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Viewpoint

Precisely incorrect? Monetising the value of ecosystem services[☆]

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ABSTRACT

Environmental scientists employ political and economic arguments to argue for the conservation of biodiversity and the maintenance of ecosystem services. However, the economic terminology has a number of connotations which makes its usefulness for the intended effect questionable.

On the one hand, the basic assumptions underlying economic valuation are far from realistic and represent rather a caricature of human behaviour. On the other hand, the methods based on these assumptions are manifold and lead to wildly diverging results. Thus the calculated value of ecosystems and their services is not a robust figure, but varies with the valuation method applied (plus a plethora of subjective assumptions). As a result, it is not possible to 'objectively' calculate the value of ecosystem services. Fortunately, it is also not necessary to do so. Given the inherent flaws of the valuation process, it seems more promising for biodiversity and its conservation to restrict the economic calculus to the role of a contribution in the implementation process for a set of politically defined targets, rather than using it as the target setting mechanism itself.

The paper lists some of the core assumptions, presents a systematic overview of the most relevant valuation methods, illustrates them by providing examples and discusses their limitations. As an alternative, political target setting is suggested, based on a multi-stakeholder, multi-criteria analysis. Market prices play a role in this analysis, as one factor amongst others. For the implementation, cost-effectiveness analysis gives important hints, and economic instruments – inter alia – can play an important role as enforcement mechanisms. However, incentives should be based on criteria of (potential) effectiveness, not on value calculations.

1. Introduction: science in society

Science has never been a purely curiosity driven undertaking; at all times scientists wanted to see their insights applied, from Aristotle via Galileo and Nobel to the WMO and IPCC. Since the uncritical belief in scientists has given room to a more balanced approach in the late 1960s, scientists have used other than scientific terminology and arguments to make their case heard. This applies to the promotion of nuclear energy by the scientific community, of genetic engineering, nanotechnology, and of biodiversity conservation: in all cases natural scientists have employed political, social and economic arguments to support their case. Talking about ecosystem services instead of ecosystem functions follows this tradition: it helps to convey the message to decision makers in a terminology they are used to and which might stimulate them to act, scientists hope (see e.g. Reid et al., 2005).

Feeling that biodiversity is valuable (in whatever sense) these scientists have taken resort to the economic definition of value to make their point understood and to provoke policy resonance. Thus schemes for 'Payment for Ecosystem Services' PES – based on a 'beneficiary pays' rather than the 'polluter pays' principle – have been suggested. Properly understood, they are not conceived as a 'silver bullet' but tailored to address a specific set of problems: those in which ecosystems are mismanaged because many of their benefits are externalities (i.e. they accrue to external persons not paying for them) from the perspective of ecosystem managers (Engel et al., 2008). The benefits can be local, but also global, and their value is usually expressed in monetary terms.

However, what does this imply? Are economists pursuing the same goal as ecologists when entering into valuation? Does the economic objective (ecosystem resources should be allocated to those uses that yield the highest gain to society, as measured through valuation in terms of benefits of each use, adjusted by its costs) lead to the same results as pursuing ecological objectives?

The reminder of this paper is structured as follows: Section 2 discusses the implications of the change in terminology when

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Table 1
A compilation of meanings of the word 'value'.

Market value	The exchange value or price of a commodity in the open market.
Intrinsic value	The value of entities that may have little or no market value, but have use value.
Intrinsic, non-use	The value attached to the environment and life forms for their own sake.
Existence value	The value attached to the knowledge that species, natural environments and other ecosystem services exist, even if the individual does not contemplate ever making active use of them.
Bequest/vicarious values	A willingness to pay to preserve the environment for the benefit of other people, intra and intergenerationally.
Present value	The value today of a future asset, discounted to the present.
Option value	A willingness to pay a certain sum today for the future use of an asset.
Quasi option value	The value of preserving options for future use assuming an expectation of increasing knowledge about the functioning of the natural environment.

Source: Kumar and Kumar (2008, p. 809), there referred to as adapted from Gilipin (2000).

'talking economics'; Section 3 provides a brief introduction to the basic concepts behind economic valuation and Section 4 presents the methods applied in economic valuation, and their results. The risks and opportunities of valuation are discussed in Section 5; Section 6 concludes.

2. Method change: from traits to functions to services to valuation

The step from ecosystem traits to functions is one from a descriptive approach listing observed facts and figures to analytical science, characterising any ecosystem phenomenon contributing to something else (i.e. almost everything) as an ecosystem function. This step provides new insights on the interaction of the observed traits, but also risks some loss of information by introducing structures and hierarchies into the observation data.

2.1. Services

The next step, from functions to services, is one from scientific analysis to subjective selection and classification: ecosystem functions are not distinct but mutually defining and interdependent—while services are so by definition. Ecosystem services are (again by definition) immaterial, and ecosystem goods are taken into account by describing the process of providing them as the ecosystem service. Functions are services if and only if they affect human needs or values (de Groot et al., 2002). Thus the MA defines ecosystem services as benefits people can obtain from ecosystems, distinguishing provisioning, regulating, supporting and cultural services (MA, 2005). Bonnedahl and Eriksson (2007, p. 101) point to the inherent risk of this transition: "ecosystem services are a perspective that certainly highlights the importance and degradation of the systems, but the raison d'être of the ecosystems is to serve humans. Thus, ecosystems are, in principle, exchangeable, and the perspective appears open for negotiation, if human needs would call for higher harvests."

Thus the ecosystem service approach introduces externally set definitions into the scientific discourse regarding which states, structures and processes do in fact contribute to human production and consumption, and which – to the best of our current knowledge and the valuation task at hand – do not. On this basis, as Nunes and van den Bergh (2001) suggest, 'engineering models' could be developed, including the socially relevant ecosystem services, but still in concept, structure and terminology reflecting the approach of scientific analysis. In a final step, the outcomes of these models would be translated into the language of interests, concepts and perceptions of lay respondents, based on their subjective, pre-theoretic patterns of perception and valuation of the phenomena at stake.

2.2. Valuation

In the next step, valuation, two basic concepts currently prevail: (a) ecological valuation based on bio-physical accounting (e.g. using an energy theory of value like Costanza, 1980, or economy–ecology models and satellite accounts) most often with total neglect of human needs and/or wants, and (b) economic valuation based upon consumer preferences out of context of system characteristics, i.e. with neglect of limits (Winkler, 2006). Both represent specific concepts of value needing closer scrutiny (for a list of relevant meanings of 'value' see Table 1; other notions of value, such as nutritional value or moral value are not taken into account here). Both tend to fall short of suitable methods in an (inherently normative) sustainability framework, as sustainable development is based on two essential principles: prioritising human needs, in particular those of the poor, and respecting environmental limits (WCED, 1987).

Both concepts of valuation are based upon an agreement in a human society; they are not necessarily anthropocentric, although they are definitely anthropogenic. As is evident from the description above, in the economic valuation approach subjective interpretations play a decisive role in establishing what to measure as the value of an ecosystem and its services (Fig. 1).

Objects of analysis	System traits	Functions	Services
Methods	descriptive	analytical	subjective, preference based
Results change with	new insights (what is worth describing) new observations	new observations new methods new models of explanation	beneficiaries, their habits, fashions, attitudes, education and income

Fig. 1. From ecosystem traits to services, via ecosystem functions. *Source*: authors.

The political argumentation (ecosystem functions as a basis for survival and development) has not been extremely successful in the communication to decision makers and the lay public, and it left hardly any footprint on the inner-science discourses. Opposed to that, the economic argumentation (ecosystem services are valuable) has had severe impacts, but more on science itself than on decision makers—with a remarkable exception in the field of climate: the Stern Report.

2.3. Money talks

(Environmental) Scientists these days use the economic terminology, talk about services, productivity, (natural) capital stocks, efficiency, etc., in order to get their science-based messages across to decision makers. More often than not, when talking about the value of ecosystems and their services, they understand this as a metaphor, usable to communicate science-based insights. Norgaard (2009) warns that the step from value as a metaphor to economic pricing as key valuation method leads to a neglect of other ways of understanding ecosystems developed in ecosystem science, and thus obscures current and potential future knowledge. Kosoy and Corbera (2009) point to the loss of holistic understanding of ecosystems enforced by the definition of discrete services. On the other side, however, economists such as Pavan Sukhdev, coordinator of the TEEB report, the international study assessing the value of biodiversity, consider valuation a key means for conserving biodiversity. In the introduction to the interim report he claims that the "lack of valuation is, we are discovering, an underlying cause for the observed degradation of ecosystems and the loss of biodiversity." Like many other economists he is convinced that valuation can make "a comprehensive and compelling economic case for conservation" (European Commission, 2008, p. 4). Here value is no longer a metaphor but has been turned into an economic figure, on par with other value, price and cost calculations.[1]

Following very similar reasoning, IUCN's Sriyanie Miththapala (2008) argues that the importance of ecosystems and their services are widely unaccounted for, leading to wrong policy priorities. She gives the example that tourism, based on marine biodiversity and coastal ecosystem services, provides the basis for the Maldives economy, accounting for 20% of the GDP and 40% of the employment. Including associated services, the sector generates 74% of the GDP, 60% of foreign exchange earnings, and 90% of the government revenues. As based on macroeconomic data about the tourism sector, it is easy to understand how these figures have been derived.

However, she also points out that ecosystem services are often the basis for household production, largely unaccounted for in macroeconomic accounting, in particular in subsistence economies. Mangrove forests, she argues, are valuable, with traditional use contributing US$ 3000/ha*yr (50% of the household income of the poorest) to local income in parts of Indonesia, and US$ 1300/ha*yr to inshore fisheries income on the Belutchistan coast of Pakistan (plus another US$ 900/ha*yr to offshore commercial fishing) (Miththapala, 2008). The intention is again clear and honourable, but how are such figures derived? What do they really imply? Where are the border lines between unquantifiable intrinsic, qualitative inherent and quantitative use values (monetary valuation is generally accepted to be not feasible for environmental goods and services with a religious or spiritual value)? How plausible are the assumptions which the valuation is based upon? Natural scientists (as well as policy makers and other 'lay' groups) are most often either not familiar with or do not reflect the connotations of the underlying valuation methods and thus the implicit implications of their argumentation.

[1] However even the TEEB interim report (European Commission, 2008) recognises that the importance (the original meaning of 'value') of environmental services such as water, non-timber forest products or medical plants for the livelihood of poor people in India and elsewhere cannot be properly measured in monetary terms; not even calculating the 'GDP of the poor' (European Commission, 2008) solves the problem. The final report will be presented to 10th Conference of Parties of the CBD in Nagoya, Japan, October 2010.

3. The economics behind valuation

3.1. The basics of economic valuation

In standard economics, the value of every good is defined to be nothing else than its price. The price is not intrinsic to any good but exclusively the result of exchange processes between homini economici in the market. This implies that where there is no market, there is no price and thus there is no economic value. Economic men (an economic woman has never been defined) exchange goods based on their different preferences, always totally determined by the imperative to maximise their personal utility. Transactions are possible once all traders have found a counterpart whose offer is representing the same utility to themselves as to the partner. Then the equilibrium is reached, supply and demand have met at the equilibrium price, and the transactions proceed immediately (real time and space play no role). This situation represents the maximum welfare, as all needs are satisfied, a situation emerging automatically in undisturbed markets. Such ideal competitive markets (without state intervention) lead to prices at which every supply finds a matching demand and every demand – expressed in purchasing power terms – is satisfied by an equivalent supply. Market clearance is always given.

For almost every good and service there is a functionally equivalent substitute, by definition (if not yet, rising prices will certainly stimulate technological development to provide one: economics is based on optimism as much as religion is based on belief, see e.g. Hussen, 1999; Costanza et al., 1998). However, in economics parlance, 'functionally equivalent' does not refer to a multitude of characteristics, but to just one: the contribution of the respective good to utility generation. As the utility is defined one-dimensionally (as a scalar, but not measurable), the different elements can simply be added up to give the whole: all elements are distinct and independent. This permits the separate valuation of individual services (but requires neglecting the multidimensional attributes of ecosystems). For instance, the utilities from water availability, climate protection, food and leisure use provided by a forest are commensurable and can be added up to total utility. In this view, man-made goods can indeed replace natural or social goods, targeting one service after the other.

3.2. Discounting the future

For the use in Cost–Benefit-Assessments CBA (a necessarily subjective measurement, and cornerstone of normative welfare economics, see Folmer et al., 1995) and subsequently in decision making processes, a model to calculate the current value of future gains and damages like those expected from biodiversity conservation, or the future gains from exploiting a resource, is needed. Here it is: provided with all information about current and future prices and market volumes, homo economicus – as a death-prone individual – will do nothing but maximise his utility by exploiting and exhausting his resource during his lifetime (there is no utility from leaving something behind). He will value the resource by comparing the surplus generated with what he would have earned by selling it immediately and putting the income into a bank to generate interests. For planning the optimal utility maximising strategy, since demand and thus prices depend on preferences, future preferences must also be known to homini economici—preferences are assumed to be inherently constant, changeable only by external influences (for a critical assessment of these assumptions see e.g. Söderbaum, 2000). Then the future gains are calculated based on the future price times the volume that can be sold. To determine the current value, the result is depreciated, applying a discount rate close to the market interest rate on long term bonds or savings, usually about 4% for public projects (Folmer et al., 1995). This way, even a high gain or damage in the distant future becomes irrelevant as compared to a lower but sooner profit or loss, constituting an inherent bias against all long term, investment dependent conservation programs (this is one key reason why also the TEEB fiercely criticises discounting the value of ESS and biodiversity). Table 5 provides an example how this works out in damage cost assessments.

3.3. Dealing with complexity

If prices were to reflect all dimensions of value (in the common, non-economic sense of the word), they would be due to change with location, over time and with the social groups involved (Folmer et al., 1995; Costanza et al., 1998). On the one hand, this would create economic problems (some phenomena of this kind exist in a globalised economy, providing a basis for speculation and real-time trading), and on the other this would render economic valuation rather worthless, as no comparable prices for multiple services over different time horizons would emerge. Thus the above assumptions are indispensable for economic valuations to be used in CBA, as economists openly admit: "The monetisation of environmental impacts can smoothen the problem of multidimensionality and thus provide a decision support in cost-benefit analyses" (Schägner, 2008, p. 25, authors' translation).

True, some of these assumptions have been relaxed in different branches of economics. For instance, experimental and behavioural economics have developed concepts of bounded rationality and dealing with incomplete information. For instance, Krugman explored the role of real space for economics, and Stiglitz the role of incomplete information—both these deviations from the standard model have been honoured by Nobel Prices in economics.

Nonetheless, these insights are still treated as modifications of the ideal model for determining prices and thus values. Deviations from the model (e.g. reality) are dealt with by on-top modifications of the approach, leaving the basic equilibrium model untouched and have thus not led to new, broadly applied calculation methods (Keen, 2001). In most valuation studies they play no role at all. In particular, the public character of ecosystem services, meaning that a variety of services accrues to society as a whole, beyond the individual utility gain, escapes economic valuation. The axiomatic approach of methodological individualism indispensable to neoclassical economics (Arnsperger and Varoufakis, 2006) makes it impossible to value those services (as all emergent properties of higher system levels): the assumption of fixed individual preferences and individual utility maximisation precludes any deeper understanding of human decision making. It disregards the diversity of human motives, values and their linkages to environment and ecosystems (for a detailed psychoanalytical discussion see Kumar and Kumar, 2008). Nonetheless, homo economicus is the basis for economic rationality, efficient markets and prices emerging from the equilibrium of supply and demand functions. Economists need a market to be able to determine a price and the value of a good (Costanza et al., 1998).

4. Economic valuation: methods and results

Most scholars are aware that the common people's perception of and relation to the value of ecosystems is quite different to what is conceptualised in economics: for them, the natural environment has a value beyond its immediate utility, i.e. beyond the "abstruse quantification and reductionism of economics" (Kumar and Kumar, 2008, p. 814).

4.1. Real markets, real cost

That does not rule out using the economic calculus when appropriate, i.e. when money values apply as in the economic assessment of biofuel strategies (e.g. Spangenberg and Settele, 2009). This is always the case when ecosystem services or their outputs are traded on markets. However, some attention should be paid to semantics: The cost incurred by combating an invasive species represents not the (negative) value of the species as such, but of the utility of having an area free of it. Similarly, economic cost incurred due to the loss of pollinators give neither the value of the species involved nor the value of pollination, but the value of goods to which pollination (alongside other factors like seed, soil and water) contributes. An example is the valuation of "pollination services, which recently have become threatened by honeybee colony collapse disorder [They] contribute to fruit, nut, and vegetable production worth $75 billion in 2007" (Swinton et al., 2007, p. 246, quoting figures from USDA, 2007; see also Klein et al., 2007). The global value of pollination dependent crops – and thus the value for direct human consumption which pollination services co-produce – has been identified as being 153 billion € in 2005 (based on world market prices and FAO statistics; Gallai et al., 2008).

As these examples illustrate, valuing ecosystem services are easiest but not unproblematic if they are traded in markets—then the official market price describes the value of the service. However, it is still a conceptual challenge to allocate values to single services in case the good traded on markets is co-produced by several services, in particular if each of them is essential for the result.

Swinton et al. (2007, p. 248) highlight that "food, fiber and fuel have market prices that provide both incentives to produce those ES [ecosystem services] as well as measures of their value to society. But many other ES lack markets." In these cases, Payment for Ecosystem Service PES schemes have been introduced to translate non-market environmental services into financial assets traded on markets. If for instance the carbon fixing by forests is made a tradable good, the market establishes a price and thus a value for this service. Customers would then pay for the non-use of a forest, thus preventing it from being felled (forest destruction makes up for about a fifth of global CO_2 emissions). That is the core of the REDD initiative, one of the few substantial results of the 2009 Copenhagen Climate Summit. It can be effective—provided the market price for non-emission certificates is higher than the market value of the timber felling the same forest would provide (non-timber services provided by the same forest are not taken into account). In the media, a global minimum of US $10 billion from non-use certificates has been discussed as necessary to compensate for the profits from felling (Bethge et al., 2008), and it remains an open question whether the global certificate market would yield these sums in a reliable and permanent fashion (otherwise price fluctuations or speculations could tip the balance against forest protection).

When applied to whole ecosystems and all their services, valuation is more difficult as it has to deal with the complexity of such systems. For instance, "neighbouring ecosystems provide food, refugia, and reproductive habitat for pollinators and biocontrol agents; they provide wildlife habitat; and they help to attenuate some of the unwelcome effects of agricultural production, including the escape of nitrogen, phosphorus, and pesticides into non-agricultural ecosystems where they may produce undesirable impacts" (Swinton et al., 2007, p. 248; see

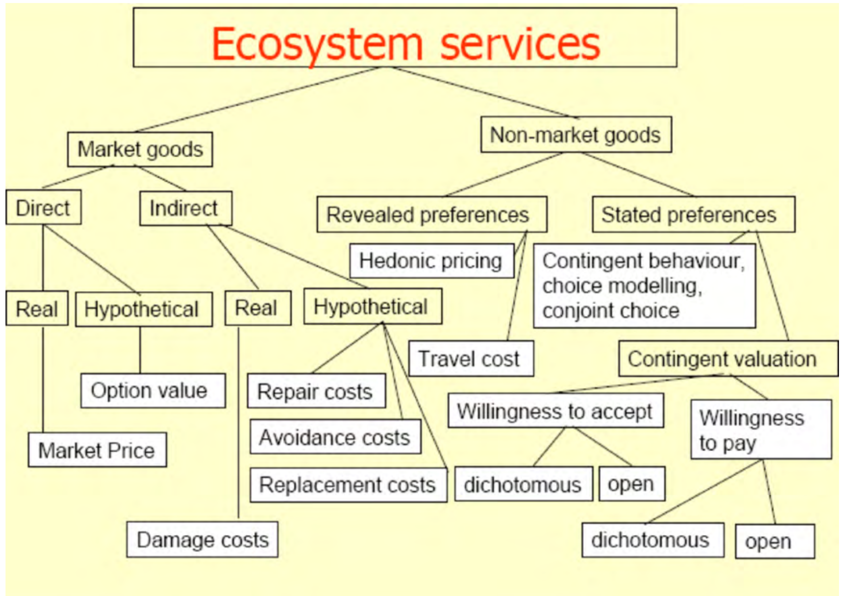

Fig. 2. Clustering economic valuation approaches according to objects and methods.
Source: Compilation by authors.

also Klein et al., 2007; for biocontrol services e.g. Way and Heong, 1994).

The calculation of market values can also be indirect, based on the damages caused by a disservice[2] (like damage costs of biological invasions) or the costs avoided by a certain service. In this view, the value of any system is given by the price of the services it provides, expressed as avoided damage cost. For instance, the value of the UV-B radiation protection service provided by the ozone layer expressed as avoided damage is significant (sum up all potential agricultural losses and the cost of medical treatment), but it was zero before the protective function was discovered: there was no demand for this service, as the consumers did not know about it, and thus no market, no price and no economic value. Based on avoided damage cost calculations, Miththapala (2008) quantifies the (annual) value of coral reefs due to their coastal protection service at several US\$ 100,000 km^{-1} in Indonesia and at nearly US\$ 1,000,000 km^{-1} in the Philippines, and of coastal wetlands, providing flood protection and water purification services at US\$ 2500 ha^{-1} for Sri Lanka.

4.2. Real markets, hypothetical cost[3]

It also makes some sense to assess hypothetical economic expenditures, e.g. for management or repair costs to re-establish the status quo ante. Even more speculative are estimates of the future costs of substituting for specific (ecosystem) services (replacement cost such as hand pollination to compensate for pollinator loss or fertilisers to compensate soil fertility decline), as they have to be based on currently known technologies and their prices, and they have to deal with economic discounting. Extremely speculative is the estimation of avoidance cost or averting costs (what would have had to be invested in the past to avoid a current damage: a figure to be compared with the actual damage cost). Their calculation can be based on production function assessments (assuming steady functions). This requires to find the second-best production technology for each function, and to calculate the opportunity cost of switching behaviour, both for each service separately, asking ‘which utilities have I missed by investing time and money here and not at another, e.g. the optimal location?’.

In all these cases, what is calculated is the hypothetical cost of sustaining an economic service provided by ecosystems, and this is taken to be equivalent to the value of the ecosystem services under analysis (with different services analytically treated as independent but commensurable, Folmer et al., 1995). However, that the assumption of commensurability is counterfactual has long been discussed in the ecological economics literature (O'Neill, 1997a; Martinez-Alier et al., 1998; Spangenberg, 2005). Formulating such hypotheses is not unfamiliar for natural scientists, and it can be a valuable step in the research process, but in economics there is usually no way to empirically test them: the hypothesis itself is the result (Fig. 2).

However, determining this kind of costs and thus values faces some informational conditions which are rarely met. Nunes and van den Bergh (2001) elaborate them for the replacement costs (they apply to repair cost as well, and indirectly to avoidance cost):

1. the replacement system must provide functions that are qualitatively and quantitatively equivalent to the original ecosystem (neoclassical, resource and environmental economists systematically neglect this condition in their one-by-one analysis of the ecosystem functions or services affected);
2. the investigated replacement is the least cost option of all possible replacements (otherwise the value of the ecosystem service to be replaced would be overestimated),
3. the aggregated willingness to pay for the replacement exceeds the costs of the replacement in face of the loss of the original ecosystem functions (otherwise replacement would imply a welfare loss).

Beyond the need for speculative assumptions, the information demand from condition 2 and 3, and the “lack of low cost measurability and valuation methods currently precludes efficient allocation of many ecosystem services through market-based approaches” (Kroeger and Casey, 2007, p. 321).

[2] Even ecosystem functions can establish disservices: dead wood needed for functions limits stem yields, and leaving residues in the forest to sustain soil functions reduces pulp and paper yield.

[3] In economics real prices in real markets are usually called ‘market prices’, whereas all hypothetic prices (in real and hypothetical markets) are in sum addressed as ‘shadow prices’. They are considered to “reflect all of the indirect consequences of an option” (Folmer et al., 1995, p. 79).

4.3. Hypothetical markets, hypothetical cost

Ecosystem services are rather frequently non-market goods (public option value goods, insurance goods, merit goods, etc.) for which neither directly nor indirectly real or hypothetical market prices can be determined. In this case, economists derive price and value calculations either from revealed preferences or from stated preferences. We will illustrate the most important valuation methods by examples taken from the 2008 volumes of Ecological Economics, a leading journal in the field, one example for each approach to illustrate each method described.

4.3.1. Revealed preferences

Revealed preferences are based on indirect calculations, deriving value figures from the effects of behavioural change associated with the service (or the lack of it) in real markets. They comprise non-use values (existence values) like knowing about the existence of a deer population in the region; non-consumptive use values (watching them) and consumptive use values (hunting them). The two main assessment methods are hedonic pricing and travel cost estimates.

4.3.1.1. Travel cost. It is mainly applicable to leisure and holiday activities where travelling is voluntary. In these cases, as the homo economicus is always maximising his utility, he will only be travelling to a certain place if the stay there provides more utility then saving the cost and abstaining from the visit (the gains and pains of travelling do not count, as real space and time do not). Then the travel cost is a stand-in for the value of what has been enjoyed at the destination. For instance, Knoche and Lupi (2007) calculate the value of the white-tailed deer (*Odocoileus virginianus*) by assessing the demand for deer hunting via the hunters' travel costs. As a result, the value of 10,000 deer more per county is the result of additional travel expenditures of US$3.94 per hunting trip for firearm hunters, and of US$1.75 per trip for archery hunters.

4.3.1.2. Hedonic pricing. "Hedonic valuations use relationships between land property prices and property characteristics to value changes in the characteristics." (Swinton et al., 2007, p. 248). They go from the assumption that services/disservices like improved or diminished environmental quality change the willingness to pay for a good associated with them, and this is reflected in the market price (assuming full knowledge and perfect markets), in particular in the housing market. The price change is then a measure of the value of the ecosystem services enjoyed, like a price increase due to the establishment of a nature reserve in the neighbourhood. However, empirical work comparing the changes in individual well-being caused by pollution to housing prices have shown that they do not necessarily reflect the local environmental quality changes (Rehdanz and Maddison, 2008).

4.3.2. Stated preferences

The alternative to revealed is stated preferences. In this methodological approach (probably the most frequently used one, at least in the economics literature), hypothetical markets are introduced and interview partners have to define a price, in different ways, for the respective ecosystem service within these markets.

According to Barkman et al. (2008, p. 51), "qualitative investigations of the pre-theoretic concepts, beliefs and values that non-expert respondents are likely to bring to the valuation task are a standard requirement for any lege artis empirical stated preference study." Then ecosystem services are described in terms of a benefit from ecosystems the respondents really care for (an assessment depending on the social group, the respective culture, attitudes of the researcher and thus due to change over time). The total welfare would then be derived by multiplying the gains calculated per capita with the size of the respective group. "Ecosystem functions for which the analyst finds no such benefit are not considered services and excluded from the valuation" (Barkman et al., 2008). For the services identified, preferences are asked for, most often in monetary terms. As usual with questionnaires, the responses are not taken at face value, but again processed and interpreted.

Table 2
A choice modelling example (figures from Brey et al., 2007).

Service	Value/welfare gain [€/yr, continuously]	Respondent group
Sequestering 68,000 t of CO_2	11.79	Catalonia inhabitants
Delaying the loss of land productivity for 10 years	0.12	Catalonia inhabitants
Picnicking in the forest	6.33	Picnic users
Picking mushrooms in the forest	12.82	Rural residents
Four wheel driving in the forests	−9.67	Catalonia inhabitants

4.3.3. Choice modelling

Questions can be asked in different ways: Choice modelling offers different set of alternatives (choice sets), in which one of the parameters is a price. People's choices then indicate which price they consider adequate in the context described (a multi-alternative regression analysis is needed).

For instance, Brey et al. (2007) used this methodology to estimate the value of aforestation areas in Catalonia, North-Eastern Spain (Table 2). Different potential user groups were identified and their answers transformed into monetary values for different ecosystem services. It is obvious that the choice of the respondent group has a significant impact on the results, e.g. rural and city population (Catalonia is an urbanised region dominated by its capital Barcelona) hold different value perceptions for the same ecosystem, as due to their different use patterns they enjoy different services.

4.3.3.1. Contingent valuation: willingness to pay WTP, willingness to accept WTA. The second kind of stated preference methods, probably the most frequently used one, is contingent valuation and essentially consists of presenting the target groups with one alternative (one variable changing) and either asking them for their willingness to pay WTP for getting or avoiding such a parameter change, or their willingness to accept WTA compensation for a damage or a foregone improvement. As both approaches address the same issues with the same group, in theory (i.e. assuming rational, utility maximising behaviour) their results should be identical. In reality, however, WTA figures tend to be significantly higher than the much more frequently used WTP figures: humans are loss-averse, to most people 1 € loss counts more than 1 € gain.

One example of WTP analysis is the study of Tseng and Chen (2008). They measured the value of climate change damage to Taiwan trout (*Onocorhynchus masou formosanus*) depending on the degree of temperature change (Table 3). The example is interesting

Table 3
A willingness to pay example (figures from Tseng and Chen, 2008).

ΔT [°C]	Population left [# individuals]	WTP [US$/cap·yr]
0.0	1612	0
0.9	740	16.22
1.8	560	25.72
2.7	146	33.60

as is shows the effects of scarcity: the less trout is left, the more its protection is valued. However, such figures also indicate some of the limits to the economic calculus: If certain measures need to be taken to safeguard the survival of the species, it may be a lethal failure to wait until the number of surviving individuals has shrunk enough to generate a WTP which in turn would justify to take preventive action without reducing the total welfare in an economic sense (see the argumentation on replacement costs above).

In general, WTA and WTP can come in two versions, a dichotomous and an open analysis. In the dichotomous case, the respondents receive the description of an event/a state change and are offered a payment, or are asked to pay. They can only accept or refuse the offer. A series of such decisions is then processed to estimate the final WTP figure. In an open analysis, the respondents are free to state what maximum figure they would be willing to pay (e.g. Wätzold et al., 2008); the results are processed to eliminate 'unrealistic' figures. As both WTP methods claim to measure the same thing, the result, i.e. the value calculated for a specific ecosystem service, should be identical. However, this is not the case, as a case study from the Yaqui river in Mexico illustrates (restoring the water flows in a river which has not reached the sea since more than half a century due to water abstraction for irrigation, Ojeda et al., 2008). The ecosystem services explored were

- maintenance of riparian vegetation, wetlands, and estuaries (limited remains after 56 years of no water flow);
- protection of habitats for birds and other fauna (dto.);
- maintenance of local marine fisheries (catch was decreasing due to the lack of nutrient supply);
- dilution of pollutants (there is no waste water treatment plant in the region);
- recreation use;
- immaterial: existence, cultural, and option value; and
- use value for future generations.

For all these ecosystem services together the average WTP found with the dichotomous questionnaire was US$4.70/month* household, whereas with the open questionnaire it was US$6.60/ month*household. The context dependency of such figures is most obvious in the case of the 'dilution service': building a waste water treatment plant would most probably reduce it to zero. When processing the responses, about 20% had been excluded from the value assessment, as either the participants had given 'unrealistically high' WTP figures, answered with a 'protest zero' ("the polluter or the government should pay, not me"), or were respondents regarding which the interviewer lacked confidence in the sincerity of the respondents answers.

It again becomes obvious that (a) a highly subjective assessment is needed before calculating value figures, and (b) respondents are reduced to the role of consumers, neglecting their role as citizens, with partly diverging preferences as demonstrated by Sagoff (1990,1998). Thus, the first two groups excluded gave a more political statement than an economic one, and are therefore rightly eliminated from the economic valuation. But this comes at a cost: the two groups eliminated tend to be the most environmentally sensitised ones, and their exclusion most probably results in a downgrading of the calculated value of the ecosystem services under analysis.

Sometimes the spread of results is too broad to be justifiably described by simple aggregates (average, mean, and median values). In this case ranges are documented, which often is still good enough to identify the preferable option in general terms. For instance, Sandhu et al. (2008) have done so for comparing the value of ecosystem services provided by organic and conventional agriculture in New Zealand, in the Canterbury district (Table 4). They find that both provide services of non-market value, higher – as might have been expected – for organic agriculture. This contributes to, but does not fully explain the higher Total Economic Value TEV[4] generated by organic agriculture.

Table 4
A total economic value calculation example (figures from Sandhu et al., 2008).

	TEV [US$/ha*yr]	Thereof: non-market value [US$/ha*yr]
Conventional	1270–14,570	50–1240
Organic	1610–19,420	460–5240

4.4. Unvalued valuables

Two more methodological caveats need to be mentioned:

(i) First, as it is the service, not the stock as such which is valued, all those elements of stocks which are not essential for producing the respective service can be given up without a change in utility provision and thus in value. In other words: economic valuation addresses the economically relevant part of biodiversity—the rest is superfluous from an economic point of view. What has been valued is compared to other utilities and may be protected or shed, depending on what creates more cash income. As a result, the economic optimum (maximum utility generated by protecting as much biodiversity as people would be willing to pay for, realised for instance by including the 'price of ecosystem services' in the market transactions: internalisation of external costs) may well represent the economically optimal destruction of biodiversity. It may even call for such devastation, by demanding to substitute non utility providing elements of ecosystems for 'more productive' ones.

(ii) Secondly, as a certain service is valued in isolation, the method is of limited applicability in more complex approaches such as ecosystem management. Taking for instance the case of water management, with water supply, water purification and flood regulation representing perhaps the most precious ecosystem services to mankind, the deficit becomes obvious. In the ecosystem management approach, water management is not only about the physical environment (which could possibly be treated as a number of separate commodities), but is "concerned with ensuring people achieve their needs through socially fair, wise economic, and ecosystem friendly processes". The means to do so is the "integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way" (Guerrero, 2008, p. 2), as practiced in ancient water dependant societies like the Ifugao, the architects of the magnificent irrigated rice terraces of Northern Luzon, Philippines (Conklin, 1980; Settele and Martin, 1998). Although economists claim their methods to be promoting conservation and sustainable use, economic instruments do not address needs (unless they are expressed as demand in the market, backed by purchasing power), social fairness and equity in resource use. Social benefits can be side effects of the management process, but whereas they do count in integrated assessments (IA) and ecosystem management, they do not so in economic valuation. Thus valuing services can only play a limited role in integrated ecosystem management, and no way replace a management system incorporating social concerns and challenges.

[4] As economic thinking knows no value of ecosystems as such, the Total Economic Value TEV is derived based on valuing the individual services flowing from them, treating them as discrete objects.

5. Discussion: pros and cons of economic valuation

"A formal expression can be extremely precise without being at all a precise representation of the underlying concept to be captured. In fact, if that underlying concept is ambiguous, then the demands for precise representation call for capturing that ambiguity rather than replacing it by some different idea—precise in form but imprecise in representing what is to be represented." Sen, 1989, p. 317.

5.1. The risks

The risk of valuation is to get the figures wrong, and it is unavoidable. More serious, however, is the risk from valuation—that the application of economic instruments becomes an end in itself and gains primacy over the initial purpose, protecting ecosystems and their services. This leads to such proposals as the one by Köck (2008, p. 18, authors' translation): "In order not to unnecessarily restrict economic activities, it has been suggested to permit the destruction of a habitat if a certificate is presented confirming that an equivalent habitat has been created somewhere else. Making the certificates tradable would create a global market, supporting a flexible and cost-effective biodiversity protection". Ecologists may be tempted to ask 'where, and for what? And how do you create an ecosystem?' Kosoy and Corbera (2009) qualify the treatment of ecosystems and their services – which are mostly public goods and do require capital or labour to be produced – as tradable goods as 'commodity fetishism'. However, the neglect of real time and space in economics, the utility thinking, and the principle of substitutability, makes such proposals sound reasonable in an economic context and increasingly seems to form the basis for the insurance of biodiversity damages.

More generally, Schägner (2008, p. 25, authors' translation, emphasis added) argues that "regardless of the weaknesses which must be attributed to the concept of Total Economic Value TEV, it must be acknowledged that the monetisation of the environment provides opportunities to take marginal changes of the environment better and more objectively into account in cost-benefit analyses and thus in policy decisions". The claim to deliver objective data as the optimal basis for decision making is upheld despite the fact that ecosystem service valuation

- isolates single services from a systems context in order to value them; is not capable of supporting multi-objective approaches (as needed for ecosystem management),
- counts only what is currently demanded and takes the prices estimated for ecosystem services to be the value of the ecosystem,
- reflects the current knowledge and the current preferences and use structures, is bound to change with consumption and production patterns and thus dependent on settlement and leisure patterns, the location of industries and on development processes in general,
- commands a variety of methods which are all based on the same set of economic assumptions, but approach the ecosystem services from different angles, with results varying widely, dependent on the methodology choice rather than on the object under analysis.

Thus ecosystem service valuation

- does not deliver a general measurement of the value of ecosystems and their services, but context and method dependent price estimates, possibly several for the same service, based on a wide range of subjective, hypothetical and partly questionable assumptions.

Therefore some ecological economists call for careful use of economic methods, and even for giving up on monetisation of environmental goods and services (Common, 2007a; Spash and Vatn, 2006; Vatn and Bromley, 1994; O'Neill, 1997b).

There are plenty of reasons for this. Most importantly, there is no such thing as an 'objective measurement' of the value of an ecosystem and its services, i.e. a measurement which (as natural scientists tend to expect) is reproducible, independent of the respective measurement methodology and of subjective assumptions during the measurement process. The sources of uncertainty, fuzziness and subjectivity are manifold:

- On the bioscience side, ecosystem boundaries and functions are far from well-defined, but depend on choices influenced by the respective research question, disciplinary background and the like.
- In the science/economics interface, the definition of what is a service is a subjective one, influenced by external (societal debates, political interests) and intrinsic factors (preferences, axiomatic convictions, research interests). It is due to change with locality, circumstances, inhabitants, their level of affluence, other agents and the broader societal discourses.
- In the sphere of economics, the results of valuation are not robust, unambiguously calculated, clear-cut value figures (although they are often presented as such), but methodology-dependent outcomes (i.e. different methods applied to the same object of measurement result in widely diverging values), influenced by a range of subjective assumptions. As no method is applicable to all ecosystem services, there is no way of defining a methodological standard, and with the divergence of results, aggregation of valuation outcomes into a total value calculation is scientifically dubious.

Figures derived in this shaky ground are hardly solid enough to base crucial decisions vis-à-vis biodiversity management and ecosystem service maintenance upon. Nonetheless it has been frequently argued that valuation, although being far from perfect, is an improvement. However, this is not necessarily the case: "the answer to the question [.][if] a fuller but incomplete accounting is guaranteed to be nearer to the truth than the conventional accounting" "is no" (Common, 2007b, p. 239)—another risk of valuation contributing to the risks from valuation.

5.2. The opportunities

Recognising these methodological flaws, in order to improve the basis for decision making, acknowledge the public good character of ecosystem services, and capture the social value of ecosystems and their services beyond mere utilitarian aspects, consensual, multi-criteria, multi-stakeholder valuation processes as developed e.g. by Munda and collaborators (Munda et al., 1994; Munda, 2004) and advocated by Funtowicz and Ravetz (1993, 1994) have been suggested as more appropriate (Kumar and Kumar, 2008). The approach we advocate requires a change of perspective, from what we have called earlier a 'vertical MCA' to a 'horizontal' one (Spangenberg, 2001; Spangenberg and Omann, 2006). This implies that not only one value system (the economic one described here, or that of a decision maker, allocating his/her weights to the individual criteria and thus determining a hierarchy of desirability) is used. Instead of one system, resulting in a ranking of options, a kind of non-monetary CBA, the diversity of stakeholders' value systems is taken into account. Thus the purpose of the exercise is not to reach an 'optimal' solution, but creating a level playing field in terms of information access as an input into a political discourse processes. The results of such processes depend on the stance of the participants; for obvious reasons they cannot be generalised.

This approach denies the primacy of economic criteria inherent to applied valuation and takes the decision on the most desirable solution, respecting the diverging value systems of different stakeholders, out of the domain of the economic calculus: the agreeable solution is not necessarily the economically optimal one. It requires the discussion of and agreement on priorities based upon a diversity of aspects, with utilitarian interest being one amongst others. Any such agreement is a political process, and there is no obvious reason for or benefit from conducting an economic valuation on top of such an agreement, in particular as the cost incurred is one of the criteria to be taken into account when deriving the solution. Here people act as citizens, not as consumers, and given the broader range of aspects taken into account, the politically defined objectives can be expected to be better informed and more adapted to the local, cultural and other specifics of the respective problem than the one-dimensional, globally applicable economic valuation can possibly be (Funtowicz et al., 1998).

However, economics still has a role to play: it can be usefully applied to some specific aspects of ecosystem service loss, in particular to those with detectable impacts on existing markets. This requires first to explicitly define which kind of direct and indirect market costs is being analysed, to reveal the methods used for indirect costs and the assumptions made for hypothetical cost calculations -and then to add up only those with the same definition.

For this specific aggregate, the meaning needs to be explained to decision makers in order to avoid misunderstandings and subsequent misallocation of resources. For instance, for invasive species it is well possible to calculate yield losses and the costs of increased pesticide use: these are the economic costs of biological invasions (see e.g. Pimentel et al., 2000). However, this does not say anything about the social, ecological or other costs, nothing about the amenity and spiritual services affected, etc. Another example is the question about the budget implications of climate change for nature reserves—management costs are not that straightforward, but it should be possible to provide an estimate. As in these cases the question is an economic one expressed in monetary units, the answer may well accept the economic reductionism, as this is already inherent to the question. However, the management cost increase says nothing about a decreasing or increasing value of the respective protected area from a biodiversity or any other point of view.

A second point in case is the use of economic instruments as incentives for biodiversity conservation: successful ecosystem management, including the use of PES schemes, has in the past not been achieved by calculating the economic value of ecosystem services, but by introducing effective incentives for their maintenance. Such incentives can be money transfers, to be paid by individual beneficiaries (Pagiola, 2008) or, if they are reluctant to pay, by communities (Wunder and Albán, 2008; Frost and Bond, 2008), or public authorities on their behalf (Bennett, 2008; Muños-Piña et al., 2008). They can also include the transfer of merit goods, or barter trade, i.e. in-kind payments (Asquith et al., 2008). In any case, the level of transfers is set according to effectiveness (not economic efficiency) criteria: they must make a difference, triggering change in institutional routines and individual habits and behaviour. The difference between both is excellently illustrated by a study of Parry (2001) calculating the damage cost caused by car traffic (see Table 5). If internalised by a gasoline tax, according to environmental economics theory, an optimum of economic welfare would be reached.

This would have implied (in 2001) an eightfold increase of the gasoline tax in Canada, a fourfold one in the USA and a 60% increase in Australia—exactly the benign effects expected from internalising environmental costs. However, for the UK it would have meant halving its gasoline tax, reducing the anyway rather limited effect on emission reduction even further. The example illustrates the limits to environmental gains from economic cost calculations, that the economic cost do not necessarily represent effective incentives, and that the economic optimum can be a far cry from an environmental one. Streamlining policy to reach the economic optimum can result in programmed environmental damages. Instead of such expert valuations, multi-criteria assessments based on stakeholder participation reflecting multiple value systems are suggested, exercises in what has been called post-normal science (Funtowicz and Ravetz, 1993). Each resulting decision comes with a price, but the price of an object should not be confused with the object itself.

Table 5
Optimal gasoline tax calculation.

Health damages	0.16 €/l
Climate change, global warming	0.08 €/l
Traffic jams	0.16 €/l
Optimal gasoline tax	0.40 €/l

If such a broader context is used in developing and applying economic instruments, like in the South African “Working for Water” programme, a PES mainly funded as a poverty relief scheme, ecosystems can be protected in the long run, by turning the local populations into their custodians (Turpie et al., 2008). On the institutional aspect, transitions are necessary at all policy levels: maintenance of ecosystem services can be achieved without economic valuation, but not without sustainability policies.

6. Conclusion

Environmental scientists have in the past analysed the reproduction cycles of nature with limited attention to their role in the economic and societal context, while the economists' views so far focused on monetising the natural capital stock. In both cases, biodiversity is considered an essential element. However, in particular for our analysis focussing on biodiversity at the ecosystem level, it is not so much the individual elements that are essential (functional substitution occurs in biological systems), but the dynamics, including the resilience and the viability, which is safeguarding the ecosystem services. Consequently, in order to get an impression of the value of biodiversity these dynamics would need to be measured and valued (although their value could hardly be measured in monetary terms), not the preferences of some humans in a specific situation for certain benefits.

Environmental (and of course economic) scientists should be aware of the pitfalls of economic theory in order to avoid promoting an approach to biodiversity conservation and ecosystem protection which is a mere caricature of their initial intentions (Rees, 2006). As has been shown, there is no sound way of calculating the value of ecosystem services beyond the immediate expenditures needed. Is this a pity? We dare say no—even a price figure for all ecosystem services, even a soundly calculated one, would not be too helpful for defining political priorities. Aggregates can be marvellous alarm bells (and for this behalf they even can have a significant error margin), but due to aggregation they do not indicate priorities.

To operationalise the intention which was the motivation and driving force for monetisation in the first place, to motivate politics to take action for the conservation of biodiversity and maintenance of ecosystem services, it is essential to address the deeper causes of biodiversity loss, i.e. the drivers behind the pressures causing biodiversity losses (Spangenberg, 2007; CBD, 2006; MA, 2005). To achieve this aim, a systematic analysis of a multi-dimensional, multi-agent and multi-level problem is needed (as e.g. in the project ALARM: Settele et al., 2005). Then priority actions can be identified, and their costs will be one issue to be considered. But due to the multi-dimensional approach, this is but one argument, and biodiversity conservation cannot be undertaken based on a cost–benefit-analysis.

Ecosystems yield ecological functions which in turn provide various types of benefits easily understood by the public and by decision makers. Safeguarding them can be a political decision, not in need of an economic justification by valuing the services. Safe from economic reservations against all policy interventions (including those for nature protection, and trying to avoid or at least minimise them; Eckersley, 2006), such a decision would address the ecosystem as whole, not individual services, as the good to be protected.

Such reporting is not dependent on economic figures: forest decline has always been measured and communicated in terms of area and severity of tree damage, not as damage costs, and for desertification, poverty or unemployment the same applies: in most cases physical figures are at least as appealing as monetary ones. Thus instead of calculating absolute values for ecosystem services, with all the problems of economic thinking and of the methods applied (for a detailed discussion see e.g. Rees, 2006 or Norton and Noonan, 2007), economic instruments should be used in a framework of politically defined priorities.

The importance of quantitative monetary figures is often overestimated. The limits of their appeal and value (sic!) apply not only to the valuation of ecosystem services: economists have calculated the increase of happiness caused by a partnership or marriage to be worth about US$ 100,000 (WZB, 2008). Hardly ever this figure has motivated humans to remain related, or to separate. Back to biodiversity: economic instruments may well be helpful safeguarding ecosystems and the services they provide, but economic analysis is not the adequate method to determine the objectives or priorities of conservation policies, and even less so by means of economic valuation used in cost–benefit analyses.

References

Arnsperger, C., Varoufakis, Y., 2006. What is neoclassical economics? Post-autistic Economic Review 38, 2–12.

Asquith, N.M., Vargas, M.T., Wunder, S., 2008. Selling two environmental services: in-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. Ecol. Econ. 65, 675–684.

Barkman, J., Glenk, K., Keil, A., Leemhuis, C., Dietrich, N., Gerold, G., Marggraf, R., 2008. Confronting unfamiliarity with ecosystem functions: the case for an ecosystem service approach to environmental valuation with stated preference methods. Ecol. Econ. 65, 48–62.

Bennett, M.T., 2008. China's sloping land conservation program: Institutional innovation or business as usual? Ecol. Econ. 65, 699–711.

Bethge, P., von Bredow, R., Schägerl, C., 2008. Marktplatz der Natur. SPIEGEL 21 132–147.

Bonnedahl, K.J., Eriksson, J., 2007. Sustainable economic organisation: simply a matter of reconceptualisation or a need for a new ethics? Int. J. Innov. Sust. Dev. 2, 97–115.

Brey, R., Riera, P., Mogas, J., 2007. Estimation of forest values using choice modeling: an application to Spanish forests. Ecol. Econ. 64, 305–312.

CBD Convention on Biological Diversity, Secretariat, 2006. Summary of the Second Global Biodiversity Outlook. CBD, Montreal.

Common, M., 2007a. Measuring national economic performance without using prices. Ecol. Econ. 64, 92–102.

Common, M., 2007b. The dangers of extended, but incomplete, accounting for measures of economic performance in a world of imperfect knowledge. Ecol. Econ. 64, 239–249.

Conklin, H.C., 1980. Ethnographic Atlas of Ifugao: A Study of Environment, Culture and Society in Northern Luzon. Yale Universtiy Press, New Haven.

Costanza, R., 1980. Embodied energy and economic valuation. Science 210, 1219–1224.

Costanza, R., Cumberland, J., Daly, H., Goodland, R., Norgaard, R., 1998. An Introduction to Ecological Economics. CRC Press LLC, Boca Raton, FL, USA.

de Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol. Econ. 41, 393–408.

Eckersley, R., 2006. From the liberal to the green democratic state: upholding autonomy *and* sustainability. Int. J. Innov. Sust. Dev. 1, 266–283.

Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: an overview of the issues. Ecol. Econ. 65, 663–674.

European Commission, 2008. TEEB The Economics of Ecosystems and Biodiversity. An Interim Report. European Commission, Banson, Cambridge, UK.

Folmer, H., Gabel, H.L., Opschoor, H., 1995. Principles of Environmental and Resource Economics. E. Elgar, Aldershot, UK.

Frost, P.G.H., Bond, I., 2008. The CAMPFIRE programme in Zimbabwe: payment for wildlife services. Ecol. Econ. 65, 776–787.

Funtowicz, S., Ravetz, J., 1994. The worth of a songbird: ecological economics as a post-normal science. Ecol. Econ. 10, 197–207.

Funtowicz, S., Ravetz, J., 1993. Science for the post-normal age. Futures 25 (7), 735–755.

Funtowicz, S., Ravetz, J., O'Connor, M., 1998. Challenges in the use of science for sustainable development. Int. J. Sust. Dev. 1 (1), 1–10.

Gallai, N., Salles, J.-M., Settele, J., Vaissière, B.E., 2008. Economic valuation of the vulnerability of world agriculture confronted to pollinator decline. Ecol. Econ., doi:10.1016/j.ecolecon.2008.06.014.

Gilipin, A., 2000. Environmental Economics: A Critical Overview. Wiley, Chiscester, UK (cit in Kumar, Kumar 2008).

Guerrero, E., 2008. Ecosystem approach to water management. Some thoughts from Latin America. IUCN-CEM newsletter April 2008. EcoBytes 1, 1–2.

Hussen, A.M., 1999. Principles of Environmental Economics. Routledge, London/New York.

IUCN, 2006. CEM World Conservation Union Commission on Ecosystem Management. Biodiversity, Livelihoods. IUCN, Gland, Switzerland.

Keen, S., 2001. Debunking Economics. The Naked Emperor of the Social Sciences. Pluto Press Australia, Annandale, NSW, Australia.

Klein, A.-M., Vaissière, B.E., Cane, J.H., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C., Tscharntke, T., 2007. Importance of pollinators in changing landscapes for world crops. Proceedings of the Royal Society B 274, 303–313.

Knoche, S., Lupi, F., 2007. Valuing deer hunting ecosystem services from farm landscape. Ecol. Econ. 64, 313–320.

Köck, W., 2008. Standpunkt. UFZ Newsletter Spezial 4/2008, 18 (http://www.ufz.de/data/ufz_spezial_april08_20080325_WEB8411.pdf).

Kosoy, N., Corbera, E., 2009. Payments for ecosystem services as commodity fetishism. Ecol. Econ. 69 (6), 1228–1236.

Kroeger, T., Casey, F., 2007. An assessment of market-based approaches to providing ecosystem services on agricultural lands. Ecol. Econ. 64, 321–332.

Kumar, M., Kumar, P., 2008. Valuation of ecosystem services: a psycho-cultural perspective. Ecol. Econ. 64, 808–819.

MA Millennium Ecosystem Assessment, 2005. Synthesis Report. MA Secretariat, Washington DC.

Martinez-Alier, J., Munda, G., O'Neill, J., 1998. Weak comparability of values as a foundation for ecological economics. Ecol. Econ. 26, 277–286.

Miththapala, S., 2008. Natural capital. World Conservation 38 (1), 21–22.

Munda, G., 2004. Social multi-criteria evaluation: methodological foundations and operational consequences. Eur. J. Oper. Res. 158 (3), 662–677.

Munda, G., Nijkamp, P., Rietveld, P., 1994. Qualitative multicriteria evaluation for environmental management. Ecol. Econ. 10, 97–112.

Muños-Piña, C., Guevara, A., Torres, J.M., Braña, J., 2008. Paying for the hydrological services of Mexico's forests: analysis, negotiations and results. Ecol. Econ. 65, 725–736.

Norgaard, R.B., 2009. Ecosystem services: from eye-opening metaphor to complexity blinder. Ecol. Econ. 69 (6), 1221–1227.

Norton, B.G., Noonan, D., 2007. Ecology and valuation: big changes needed. Ecol. Econ. 63, 664–675.

Nunes, P.A.L.D., van den Bergh, J.C.J.M., 2001. Economic valuation of biodiversity: sense or nonsense? Ecol. Econ. 39, 203–222.

O'Neill, J., 1997a. Value Pluralism, Incommensurability and Institutions. Valuing Nature? Economics. Ethics and Environment. Routledge, Foster J. London.

O'Neill, J., 1997b. Managing without prices: the monetary valuation of biodiversity. Ambio 26 (8), 546–550.

Ojeda, M.I., Mayer, A.S., Solomon, B.D., 2008. Economic valuation of environmental services sustained by water flows in the Yaqui River Delta. Ecol. Econ. 65, 155–166.

Pagiola, S., 2008. Payments for environmental services in Costa Rica. Ecol. Econ. 65, 712–724.

Parry, I., 2001. Are gasoline taxes in britain too high? www.rff.org (retrieved June 2001).

Pimentel, D., Lach, L., Zuniga, R., Morrison, D., 2000. Environmental and economic cost associated with non-indigenous species in the United States. Bioscience 50, 53–64.

Rees, W.E., 2006. Why conventional economic logic won't protect biodiversity. In: Lavigne, D.M. (Ed.), Gaining Ground: In Pursuit of Ecological Sustainability.. Canada, International Fund for Animal Welfare; Limerick, Ireland, University of Limerick, Guelph.

Rehdanz, K., Maddison, D., 2008. Local environmental quality and life-satisfaction in Germany. Ecol. Econ. 64, 787–797.

Reid, W., Watson, R., Mooney, H., 2005. 'Ecosystem Services': A Vital Term in Policy Debates. , Retrieved from http://www.scidev.net/Editorials/index.cfm? SciDevNet (06.07.05).

Sagoff, M., 1998. Aggregation and deliberation in valuing environmental goods: a look beyond contingent pricing. Ecol. Econ. 24, 193–213.

Sagoff, M., 1990. The Economy of the Earth. Cambridge University Press, Cambridge, MA, USA.

Sandhu, H.S., Wratten, S.D., Cullen, R., Case, B., 2008. The future of farming: the value of ecosystem services in conventional and organic arable land. An experimental approach. Ecol. Econ. 64, 835–848.

Schägner, J.P., 2008. Die monetäre Bewertung unserer Wälder—Der Wert des Waldes in einer ökonomisierten Welt. Ökologisches Wirtschaften, pp. 24–26.

Sen, A., 1989. Economic methodology: heterogeneity and relevance. Social Res. 56, 299–330.

Settele, J., Martin, K., 1998. Rice terraces of ifugao (n-luzon, philippines)—ecological history and developments. In: Settele, J., Plachter, H., Sauerborn, J., Vetterlein, D. (Eds.), Rice Terraces of Ifugao (Northern-Luzon, Philippines). Conflicts of Landuse and Environmental Conservation., UFZ-Bericht 5/1998, UFZ, Leipzig, pp. 13–28.

Settele, J., Hammen, V., Hulme, P., Karlson, U., Klotz, S., Kotarac, M., Kunin, W., Marion, G., O'Connor, M., Petanidou, T., Peterson, K., Potts, S., Pritchard, H., Pysek, P., Rounsevell, M., Spangenberg, J.H., Steffan-Dewenter, I., Sykes, M., Vighi, M., Zobel, M., Kühn, I., 2005. ALARM: assessing large-scale environmental risks for biodiversity with tested methods. GAIA 14, 69–72.

Söderbaum, P., 2000. Ecological Economics. Earthscan, London.

Spangenberg, J.H., 2007. Biodiversity pressures and the driving forces behind. Ecol. Econ. 61, 146–158.

Spangenberg, J.H., 2005. Die ökonomische Nachhaltigkeit der Wirtschaft. Theorien, Kriterien und Indikatoren. Sigma, Berlin.

Spangenberg, J.H., 2001. Investing in sustainable development. Int. J. Sust. Dev. 4 (2), 184–201.

Spangenberg, J.H., Settele, J., 2009. Neither climate protection nor energy security: bio-fuels for biofools? Uluslararasi Iliskiler—Int Rel 5, 89–108.

Spangenberg, J.H., Omann, I., 2006. Assessing social sustainability: social sustainability and its multicriteioa assessment in a sustainability scenario for Germany. Int. J. Innov. Sust. Dev. 1 (4), 318–348.

Spash, C.L., Vatn, A., 2006. Transferring environmental value estimates: issues and alternatives. Ecol. Econ. 60, 379–388.

Swinton, S.M., Lupi, F., Robertson, G.P., Hamilton, S.K., 2007. Ecosystem services and agriculture: cultivating agricultural ecosystems for diverse benefits. Ecol. Econ. 64, 245–252.

Tseng, W.-Ch., Chen, C.-C., 2008. Valuing the potential economic impact of climate change on the Taiwan trout. Ecol. Econ. 65, 282–291.

Turpie, J.K., Marais, C., Blignaut, J.N., 2008. The working for water programme: evolution of payments for ecosystem services mechanisms that address both poverty and ecosystem service delivery in South Africa. Ecol. Econ. 65, 788–798.

Vatn, A., Bromley, D.W., 1994. Choices without prices without apologies. J. Env. Econ. Manage. 26, 129–148.

Wätzold, F., Lienhoop, N., Drechsler, M., Settele, J., 2008. Estimating optimal conservation in the context of agri-environmental schemes. Ecol. Econ. 68, 295–305.

Way, M.J., Heong, K.L., 1994. The role of biodiversity in the dynamics and management of insect pests of tropical irrigated rice—a review. Bull. Entomol. Res. 84, 567–587.

WCED World Commission on Environment and Development, 1987. Our Common Future (The Brundtland Report). Oxford University Press, Oxford.

Winkler, R., 2006. Valuation of ecosystem goods and services. Part 1. An integrated dynamic approach. Ecol. Econ. 59, 82–93.

Wunder, S., Albán, M., 2008. Decentralised payments for environmental services: the cases of Pimampiro and PROFAFOR in Ecuador. Ecol. Econ. 65, 685–698.

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