings on cost-benefit assessment at the local level are offered.

Biodiversity

46. Many studies provide economic estimates for biodiversity. A recent review of the literature by Bartkowski, Lienhoop and Hansjürgens (2015) identified 123 distinct studies, about half of which were from Europe. Notably ecologically more valuable areas in the developing world remain understudied. More than 80% of them used contingent valuation or choice experiment methods to elicit values, with the rest using either other stated preference or travel cost methods. A similar share used biodiversity proxies belonging to two attribute categories, based on the notions of habitat protection and rare/endangered/alien species respectively. As the authors state: "Even though the complexity and multi-dimensionality of the biodiversity concept are well recognised, only a few studies tried to approach it in a multi-attribute way".

20 . This study is also referred to in the agriculture and grassland section.

47. From this literature, a small number of studies that actually used the valuation data to make some kind of benefit cost assessment were found.[21] These are summarised in Table 7. They demonstrate that the main problems are: (a) pick up the multi-dimensional attributes of biodiversity in a set of metrics that can then be valued and (b) to link changes in these metrics to specific policies and measures. Naidoo and Adamowicz (2005) link avian diversity to forest areas but, as they acknowledge, the relationship is more complex than their modelling. Likewise, Markandya et al. (2008) find it difficult to estimate the recovery in vulture numbers when the cause of the decline is tackled through a change in the drug administered to cattle. The Portuguese study of the Castro Verde (Marta-Pedroso et al., 2007) uses a CV valuation of the biodiversity that the programme seeks to protect but the study is small and the results would need to be confirmed.

48. Given there are a large number of studies that value biodiversity but only a handful that undertake a cost-benefit assessment of biodiversity changes, the reason is most likely to be difficulties in dealing with the difficulties described above.

Agriculture and grasslands

49. There are many studies that have evaluated alternative uses of land – from forest to agriculture or vice-versa, converting agricultural land back to forest or to protected areas. Some of them are considered in the subsequent section on forests. Here the focus is on different ways of managing land that is predominantly agricultural and how including ESS can change the assessment of different practices.

50. From a wide literature estimating how different regulating ESS impact on food and fibre provision, there are only a handful that carry out something akin to cost-benefit analysis. These are summarised in Table 8. The study by Pimental et al. (1995) on the provisioning and cultural service losses due to soil erosion and the costs of preventing it has already been mentioned. Valuation of losses to the quality of rivers and streams due to the deposition of soil particles is very limited though much more is known now about these effects. Hence, the overall costs would probably be higher if re-estimated using the latest data. The Gascoigne et al. study is a good example of the trade-offs between conservation and conversion of grasslands in the US, with some of the important non-market ESS being valued as part of the comparison.[22] The Ghalley and Porter paper is not a cost-benefit analysis as such but provides some of the key ingredients for comparing the ecosystem functions of soil water storage and nitrogen mineralisation and ecosystem services of food and fodder production and carbon sequestration under different farming practices for winter wheat. The key parameter is soil organic matter which is strongly linked to these functions and services and varies with management systems.

Forests

51. The trade-offs in conserving forests versus allowing the land to be used for other purposes (including temporary use and permanent conversion) have been analysed in a wide range of studies covering topics such as hydropower development, increased agricultural production etc. Cost-benefit methods have been used to appraise such choices by national governments and international financial institutions for many years and have considered losses of environmental services when choosing development options.[23] Here the focus is on some of the more recent studies that have valued ESS in a relatively comprehensive way when making such comparisons and highlight the important recent developments that they demonstrate.

52. Table 9 summarises the results from four such studies. The Beukering et al. (2003) study of the Leuser national park in Indonesia compares marketed and non-marketed ESS under three different scenarios and looks in detail at a very wide range of ESS. In spite of this, coverage is partial for services such as biodiversity.

21 . An earlier study that looked at the costs of biodiversity conservation was Ward and Booker (2003). They estimated examines economic costs of meeting habitat needs for the silvery minnow to save it from extinction in the Rio Grande. The study concluded that this could be done by changing flows in such a way that some riparians lost out while others would gain, but in total the economic value of the changes would be positive.

22 . There are other examples of trade-offs between conservation and expansion in which ESS have been valued extensively. They have been classified under forest and freshwater habitats as that is the main feature relevant to them.

23 . For a review of methods and some case studies, see Markandya et al., (2002).

Table 7. Cost-benefit studies of policies and measures for ESS that impact on biodiversity

Study	ESS valued & methods used	Main findings	Comments
Naidoo and Adamowicz (2005)	Avian species diversity in forests in Uganda valued through choice experiments on tourists visiting the parks	While an increase in fees based on surveys of willingness to pay to protect 80% of forest bird species could provide protection for that percentage, the same level of protection would be too expensive by purchasing agricultural land for ecological rehabilitation.	Species -forest relationship modelling and assumption of species independence need to be verified. Results could be of direct use for biodiversity conservation
Marta-Pedroso et al. (2007)	The Zonal Program of Castro Verde aims to avoid the loss of suitable habitat for steppic bird species by financial compensation for farmers who agree to maintain certain farming practices. Study compares the costs of the program in terms of soil erosion and compensation against the benefits of species preservation and landscape identity as measured through a CV survey	Cereal steppe is the main landscape unit within the Municipality of Castro Verde, Southern Portugal, making up about 82% of the agricultural area. Although marginal, with yield less than half the average yield in the European Union, this low intensity dry land cereal farming holds a large proportion of steppic bird species threatened at their global scale of distribution. The benefits of preservation are well in excess of costs as estimated in the study.	The CV survey is limited small and rather limited in scope and estimate of erosion based on replacement costs is an imperfect method of cost assessment. Nevertheless given the large difference between benefits and costs the results indicate the value of the program.
Markandya et al. (2008)	Compares the costs of the decline in vultures in India through the increase in feral dog population and its impacts on cases of rabies with the costs of eliminating the source of the vulture decline – a drug called diclofenac used to treat cattle	The study shows the high costs of vulture decline in monetary terms. Costs of alternative drugs to treat cattle are available and though expensive would have significant benefits in restoring vulture numbers, along with other measures such as captive breeding.	The study is a partial cost-benefit assessment but does not carry out a full comparison of the costs of replacing diclofenac.
Finger and Buchmann (2015)	Estimated the potential risk-reducing effects of species diversity in terms of yields and their temporal stability from a farmer's perspective using panel data for plots in Germany	Find empirical evidence for the risk-reducing effect of species diversity and the economic assessment reveals significant insurance values associated with diversity for a risk-averse decision maker.	Not a full cost-benefit analysis but it indicates that insurance costs could decline with the presence of species diversity in grassland and croplands

Table 8. Cost-benefit studies of policies and measures for ESS that impact on agriculture and grasslands

Study	ESS valued & methods used	Main findings	Comments
Pimental et al. (1995)	Estimates damages caused by soil erosion in the US and compares them against the costs of avoiding erosion. Erosion is valued in terms of additional energy, nutrients and water needed to maintain a given level of production, as well as the costs of siltation and damage caused by soil particles entering streams and rivers and harming habitats.	Total damages amount to about USD 100 ha^{-1} yr^{-1}, Costs of conservation through methods such as ridge planting, no-till cultivation, contour planting, cover crops and windbreaks are estimated at 17 tons ha^{-1} yr^{-1} on cropland and pasture land would cost around USD 45 ha^{-1} yr^{-1}, thus providing a healthy net benefit in overall terms.	Valuation methods do not include the recent work on damages from pesticides and fertilizers on streams and rivers.
Gascoigne et al. (2011)	Compares the societal values of agricultural products and ecosystem services produced under policy-relevant land-use change scenarios, and explore the effectiveness of mitigating loss with conservation programs in the native prairie pothole regions of Dakota. Crops valued using market data. ESS of carbon sequestration, sedimentation and water fowl production estimated by biophysical models and valued by benefit transfer.	Four scenarios evaluated for a 20 year period ranging from aggressive conservation to extensive conversion for agriculture in terms of changes in market and non-market ESS and including any costs incurred in implementing these scenarios. In benefit cost terms scenarios where native prairie loss was minimized, and Conservation Reserve and Wetland Reserve lands were increased, provided the most societal benefit.	Study looks at how benefit cost rankings change with uncertain parameters such as social value of carbon. ESS coverage is not complete but major issues are addressed. Valuation methods of benefit transfer also entail uncertainties not fully examined.
Ghaley and Porter (2014)	Links soil organic matter (SOM) in winter wheat production system in Denmark to ecosystem functions (EF) of soil water storage and nitrogen mineralization and ecosystem services (ES) of food and fodder production and carbon sequestration using DAISY a soil–plant–atmosphere system dynamic model, which simulates plant growth and soil processes. All values are based on market prices of inputs and outputs except carbon, which is taken from the European Emissions Trading price.	There is a positive relationship between the SOM and different EF and ES provision, which are particularly soil-based. The depletion of SOM has adverse effects on soil productivity and on climate change by releasing carbon into the atmosphere. Management practices like inclusion of grass into crop rotations, adoption of catch/green manure crops, residue management, balanced manure and fertilizer application and tillage intensity can have significant positive effects on SOM.	Not a benefit cost analysis as such but identifies practices that manage SOM for which it gives the associated values to EF and ES. The aim then is to select those that maximise social value.

Table 9. Cost-benefit studies of policies and measures for ESS that impact on forests

Study	ESS valued & methods used	Main findings	Comments
Beukering et al. (2003)	Valued alternative uses of the Leuser National Park in Indonesia. Economic benefits considered: water supply, fisheries, flood and drought prevention, agriculture, hydro-electricity, tourism, biodiversity, carbon sequestration, fire prevention, non-timber forest products, and timber. Production functions and market prices used to value all except tourism and which were valued from a CV survey, biodiversity from values of medicinal plants and carbon sequestration from IPCC estimates.	Three scenarios examined were: deforestation, conservation and selective use. Total Economic Value at 3% discount rate over 30 years showed the conservation option was most beneficial (USD 9 538 million) compared to selective use (USD 9 100 million), and deforestation (USD 6 958 million). Discount rate used was 4%.	Sensitivity of results to key parameters was carried out and results still held. Range of services covered is wide but biodiversity values are only partial.
Naidoo and Ricketts (2006)	A spatial evaluation of the costs and benefits of conservation for a landscape in the Atlantic forests of Paraguay. Considered five ecosystem services (sustainable bush meat and timber harvest, bioprospecting for pharmaceutical products, existence value, and carbon storage in aboveground biomass) and compared them to the opportunity costs of conservation	Found a high degree of spatial variability in both costs and benefits over a relatively small (3 000 km^2) landscape. Benefits exceeded costs in some areas, with carbon storage dominating the values and swamping opportunity costs. Other benefits associated with conservation were more modest and exceeded costs only in protected areas and indigenous reserves	Valuation of some services (e.g. bush meat, bioprospecting and existence values) has large uncertainties. Important finding is the spatial variability of benefits and costs.
Olschewski et al. (2012)	Compares WTP for reduced risks of avalanche damage in the Swiss alps based on a choice experiment against the costs of providing such protection using a variety of methods including forest.	WTP for avalanche protection of forests is substantially higher than the costs of silvicultural measures to maintain protection forests and similar to measures such as logs and grills but less than engineering solutions with steel bridges and nets.	Uses risk based evaluation techniques to compare individual preferences for ESS versus engineering based solutions to risk reduction.
Ninan and Inoue (2013)	Values a range of ESS from the Oku Aizu forest reserve in Japan and compares it to other land uses. Values I include water conservation, water purification, nutrient recycling, air pollution absorption (valued in terms of alternative cost of providing the service), soil protection (valued in terms of the value of lost productivity), carbon fixation (valued at the carbon price and damage estimates) and recreation (valued using stated preference methods)	This value for the Oku Aizu forest reserve ranged USD 1.427–1.482 billion or about USD 17 016–17 671 per ha. These are considered to be well in excess of other uses of the land.	Even without considering: habitat, biodiversity, some cultural values, flood protection, pollination and NTFP this one is one of the more comprehensive, combining regulation and provisioning services. Issue of double-counting may arise.

53. Ninan and Inoue (2013) have a similar assessment for the Oku Aize forest reserve in Japan, where they also value a wide range of regulating and provisioning services and conclude that alternative uses of the land would have lower value. There is a question of double-counting that arises with respect to including both regulating and provisioning services. For example, air pollution absorption is valued using the cost of alternative ways of reducing the pollutants from the atmosphere while recreation is valued in terms of WTP through stated preference methods. Yet part of the WTP for recreation is because the air is cleaner in the parks, so there is some double-counting. This may be addressed by eliciting the WTP separately from benefits of clean air but that is difficult.

54. Naiddo and Ricketts (2006) is interesting in comparing costs and benefits on a disaggregated spatial scale for forests.[24] It finds that the balance between the two can be very different, depending which locations are considered, showing that spatial cost-benefit analysis can powerfully inform conservation planning. It can help one to understand the synergies between biodiversity conservation and economic development when the two are indeed aligned and to clearly understand the trade-offs when they are not. The problem is that such analyses are data-intensive and the information is not always available. Their study involves a number of valuations that involve benefit transfer and are highly uncertain (valuation of bush meat, bioprospecting and existence values of forests). This uncertainty is not really reflected in the reported results, something that is commonly the case in the literature.

55. The last study (Olschewski et al., 2012) is unusual in valuing ESS in a risk reduction context. The authors look at the WTP for reduced risks of avalanche damage in Switzerland and compare that with different methods of achieving the reduction, including some that involve silvicultural methods and other that are more engineering based. The forest methods have other benefits which are not included in the study but it shows that risk reduction can be an important ESS that is often not included in the list of benefits from different natural habitats.

Freshwater habitats (including inland wetlands)

56. There are many more studies carrying out a cost-benefit analysis of freshwater habitats and wetlands than of any other habitat. Table 10 summarises ten of them to show some of the key features that emerge in this literature.[25]

57. First, it is noted that even though studies are becoming more comprehensive, the number of ESS they cover is still incomplete. In this sense the estimates of benefits can be seen as a lower bound. In spite of that one finds riverine and wetland protection has a high benefit relative to cost. Second, some studies value packages of benefits from a conservation programme, including changes in several ESS; they do not look at the individual components of that package. In such cases it is not possible to say which of the services are the most important and whether an alternative programme focusing on a different package would be more beneficial. One study (Holmes et al., 2004) has noted, however, that when individual components are valued, the benefits of a package are often not equal to the sum of the benefits of the items in the package valued individually – there is a phenomenon referred to super-additivity. The sum is often greater than the individual components. This suggests that one needs to consider packages of benefits but vary the weight of components within it to get a fair representation of the possible options.

58. The method of assessment is dominated by stated preference methods (contingent valuation or choice experiments). These are combined with estimates of changes in services of markets products based on market prices and others such as carbon capture and sequestration, which are based on

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24 . Spatial disaggregation has now become more widespread in ESS applications that cover wider areas, such as the global analysis of Hussain et al. (2011) and the NEA of the UK, reported earlier in this section. It is still, however, not as common as it needs to be.

25 . More examples of benefit cost analysis involving wetlands, mangroves and river basins can be found in Hanley and Barbier (2009), Barbier and Markandya (2012).

biophysical estimates of changes in flows of services valued from other studies of the cost of carbon or from markets in carbon abatement. Some researchers have commented on how much respondents understand about the ecological impacts of different restoration packages. The link between the programmes and the changes in ESS is complex and respondents generally have to take on trust what they are informed about these relationships. Their response will be influenced in part by how much they believe in the accuracy of that information.

59. The study by Amigues et al. (2002) attempted to compare the benefits of habitat preservation in terms of WTP with the costs of preservation estimated in terms of the willingness to accept (WTA) payment on the part of the parties that would lose out from the preservation. The study had considerably difficulties in eliciting WTA and the results were relatively difficult to interpret for that reason.

60. An issue that comes up in cost-benefit analysis is the choice of the discount rate. Projects and programmes involving conservation and/or sustainable use can have long lifetimes, with benefits over many years into the future. A high discount rate will place a low present value on such projects, while a low rate is difficult to justify on resource allocation grounds. Recent literature on the topic has advocated the use of declining rates, so that a higher discount rate is applied to earlier periods than to later periods.[26] Birol et al. (2010) look at the implications of different discount rates for a project to replenish a depleted aquifer in a water-scarce region and show how the overall values are sensitive to the choice of the rate.

61. While most projects for ESS in freshwater habitats relate to conservation versus business-as-usual, not all are on that issue. Honey-Rosés et al. (2013) show how additional riparian forest restoration along the Llobregat River in Spain could generate economic savings for water treatment managers in excess of the costs of the restoration,[27] Birol et al. (2010) look at the role of aquifer regeneration in enhancing ESS, Yang et al. (2008) look at the possible benefits of constructed wetlands through improved water quality and groundwater protection in China and Grossman considers the role of floodplains to act as a nutrient retention source for the River Elbe in Spain.

62. Lastly it is noted that the bulk of the studies are from North America or Europe, with only a few from developing countries or emerging economies. Apart from the China study cited here, there have a number that conducted some kind of cost-benefit analysis for mangroves in South East Asia (Barbier, 2007) and for water basins in South Asia and Africa (Lopez and Toman, 2006) but there are few of them relative to the number of areas where ESS are being degraded and need to be evaluated.

26 . Declining discount rates will only make a difference if the project under consideration has a long lifetime. Rates can, for example, drop in 0.5% steps after 30 years, 75 years, 125 years, 200 years and 300 years. Most ESS projects are not evaluated over such long periods, so it matters what initial rate is used.

27 . This is something that has been noted before in relation to water supply in New York and the role of the Catskill Mountains in water purification, but that has been questioned and not subjected to a detailed cost-benefit analysis. See *www.perc.org/articles/catskills-parable*.

Table 10. Cost-benefit studies of policies and measures for ESS that impact on freshwater systems

Study	ESS valued & methods used	Main findings	Comments
Loomis et al. (2000)	Five ecosystem services that could be restored along a 45-mile section of the Platte river were valued: dilution of wastewater, natural purification of water, erosion control, habitat for fish and wildlife, and recreation. Method was WTP for improvements in these services based on a higher water bill.	WTP of households living along the river yields a value of USD 19 to USD 70 million, depending on whether those refusing to be interviewed have a zero value or not. Even the lower bound benefit estimates exceed the high estimate of water leasing costs (USD 1.13 million) and conservation reserve program farmland easements costs (USD 12.3 million) necessary to produce the increase in ecosystem services.	Valuation is for a 'package' of ESS, but it would be useful to have information on individual components. Not all ESS may have costs in excess of benefits.
Amigues et al. (2002)	The Contingent Valuation Method (CVM) was used to obtain the willingness to pay (WTP) of households in the contiguous area of the Garonne River near Toulouse, France for different stretches for habitat preservation, and the willingness to accept (WTA) of households that currently own land on the banks of the river to provide a strip of riparian land for habitat preservation.	The range of WTP values, which represented the benefits, was higher than the range of WTA values, which represented the cost.	WTA observations were few and refusal rates high. This made the analysis unreliable. Alternative methods of assessing costs were tried and results found to still hold.
Holmes et al. (2004)	A study was undertaken to estimate the benefits and costs of riparian restoration projects along the Little Tennessee River in western North Carolina. Restoration benefits were described in terms of five indicators of ecosystem services: abundance of game fish, water clarity, wildlife habitat, allowable water uses, and ecosystem naturalness. A sequence of dichotomous choice contingent valuation questions were presented to local residents to assess household willingness to pay increased county sales taxes for differing amounts of riparian restoration.	Results showed that the benefits of ecosystem restoration were a non-linear function of restoration scale and the benefits of full restoration were super-additive. Costs of riparian restoration activities were estimated by collecting and analysing data from 35 projects in the study area. The benefit/cost ratio for riparian restoration ranged from 4.03 (for 2 miles of restoration) to 15.65 (for 6 miles of restoration).	Finding that benefits are super additive in the sense that total benefits from a whole package are greater than the sum of the benefits of each part is interesting. Authors question how much respondents understand about the ecological impacts of different restoration packages.
Birol et al. (2006)	A choice experiment is used to estimate the values of changes in several ecological, social and economic functions that Cheimaditida wetland provides to the Greek public. ESS include biodiversity (no of species and size of habitats) and open water surface area.	The WTP for the different components is valued individually and used to construct management scenarios with low, medium and high impact on the two ESS. These are compared with the costs of the scenarios and in each case the benefits exceed the costs, with the biggest difference for the high impact scenario.	Has the advantage of valuing individual components of the ESS but only two are considered whereas others may also be important.
Karanja et al. (2008)	Environmental flow provision in Gwydir Water Sharing Plan in NSW Australia aims at improving wetland and aquatic ecosystems' health. However, irrigators are concerned that the Plan could lead to significant reductions in irrigation water. Study valued four ecosystem services from provision of environmental flow: water bird-breeding events, habitat provision, improved wetlands grazing and biodiversity benefits (native fish species). Method used was benefit transfer.	The present value economic cost related to provision of environmental flow (40 gigalitre), valued as the opportunity cost of foregone agricultural profit in Gwydir was AUD 15 million. The ESS benefits gained totalled AUD 94 million, using NSW households. The NPV was AUD 79 million or an annual equivalent of AUD 160/ML/yr at a 7% discount rate.	Valuation of costs of water diversion from agriculture in terms of shift of crops from cotton to wheat is perhaps too simple. All ESS valuations are based on benefit transfer. Study is interesting in comparing costs and benefits of low flow regulation.

Yang et al. (2008)	Using the Constructed Wetland (CW) located at the Hangzhou Botanical Garden in China the contingent valuation method (CVM) and shadow project approach (SPA) were applied to estimate the economic values of CW system ecosystem services.	Valuations of the benefits of the CW through improved water quality in the fishpond and groundwater protection in terms of WTP or the costs of alternative methods of achieving the same goals are found to exceed the costs of the CW	Specific ESS provided by a CW is valued in terms of WTP. Benefit cost ratio with WTP method is close to one, implying further work needed.
Birol et al. (2010)	Considers the implementation of a water-resource management plan in a water-scarce region of the world, namely Cyprus*. The plan proposes to replenish a depleting aquifer with treated wastewater. The proposed methodology identifies the key stakeholder groups (farmers and the general public) who derive economic values (benefits) from implementation of this plan, and then uses stated-preference methods to capture the total economic value of these benefits.	Benefits are weighed against the total costs of implementing the plan in a long-run cost-benefit analysis. The results are estimated over 200 years (estimated life of the aquifer under proposed management with constant (3.5% and 6%) and declining discount rates. Although net benefits exceed costs with all discount rate the net present value is much higher with declining discount rates.	Interesting to consider impacts of different discount rate. Assumptions that the cost of the recharge will not decrease due to technological changes and that water consumption will remain constant over that time as will WTP of farmers and residents are questionable.
Jenkins et al. (2010)	This study assesses the value of restoring forested wetlands via the U.S. government's Wetlands Reserve Program (WRP) in the Mississippi Alluvial Valley by quantifying and monetizing ecosystem services. The three focal services are greenhouse gas (GHG) mitigation, nitrogen mitigation, and waterfowl recreation. Valuation of GHGs is in terms of the social cost of carbon. Nitrogen mitigation is valued using studies of its impacts on hypoxia in the Gulf of Mexico and wildfowl recreation is valued from previous studies.	The total ecosystem value of the wetlands restoration. Social welfare value is found to be between USD 1 435 and USD 1486/ha/year. Current market values of the lands is only USD 70/ha and costs of restoration are so small they would be recovered in one year. Likewise farmers would need a much smaller compensation to convert land to the programme than they earn from the land.	Only three ESS are valued and benefits (hence estimates area lower bound) are mostly based on benefit transfer.
Grossman (2012)	Restored floodplains provide phosphate and nitrogen nutrient retention services in the river Elbe. The paper estimates the shadow prices of these services and compares it to the costs of restoration. Shadow prices give the cost per ton retained based on the next best alternative method of achieving the reduction in nutrients in the river.	The restoration program is based on an optimisation model that attains a given retention level at least cost. In spite of the large investment costs for dike realignments, the nutrient retention effects alone can in many cases generate sufficient benefits to generate an economic efficiency gain.	Other abatement efforts to reduce nutrients in the river may change the benefits of the restoration program. These were not included in the study.
Honey-Rosés et al. (2013)	In the Llobregat River in north eastern Spain, higher stream temperatures require water treatment managers to switch on costly water treatment equipment especially during warm months. This creates an opportunity to align the economic interests of downstream water users with the environmental goals of river managers. A restored riparian forest or an increase in stream flow could reduce the need for this expensive equipment by reducing stream temperatures below critical thresholds. Study estimated the impact of increasing shading and discharge on stream temperature at the intake of the drinking water treatment plant.	The value of the stream temperature ecosystem services provided by existing forests is EUR 79 000 per year for the water treatment facility, while additional riparian forest restoration along the Llobregat River could generate economic savings for water treatment managers in the range of EUR 57 000 – EUR 156 000 per year. Stream restoration at higher elevations would yield greater benefits than restoration in the lower reaches. Moderate increases in stream discharge (25%) could generate savings of EUR 40 000 per year.	The study is interesting in estimating the ESS benefits in terms of what they provide for a specific economic function, namely water treatment. Other studies have also shown the benefits of ecosystems in reducing water treatment costs.

* The information in this document with reference to "Cyprus" relates to the southern part of the Island. There is no single authority representing both Turkish and Greek Cypriot people on the Island. Turkey recognises the Turkish Republic of Northern Cyprus (TRNC). Until a lasting and equitable solution is found within the context of the United Nations, Turkey shall preserve its position concerning the "Cyprus issue". The Republic of Cyprus is recognised by all members of the United Nations with the exception of Turkey. The information in this document relates to the area under the effective control of the Government of the Republic of Cyprus

Marine and coastal habitats

63. Research on of marine ESS is rarer than for other categories, and while valuation studies are becoming more common,[28] cost-benefit assessments are very few. In Table 11, four of these are summarised.

64. The first is a study comparing the gains for mariculture against the losses caused by the decline in services such as of oxygen production, climate regulation and waste treatment**.** In the particular site looked at (Sanggou Bay in China), the authors find the gains can be greater than the losses when good management practices are followed. The study does not address all the issues arising in such an assessment (changes in ESS values over time and full costs of mariculture management), but is nevertheless a good point of departure for further work.

65. The second study is a comparison of two different methods of coastal restoration in Louisiana in terms of costs but also benefits to habitat, water quality and storm surge protection. Benefit estimates are not closely tied to the individual sites evaluated but the interesting finding is that for a wide range of such benefits one method of restoration is dominant. The study effectively then uses a cost-effectiveness method of assessment, supplemented by some information on the benefits.

66. The third paper by Waite et al. (2014) is not a cost-benefit analysis as such but is useful in understanding how valuation studies of marine habitats have influenced policy in the Caribbean, Mexico and the state of Florida in the US. The authors find cases where well-executed valuation studies have had an impact on the creation of marine parks and protected areas, charges for visits to such parks and successful campaigns against offshore oil drilling. It is interesting to note, however, that while the values of ESS were important, cost-benefit analysis appears to have played a limited role in such decisions. An exception is the Belize Integrated Coastal Zone Management Plan, which compares ecosystem service provision and value under three scenarios developed by stakeholders: 'conservation', 'development' and 'informed management'. The study by Clarke et al. (2013) estimates the net avoided damages to ESS over the period to 2025 by using the InVest Model to estimates changes in marine habitats (coral, sea grasses and mangroves) and in the ESS such as fishery, agriculture, aquaculture, tourism and recreation, coastal development and coastal protection. It concludes that the informed management scenario has the highest net avoided damages. The study is an interesting application of ESS valuation to planning of marine areas, but it is something of an exception.

67. The last study is one that seeks to estimates benefits and costs of expanded cruise tourism by widening shipping channels in Bermuda against damages to coral and thereby to related tourism. Although it does not actually estimate changes in tourism as a result of loss of corals, preliminary figures indicate the relative importance of the two and suggest that a precautionary approach would be to avoid such losses for the sake of an increase in cruise ship visitation rates.

28. For a recent review on marine ESS, see Nunes and Gowdy (2015).

Table 11. Cost-benefit studies of policies and measures for ESS that impact on marine and coastal habitats

Study	ESS valued & methods used	Main findings	Comments
Zheng et al. (2009)	The model not only calculates market income of mariculture but also monetises the positive and negative effects of mariculture activities on ecosystem services. The model is applied to a typical mariculture bay in China (Sanggou). ESS valued are food production (market prices), oxygen production and waste treatment (N and P fixed by phytoplankton and kelp) (by costs of replacement) and carbon sequestered (at market price of traded carbon).	Development of mariculture has benefits of food production but results in decreases in in the value of oxygen production, climate regulation and waste treatment. The results show that mariculture development has a positive benefit to cost ratio given the improvements in management practices.	Costs of establishing the mariculture system not accounted and changes in future values of ESS not allowed for. Paper is unusual in showing how mariculture has a net social benefit with the right management. Further sensitivity analysis is needed.
Caffey et al. (2104)	Dredge-based “marsh creation” (MC) and diversion-based (DIV1) restoration projects were selected through a review of authorized projects submitted for coastal restoration in Louisiana. The average annual ecosystem service value was taken as USD 4410/acre/year, which represents the aggregate of annual services values for habitat (USD 282), water quality (USD 866) and storm surge protection (USD 3 653). Values based on benefit transfer	The study estimated benefit to cost ratios for different restoration projects in selected areas. In all case simulations, the projected benefit to cost ratio of MC projects exceeds that of DIV1 projects for projects of similar target scale. Study notes that break-even ESS values are lower for MC and DIV1.	Values of ESS are based on benefit transfer and may not apply to all sites. Some sensitivity analysis is needed and further primary ESS estimates may be warranted but cost effectiveness analysis indicates preference for MC.
Waite et al. (2014)	Identified 17 ‘success stories’ where economic valuation for coral reefs, mangroves, sea grasses and beaches has led to improved decision making in the Caribbean, Mexico and the US. Methods for valuing ESS included stated preferences (8 studies), production function and market prices (9 studies), to travel cost (3 studies) and hedonic price (1 study). Some studies used more than one method.	Paper does not report benefit cost analysis as such but shows how the valuation studies were used to justify, support and advocate policies that protected coastal ecosystems. Important factors influencing use of valuation studies are: threats to resources, dependence on them, good governance and ability to show how ESS impact on users, a clear policy question that needs addressing with relatively accurate estimates of values.	The role of benefit cost analysis is mentioned for only one case. In general the study considers other factors as more important.
Van Beukering et al. (2015)	Values of coral reefs to tourism and WTP for preservation in Bermuda are estimated using travel cost and stated preference methods. These are compared against revenues from cruise ship tourism. Program to upgrade shipping channels may lead to impacts on coral reefs which would affect that revenue source.	Paper sets out revenue streams from coral reefs (USD 406 million p.a.) against that from cruise tourism. Increases in cruise tourism via upgrading shipping are not directly compared against losses due to possible damage to coral but figures suggest precautionary approach.	Does not carry out a full cost-benefit analysis but lays out the items needed to make such an analysis. Initial figures suggest that coral protection should take precedence over development.

Summary

68. This section has demonstrated the wide range of applications of cost-benefit assessments using ESS values to evaluate different policies and measures. Yet what is perhaps striking is the relatively few studies that actually conduct a full-blown cost-benefit analysis relative to the number of studies estimating the value of ESS in various contexts. The discussion identifies some of the reasons for this. Probably the most important is the need to estimate changes in ESS as a function of a policy change rather than value the ESS in their current form. The additional data required to do that is not easily available and cannot always be collected.

69. A number of other points emerge from the review:

1. The majority of cases studies have resorted to primary studies of the value of ESS. Benefit transfer methods have also been used when more approximate values are sufficient for the purposes of the analysis but error ranges are greater and possibly the results are less convincing to policy makers.
2. The methods used vary a lot, with stated preference being perhaps the most common, along with production function methods that provide estimates of physical changes in goods and services, which are then valued using external price or cost data. Other methods such as travel cost and hedonic pricing are relatively infrequent.
3. The range of ESS covered in studies has been increasing and is impressive but it is rarely comprehensive. Thus estimated values have often to be seen as lower bounds to the impacts of policy changes. Related to that are two factors. One is the finding that the value to individuals from a package of changes in ESS is often not equal to the sum of the individual values, in which cases a range of packages need to be considered. The other is the problem of double-counting when both regulation and provisioning services are included. Study design has to be careful to avoid that.
4. The studies do conduct some sensitivity analysis but, given the range of uncertainties more should be done. Parameters for which sensitivity is rarely reported include the discount rate and rates of change of unit values of ESS in the future.
5. The kinds of polices evaluated vary, with trade-offs between development and conservation being the most common, but recent studies have also looked at different ways of mitigating risk, provision of ESS through artificial structures and provision of some services such as clean water through management of habitats.
6. Recent work has shown the importance of a careful consideration of spatial variability. ESS are not equal across space and neither are the costs of policies and measures to protect them.
7. It is rare for studies to look at the distribution of benefits and costs, which is of great importance as a complement to cost-benefit analysis. Some studies do identify gainers and losers in some detail but more often the analysis is in terms of overall net benefits.

4. Impacts of ESS valuation and cost-benefit analysis on decision-making

70. How much has the work on ESS valuation, especially through cost-benefit analysis, impacted on policy making in the field of environmental management? As noted, a number of studies have had some impact. Section III indicated how the UK NEA, which was in part a cost-benefit assessment, did have some impact in making UK environmental policy more evidence-based and in bringing economic values of ecosystems in the mainstream discussions of economic policy. However, in other countries where the NEAs did not go so far in estimating costs and benefits there was also an increase in awareness of the importance of ESS in economic terms (Wilson et al., 2014). Other regional assessments have also shown how cost-benefit assessment can be used in policy. A study which estimated the benefits of sites of special interest (SSSI) in England and compared them with the costs of conservation fed into the process for the design and management of such sites (Christie and Payment, 2012). In the case of biodiversity, the study comparing the damages caused by the decline in vultures with the costs of measures to reverse the loss affected regulations on drugs that were in part responsible for the loss (Markandya et al., 2008). In the case of coastal zone planning the study by Clarke et al. (2013) was a detailed cost-benefit assessment of alternative development scenarios for coasts of Belize, which did impact on decision-making there.

71. At the national level, there have been reviews in Australia that point to the role ESS thinking has played in the management of natural resources. Pittock et al. (2012) cite examples of ESS valuations having influenced management decisions in land use allocation (Little Desert area of Victoria), various catchments, management of water allocation in the Murray Darling Basin and support for indigenous land and sea ranger programs. The authors do not specify how much of the role of ESS was based on economic values in monetary terms and what part was played by cost-benefit analysis but they do stress that ecosystem services frameworks have facilitated strategic dialogue within and among governments at state and national scales.

72. There is a sense therefore that the overall impact has been positive. As De Groot et al. (2010) note:

> "Although consensus on a coherent and integrated approach to ecosystem service assessment and valuation is still lacking, and empirical data is still scarce, efforts to fill these gaps have changed the terms of discussion on nature conservation, natural resource management, and other areas of public policy. It is now widely recognized that nature conservation and conservation management strategies do not necessarily pose a trade-off between the ''environment'' and ''development'' but that investments in conservation, restoration and sustainable ecosystem use generate substantial ecological, social and economic benefits" (p. 272).

73. A similar view is expressed by Atkinson et al. (2012). They note challenges in the use of information on ESS values as including the need for a greater understanding of ecological production, especially as it relates to spatial variability, the significance of the gaps in the empirical data on impacts and values and recognition of the limits in using the evidence base to inform practical decision-making. They cite the case of the UK NEA as an example of a study where valuation of ESS partly in the framework of a cost-benefit assessment has influenced policy but they also note that not all policies relating to environmental management are so affected by data on costs and benefits. Examples they cite include the UK Entry Level Stewardship (ELS) scheme, which offers a flat-rate payment to all farmers, irrespective of their location. Such schemes fail to target payments on 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 schemes are those under Pillar Two of the EU Common

Agricultural Policy where substantial payments are made without reference to the level of the net benefits they provide.[29]

74. Overall one might conclude therefore that ESS valuation and benefit cost analysis is widespread in environmental policy circles. Yet, as this chapter has shown, examples of actual use of such analysis are relatively few and cases where they have influenced policy are even fewer. A similar observation was made some time ago by Adamowicz (2004) who noted that environmental valuation is not as widely used in policy analysis as it could or should be. He speculated the reasons may be concern about the methods or a failure to communicate the findings and focus them more on policy applications. He also argued that there was a need to develop better methods of benefit transfer to facilitate application of ESS values in a wider number of cases. While the situation has improved somewhat since his article the broad sentiment is still true.

75. In summary the picture is mixed. ESS values have entered the mainstream of thinking about the environment and, in some cases, they have influenced policy. Yet the number of cases one can quote where policy has firmly been influenced by such thinking are few and that is even more so if focusing only on the use of cost-benefit methods. There is still much to be done to apply the methods more widely, including work on improving them, but also making them more easy to use.

5. Conclusions

76. There are now a large number of valuation studies on the benefits of biodiversity and on the services provided by different ecosystems (ESS). Both ideas have been used to elicit values from nature but in recent years the research community has focussed on ESS as the main organizing framework, with some additional use of the biodiversity concept to value entities that have intrinsic value and are of an extraordinary nature.

77. Estimates are available for the services from most habitats, by type of ecosystem service, usually expressed in USD per hectare per year. Coverage varies by habitat and region, as does the quality of the assessment, but it is possible now to carry out an estimation of changes in values for a number of ecosystem services a result of the introduction of a new policy or of a physical investment that modifies the ecosystem. While this is a positive development, there remain some issues to be resolved. One is the possibility of double-counting of services when using the standard categories of provisioning, regulating/supporting and cultural ESS. Regulating and supporting services are the basis of the provisioning services and so values estimates for the two cannot always be added up. Yet, there are cases where the estimate made under regulation has no counterpart under provisioning (e.g. climate regulation is an example), and in such cases the two estimates are complementary. Care is needed when using data obtained from studies for different categories of ESS to obtain a total value.

78. The literature reveals wide ranges of values for services by habitat, indicating that spatial variability is of great importance. This factor has been underscored in the recent research that values ESS on a spatial basis, such as the global assessments conducted as part of the TEEB and the NEA in the UK. It is also noted that biodiversity as such is not well integrated in the ESS framework, although some attempts have been made to include it in valuing habitats changes using the concept of Mean Species Abundance.

79. A few studies at the global and national or regional level have shown how benefits and costs of alternative development options can be compared within an ESS framework. The TEEB study of Hussain et al. (2011) and the economic basis of the UK NEA (Bateman et al., 2013) are both examples of these.

29. Schemes of this kind are examples of the class of Payments for Environmental Services (PES). For more details on such schemes, the reader is referred to OECD (2010).

Neither are comprehensive in their coverage of ESS (biodiversity-related and marine ESS are conspicuous in their absence) and there are significant uncertainties in the figures obtained but they are, nevertheless of enough value to provide guidance on the net benefits of different policy options. There are also other studies at the national level that have used ESS in a partial cost-benefit framework and these are described in the report.

80. At the local level, the relatively few studies that actually conduct a full blown cost-benefit analysis relative to the number of studies that estimate the value of ESS is noted. One reason is the need to estimate changes in ESS as a function of a policy change rather than value the ESS in their current form and difficulties in linking policy changes to changes in ESS.

81. A number of other points emerge from the review. One relates to the role of benefit transfer compared to primary data collection. For policy assessment and cost-benefit analysis primary data collection seems to be required quite often. This may reflect the literature surveyed, which was from leading journals in the field, but it also reflects the fact that changes in ESS are not so easily valued on the basis of the existing literature.

82. Other key points noted relate to the methods of elicitation, the range of ESS coverage, the importance of sensitivity analysis and of evaluating the distribution of costs and benefits and well as of their total value. On methods, there is still some scepticism on the use of stated preferences or indirect valuation methods compared to methods that rely or market information. However, the evidence shows that non-market methods can give reliable estimates of values and for some categories of value such as non-use they are the only method available. Given the uncertainty about many of the key parameters, sensitivity analysis has to be part of the toolkit of cost-benefit analysis when ESS are involved. Lastly, in many situations where the appraisal involves comparing gains or losses from ESS against other costs and benefits, there are important distributional considerations. These have to be taken into account in making decisions where the trade-offs between conservation or development at the heart of the debate.

83. The acceptability and use of cost-benefit methods for policy purposes is still relatively limited. Commentators have noted the increasing sensitisation of policy makers to economic values of ESS, which is encouraging, but examples where policies have been influenced by a formal analysis of benefits and costs are still very few. One can expect an increased use of these methods as governments become more convinced of their credibility and begin to see how they can help in making decisions of greater benefit to society.

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