Environ Resource Econ (2011) 48:161–175
DOI 10.1007/s10640-010-9413-2

Bringing Ecosystem Services into the Real World: An Operational Framework for Assessing the Economic Consequences of Losing Wild Nature

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Accepted: 8 September 2010 / Published online: 20 October 2010

Abstract Policy action to halt the global loss of biodiversity and ecosystems is hindered by the perception that it would be so costly as to compromise economic development, yet this assumption needs testing. Inspired by the recent Stern Review on the Economics of Climate Change, the leaders of the G8+5 nations commissioned a similar assessment of the economics of losing biodiversity, under the Potsdam Initiative on Biodiversity. Here, we propose a conceptual framework for such a global assessment which emphasizes several critical insights from the environmental economics and valuation literature: contrasting counterfactual

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Electronic supplementary material The online version of this article (doi:10.1007/s10640-010-9413-2) contains supplementary material, which is available to authorized users.

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scenarios which differ solely in whether they include specific conservation policies; identifying non-overlapping benefits; modeling the production, flow, use and value of benefits in a spatially-explicit way; and incorporating the likely costs as well as possible benefits of policy interventions. Tackling these challenges, we argue, will significantly enhance our ability to quantify how the loss of benefits derived from ecosystems and biodiversity compares with the costs incurred in retaining them. We also summarise a review of the current state of knowledge, in order to assess how quickly this framework could be operationalized for some key ecosystem services.

Keywords Ecosystem services · Biodiversity · Ecosystem modeling · Benefits · Millennium ecosystem assessment · Scenarios · Conservation · Development

1 Background

Humans have accomplished their domination of the Earth by spending legacies accumulated by the other species on the planet: burning fossil fuels, overharvesting wild populations, and replacing or degrading natural ecosystems. The resulting transformation of the atmosphere and the biosphere now threatens our sustainability through two unintended consequences—climate change, and the loss of ecosystem services provided by nature (MA 2005; IPCC 2007). Policy action to tackle these linked problems has been hindered by the perception that it would be so costly as to compromise economic development. However, in the case of climate change, the United Kingdom Government's Stern Review (Stern 2007) recently concluded that the benefits of strong and early remedial action would far outweigh its economic costs. This inspired the leaders of the G8+5 nations to commission a similar assessment of the economics of losing biodiversity, under the Potsdam Initiative on Biodiversity (2007). Here, we propose a conceptual framework for such a global assessment: an operational roadmap for quantifying how the loss of benefits derived from ecosystems and biodiversity compares with the costs incurred in retaining them.

We focus on instrumental economic value (which is only one dimension of the overall value of nature (Turner 1999)), even though there are many, ourselves included, who believe that biodiversity has intrinsic values which by themselves justify its conservation. If economic assessments find that conservation (in its myriad forms) confers a net economic gain, then that simply adds an economic argument against losing biodiversity and ecosystems, alongside the moral argument. If the results are that conservation incurs a net economic loss, then they will help quantify the net conservation bill.

Understanding the economic consequences of losses in biodiversity and ecosystems is distinct from estimating the value of nature as a whole (Costanza et al. 1997), which is of course infinite. Even within the current biodiversity crisis we are (fortunately) far from erasing all wild nature and the benefits we derive from it. Instead, the relevant question for society is: do the economic benefits of preventing further losses of biodiversity and ecosystems exceed the costs? Inevitably, as biodiversity and/or ecosystems decline, there is a point where the marginal benefits of conservation exceed the marginal costs. The marginal benefits tend to increase as wild nature and the services it provides become scarcer, while the marginal costs (particularly the opportunity costs) tend to decline as human activities expand away from the areas best suited for them. Fig. 1 is a conceptual representation of this dynamic, ignoring the specific units of loss, but highlighting the point at which further declines in biodiversity or ecosystems become uneconomic. The framework we now go on to propose tries to address some key limitations of previous assessments (e.g. MA 2005; Costanza et al. 1997), and by

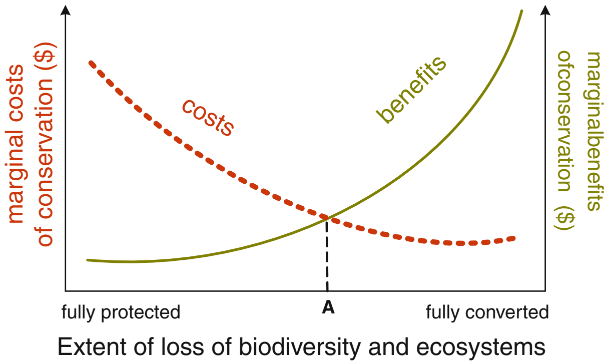


Fig. 1 Conceptual representation of changes in the marginal costs and benefits of conservation as biodiversity and/or ecosystems are lost. Halting such loss makes economic sense when the benefits exceed the costs (to the right of point A)

so doing provide a way of discovering whether, for any given system, we have now reached the crossover point in Fig. 1 so that alongside moral arguments there is an economic case for its conservation (Balmford et al. 2008).

2 Operational Framework for Evaluating Ecosystem Services

Our proposed framework is best explained graphically. Starting at the top left of Fig. 2, estimating marginal costs and benefits requires that we first define specific states of the world (circles), generated from counterfactual scenarios. There has been much social science research on how scenarios should be developed and by whom (Kok et al. 2007; Biggs et al. 2007). We consider two contrasting scenarios to illustrate our framework, but there is no reason why a suite of scenarios could not be investigated. Here we have one scenario reflecting continuing business-as-usual (with the associated loss of biodiversity and the benefits it bestows on people) and another otherwise identical scenario, where policies to achieve particular conservation goals are in place. Those goals and policy actions must be defined explicitly—which in turn requires understanding the causes of biodiversity and ecosystem loss. For example, a goal of halving global rates of tropical deforestation by 2020 would necessitate actions (such as establishing protected areas, or reinforcing legislation) that tackle its underlying causes (e.g. poverty, agricultural expansion, unregulated logging). The analysis undertaken with direct respect to the Potsdam Initiative here proscribes a level of success of these interventions (likely total success) since the concept behind the analysis is to indicate a loss of human welfare benefits by failing to meet a given policy goal. For example, what is the difference in delivery of ecosystem services and levels of biodiversity between a successful set of polices or interventions and the current trend of things? Of course the selection of policies and interventions will strongly affect the values attributed to ecosystem services, as well as the net gains and losses across stakeholders. For example, 'halving the global rate of deforestation' as a policy goal implicitly places opportunity costs on parts of the world least likely to be able to absorb these costs without welfare losses. However, for such an undertaking as a global understanding of the costs of losing biodiversity and ecosystems, such a process of selecting the policies and interventions must be made in a deliberative and transparent process, ideally listing alternative policies not assessed.

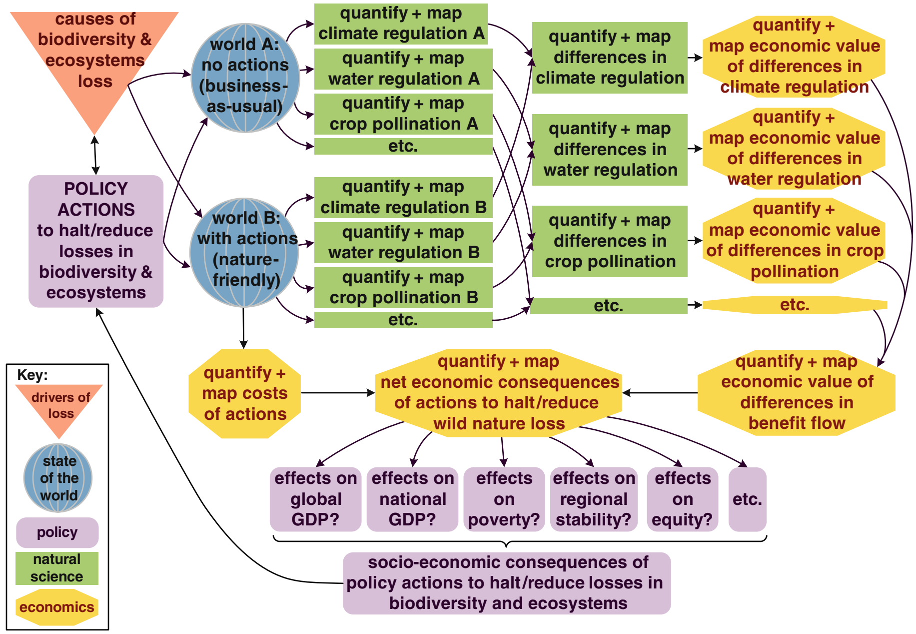


Fig. 2 Framework for assessing the economic consequences of losing biodiversity and ecosystems. The evaluation is done by contrasting—in a spatially-explicit way—the differences in costs and benefits between two hypothetical states of the world, equal in everything except for the implementation (or not) of a specific set of policy actions for reducing losses in biodiversity and ecosystems

Quantifying the consequences of these policy alternatives requires identifying the different ways in which changes in ecosystems affect human welfare. For example, establishing forest protected areas may impact global climate regulation, regional water supply, local crop pollination, global availability of pharmaceuticals, local supply of wild meat and medicinal plants, and national and international nature-based tourism (MA 2005). These varied linkages between ecosystem function and human welfare are termed ecosystem services (Daily 1997; MA 2005). Their economic values are typically assessed separately and then combined, but if the resulting overall estimate of the economic consequences of policy alternatives is to be robust, it is vital that the services considered must make non-overlapping contributions to human wellbeing.

This is not trivial. The Millennium Ecosystem Assessment (2005) proposed a classification of ecosystem services that has become widely used in policy and education, but which is prone to double-counting (for example, by listing "water purification," and the "provision of freshwater" as separate services (Boyd and Banzhaf 2007; Fisher et al. 2009). Here instead we disaggregate ecosystem services into three interlinking sets, which differ in their proximity to human wellbeing: core ecosystem processes, beneficial ecosystem processes and ecosystem benefits (Fig. 3; corresponding, respectively, to the concepts of intermediate services, final services and benefits in Fisher and Turner (2008)). We make a distinct division between processes and benefits. Processes are biophysical functions, while benefits are the end goods and services that directly affect the human welfare function. In this classification, core ecosystem processes are the basic ecosystem functions (e.g. nutrient and water cycling) which directly underpin those processes that in turn generate benefits to humans. Our beneficial ecosystem

Fig. 3 Illustration of the relationship between core ecosystem processes, beneficial ecosystem processes, and ecosystem benefits. The lists are not exhaustive. Further details of the correspondence between this classification and that followed by the Millennium Ecosystem Assessment is given in Fig. S1

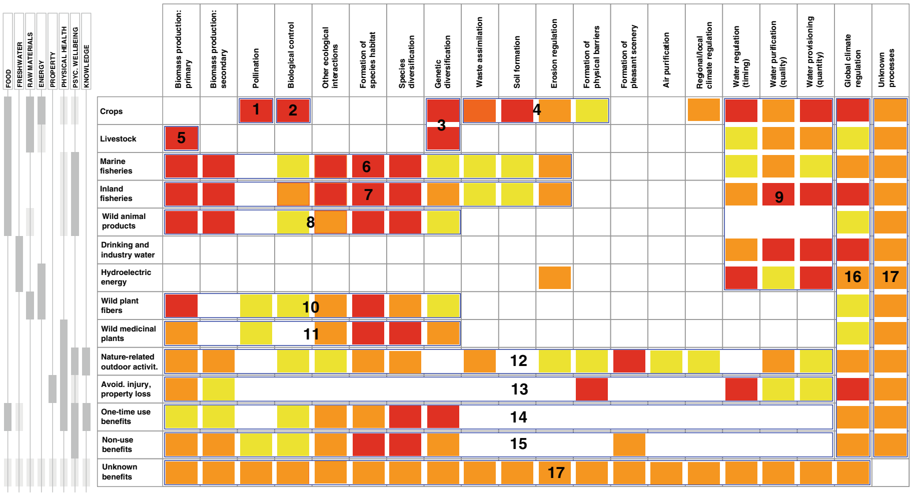


Fig. 4 Relationship between main types of ecosystem benefits (*rows*) and beneficial processes (*columns*); *Shades* indicate an estimate of the importance of the corresponding link between the two (the darker the shade the more important the link). For example, we know that pollination as a process (*column*) is very important for benefits (*row*) of particular crops, but is likely to make a less direct contribution to the benefit of wild plant fibers for construction. Numbers correspond to the 16 ecosystem benefits and beneficial processes (each delimited by a *bold line*) for which we obtained an expert-driven review of the current state of knowledge (Table 1), plus a general category of "unknown benefits" acknowledging our incomplete knowledge, but for which no review was possible. The *grey bars* on the *left* indicate the main types of benefit

processes (e.g., primary and secondary biomass production, species diversification) are then the specific processes that deliver benefits to humans. Finally the benefits—usually with the addition of some other forms of capital, such as labour—are the end-products of the beneficial ecosystem processes (e.g. food from capture fisheries) which impact on human welfare.

The double-counting problem is best avoided by estimating values only for the separate end-benefits provided by ecosystem services (i.e. the right-hand elements in Fig. 3; Fisher and Turner 2008). However, in practice, some beneficial processes (such as climate regulation) have impacts on many benefits (food, drinking water, water for irrigation), and are more amenable to analysis if considered in this cross-cutting way (for further elaboration, see Sect. 3; Fig. 4). In these cases we recognize that it may be necessary to place values on beneficial processes (rather than separately, on each benefit that they impact), provided that care is taken to minimize double-counting. This is why the distinction between beneficial and core processes is essential.

The next stage in the framework—examining the production, flow and use of each service (rectangles in Fig. 2)—needs to recognize and address the fact that these vary spatially and that each service will have to be mapped at the scale that it occurs (Fisher et al. 2009). As examples of such variation, the impact of forests on climate regulation (through carbon storage and sequestration) varies with soil composition and is delivered on a global scale; the role of forests in regulating water supplies varies with topography and is delivered regionally; and their contribution to crop pollination varies with their distance to local crop fields. Clearly, while the service is produced in one area, the benefit may flow to another.

Predicting how policy actions affect each service (compared with business-as-usual) therefore requires modeling its production, flow and use in a spatially-explicit and scale-sensitive way. Most modeling to date has been unavoidably simplistic (e.g. extrapolating from a few inevitably unrepresentative studies to the whole of a biome; Costanza et al. 1997). This can generate misleading and potentially biased results, and a more refined approach is now required. Fortunately, for several services ecological models are now much improved (Balmford et al. 2008, and Sect. 3 below), and for others, being spatially explicit should highlight simplistic assumptions and thereby identify where model elaboration is most needed.

A spatially-explicit approach is also required in the next step of the framework—attaching economic values to policy-contingent changes in ecosystem service delivery (right-hand octagons in Fig. 2). Some services—such as climate regulation through carbon sequestration—are equally valuable wherever they are generated. But the value of others varies spatially—the value of water flow regulation, for instance, depends on factors influencing local supply (e.g. rainfall) and demand (e.g. human population density). Some service values will also vary between scenarios, reflecting price elasticity of demand (e.g. the price of fish will change as supply changes).

There is currently a large toolkit of methods suitable for valuing a wide variety of ecosystem services. For example, revealed preference techniques such as the travel cost or hedonic pricing method can be applied to assess values as diverse as the pleasures of bird watching or the enjoyment of a quiet country garden. Similarly production function methods can be used to assess values as abstract as the pollination service of wild bees for increasing coffee yields (see Ricketts et al. 2004). However, these are all instrumental services, generating value through either direct or indirect human use. Valuation becomes less tractable when we consider non-use values such as those associated with the existence of biodiversity (see limitations below).

Declines in biodiversity and ecosystems affect not only the delivery and value of ecosystem services but also the resilience of natural systems (Folke et al. 2004). Our understanding of this relationship is still incipient, but when available, this information should be incorporated in estimating changes in service production, flow, use and value, in a spatially explicit way (Bateman et al. 2010; Polasky et al. 2010). For example, there is now some information on the conditions that trigger collapse in marine fisheries (Worm et al. 2006) and so it may be possible to obtain at least some indication of how far different fisheries are at risk under each scenario, and how that may affect the provision of benefits from marine fisheries.

Subjective judgments are often inherent in valuation exercises, and therefore assumptions and elicitation techniques must be transparent. Resultant imputed values for distinct individual services may be combined, to provide an overall, spatially-explicit quantification of how the gross economic benefits derived from nature are affected by a conservation-relevant policy. The next step—assessing its net economic consequences (bottom octagons, Fig. 2)—requires understanding its costs as well as benefits (Naidoo and Ricketts 2006; Bruner et al. 2008). Decision-makers are particularly sensitive to the costs of actions, yet this information is often lacking from economic evaluations of ecosystem services (e.g. a recent study (World Bank FAO 2008) estimated that through unsustainable harvesting practices we lose about US$50 billion/year in potential revenues from marine fisheries, yet omitted the costs of the global fisheries reforms needed to capture those lost benefits). Costs again vary spatially, but usually in a different way from benefits. Although the benefits of many ecosystem services accrue at regional and global scales, the often-sizeable opportunity costs of conservation (such as foregone agricultural expansion) are usually borne by local, typically poorer, people. This geographical mismatch between winners and losers helps explain the continued degradation of natural resources even when, in net terms and at a global level, conservation often appears to make economic sense (Balmford et al. 2002). Understanding this is essential for making new policy interventions (such as proposed mechanisms for reducing emissions from deforestation and forest degradation (Miles and Kapos 2008)) effective and equitable.

Finally, with costs and benefits arising from specific policy actions now mapped out, comparing the two can help inform key questions about the broader socio-economic consequences of losing biodiversity and ecosystems versus intervening to conserve them (bottom boxes, Fig. 2). The results at this scale will not be precise, but rather give a general relationship and perhaps magnitude of the spatial costs and benefits between the counterfactual scenarios, potentially informing questions such as: What will be the likely effects of conservation actions on the global economy, or on the relative wealth of different countries? How does the spread of gains and losses compare with spatial indicators of poverty? And how, therefore, might conserving nature impact development goals, equity, and socio-political stability?

3 Real-World Applicability of the Framework

Our framework can in principle be applied to evaluate the economic consequences of conservation actions at any spatial scale, from local to global—from assessing the impact of one reserve to that of the worldwide protected area system. By adopting differing temporal scales and discount rates, the effect of different assumptions about inter-generational equity can also be explored (Stern and Taylor 2007). Here we described the framework based on two scenarios, but other research ventures will want to include scenario analysis with the number and type of scenarios deemed necessary by the appropriate stakeholders.

Of course the challenges in generating the necessary data are substantial, especially at large spatial scales. However, we believe they are not insurmountable. Significant progress has recently been made in developing global and regional scenarios (MA 2005), and in understanding and mapping the costs of conservation interventions (Naidoo and Iwamura 2007; Bruner et al. 2008). Crucially, there has also been good progress in developing spatially-explicit models for predicting the production of services from biodiversity and ecosystems under different scenarios, in both marine (e.g. Alder et al. 2007) and terrestrial (e.g. Kindermann et al. 2008) realms.

A crucial aspect of applying the framework is identifying ecosystem services that can be evaluated in practice. The ideal is that, in a given project, an ecosystem service of interest is distinct and easily distinguishable from other services; its benefits flows are clear and unique; and that the services or benefits can be measured, mapped and modeled; and valued meaningfully. However socio-ecological systems are complex and this ideal is not easily met. To understand, in our global example, which ecosystem services can be empirically evaluated, we first devised a matrix describing the relationships between key beneficial processes and benefits (columns and rows in Fig. 4, where our estimate of the relative contribution of a given process to a particular benefit is indicated by the colour of the cell where they intersect). The initial identification of the benefits and beneficial process came from a synthesis of several lists in the literature (Daily 1997; Costanza et al. 1997; de Groot et al. 2002; MA 2005) and were extensively discussed in a series of workshops for "The Economics of Ecosystems and Biodiversity: Scoping the Science" report (Balmford et al. 2008). Next, we identified 16 benefits or beneficial processes (indicated by bold lines in Fig. 4) that are potentially amenable to modeling and valuation, but which are non-overlapping and which thereby avoid the double-counting problem. These usually correspond to benefits (rows in Fig. 4), but in some instances (such as climate regulation—Stern 2007), ecological and economic knowledge is better aligned with beneficial processes (columns in Fig. 4). We also included a category of "Unknown benefits" acknowledging our incomplete understanding of all the links between the functioning of ecological systems and human welfare.

We then asked leading experts across a range of disciplines to assess whether current knowledge is sufficient to model and map the global flow, use and value of each of our 16 key benefits and beneficial processes. Our experts were identified by a snowball-sampling technique where a core group of around a dozen scientists and economists suggested experts in the most relevant fields. These experts then often got involved in the project and/or suggested others. In the end there were over 50 scientists and economists involved in the assessment (but see Balmford et al. 2008 for the full account). Our experts identified specific analyses needed to address major knowledge gaps (Table 1), which we then prioritised according to two criteria:

- *Importance to human wellbeing*: The degree to which the service is likely to affect the overall results of an economic assessment, based on a qualitative appraisal of the predicted magnitude of its overall economic value (coded in decreasing order of importance as A, B, or C).
- *Feasibility*: The likelihood that the particular analysis recommended can be successfully undertaken in a short timeframe (one year, given appropriate funding) (coded in decreasing order of feasibility as 1, 2 or 3). High feasibility means that adequate global models and data already exist or were considered to be attainable in the time available (for detailed results, see Balmford et al. 2008).

The combination of these two criteria produced an overall priority ranking (Table 1), from very high priority, when the analysis was considered to be both highly important and highly feasible (A1), to very low priority, when the analysis was considered of lower importance and not particularly feasible in the short-term (C3). Note that the feasibility and importance of the rankings shown here may change as knowledge evolves, and are likely to vary from region to region.

While this categorization is tentative, rather than definitive, we believe it demonstrates that it would be possible to assess in the short term the economic consequences of declines in several important services. For example, our understanding of the economic consequences of changes in marine fisheries (Alder et al. 2007) and global climate regulation (Stern 2007) is adequate and they are both of high importance at the global scale. Developing sophisticated models of some other services—such as nature-related outdoor activities or natural hazard regulation—will take longer (Table 1). However, such gaps should not prevent a global assessment from being done based on better-known services—it simply means that early results will grossly underestimate the net economic value of conservation.

4 Limitations

As mentioned above, there are numerous data and modelling concerns for implementing our proposed framework. Often, there may not be sufficient knowledge of how the ecological system functions, let alone its complex interactions with the social system of interest—and hence its value to society. But by making such inadequacies clear—for example by making it explicit that a particular benefit is inappropriately assumed to be generated at a uniform rate or to have a constant value across the globe—applying our proposed framework should highlight rather than mask gaps in our understanding, and pinpoint those gaps of greatest importance for decision-making.

Of course there are limitations to valuation both from practical and methodological perspective (Kahneman 1986; Vatn and Bromley 1994; Bateman et al. 1997; Norgaard et al. 1998; Gowdy and Mayumi 2001). First is the assumption that preferences for nonmarket

Table 1 Main types of benefits or beneficial services by which biodiversity and ecosystems contribute to human wellbeing

Benefit or process	Recommended analysis	Priority
All	Scenario development	Essential
1. Wild crop pollination	Global pollination model building from landscape-scale assessments	B2
2. Biological control of crop pests	Global crop biological control model building from landscape-scale assessments	B3
3. Genetic diversity of crops and livestock	Global risk assessment of crop/livestock disease from loss of genetic diversity	B2
4. Soil quality for crop production	4a. Global valuation of improvement in soil quality from changes in soil biota (internal effects)	B3
	4b. Global valuation of soil subsidies when agriculture expands into natural ecosystems (conversion effects)	B1*
	4c. Global model of the protective effects to crop soils of neighbouring natural habitat (neighbouring effects)	C2*
	4d. Global model of value of wildlife fertilisers.	C3*
5. Livestock	Global model of rangeland contribution to livestock production	B1*
6. Marine fisheries	Global model of marine fisheries provision	A1*
7. Inland fisheries	Global model of inland fisheries provision including aquaculture	B2*
8. Wild animal products	8a. Pantropical model of wild meat provision	B1*
	8b. European and North-American model of domestic recreational hunting	B1
9. Fresh water provision and regulation	9a. Global hydrological model for water provision (quantity)	A1*
	9b. Global hydrological model for water regulation (timing)	A2*
	9c. Global hydrological model for water purification (quality)	A2*
10. Wild harvested fibres	Global model of sustainable timber production from natural forests	B2*

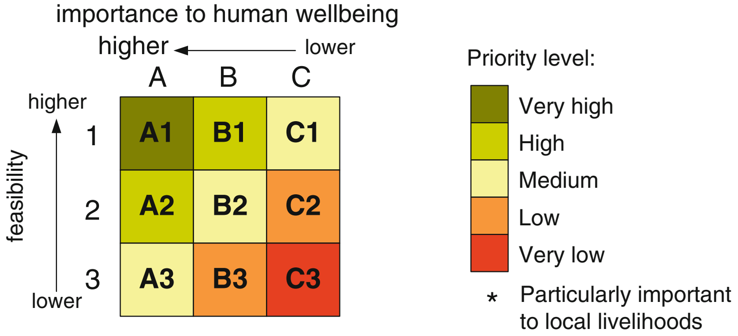
goods such as some ecosystem services can even be revealed (Gowdy and Erickson 2005; MacMillan et al. 2006). It is difficult to defend the assumption of invariant preferences and perfect information. However, some assumptions about values can be made based on the probable shape of the marginal cost and benefit curves under different level of service provision (Bateman et al. 2010). The practical problems of valuation become less tractable when

Table 1 continued

11. Wild medicinal plants	Global model of provision of medicinal plants (harvesting)	**B3***
12. Nature-related outdoor activities	12a. Global model of tourism in protected areas	**B3**
	12b. Global model of the value of green areas for local recreation	**B2**
13. Natural hazard regulation	13a. Global model of the value of forests (and possibly wetlands) for regulating catchment-borne floods	**B2***
	13b. Global model of the value of coastal ecosystems for mitigating the effects of tsunamis	**C2***
	13c. Global model of the value of coastal ecosystems for mitigating the effects of storm surges	**B2***
	13d. Global model of the value of forests for preventing landslides	**C2***
14. One-time biodiversity use	Global model of loss of option values for benefits derived from single species	**B1**
15. Non-use values	Global valuation of non-material benefits from biodiversity	**B3**
16. Global climate regulation	Global terrestrial model of carbon storage and sequestration	**A1**
17. Unknown benefits or processes	Non-quantifiable	**Not feasible**

NOTE: For each benefit or beneficial service, we recommend one or more possible analyses to better understand how it might change as biodiversity and ecosystems change. These analyses are coded, in decreasing order of economic importance, as A, B, or C and, in decreasing order of feasibility, as 1, 2 or 3. Those additionally marked with an asterisk were considered particularly valuable for local livelihoods (so where people are poor, these may directly influence the wellbeing of millions of people despite having low economic value).

Table key: Prioritization according to importance to human wellbeing (A to C) and feasibility (1 to 3):



we move to consider pure non-use values such as those associated with the continued existence of species which humans do not, even in some indirect, non-consumptive manner, make use of. The problem is not that there are not, in principle, methods that are theoretically applicable to such situations. Stated preference (SP) methods such as the contingent valuation of choice experiment approaches can, in theory, be applied to any good that generates values within an individual's utility function. Rather the problem is that most members of the public have never considered the value they might hold for the continued existence of some poorly understood species residing in a far off land. In such circumstances, individuals do not arrive at the SP exercise pre-armed with well-formed preferences. Of course in theory the SP study will make amends for this by providing respondents with the information needed to form those preferences during the exercise. However, studies have shown that apparently small and (from a theoretical perspective) irrelevant variations in the framing of valuation questions can yield substantial impacts on resultant values in such situations (see for example, Bateman et al. 2008). This does not undermine the use of SP techniques for situations where respondents do have prior, well-formed preferences or can discover them within the confines of the exercise. But issues such as existence value seem to pose a substantial challenge to any form of valuation assessment. Appending values to biodiversity, or making choices which involve both monetary estimates and species, is likely to be widely outside the normal range of decision making for a person—and therefore unlikely to yield robust or meaningful values. In these cases a decision is likely to be more informed by the precautionary principle, safe minimum standards, and/or a cost-effectiveness approach (Polasky et al. 2001, 2008; Laycock et al. 2009; Bateman et al. 2010).

Also, economic values are typically a function of income distribution and current allocation of rights, including property rights (Bockstael et al. 2000)—ignoring this might mean that ecosystem service assessments heavily reliant on stated preference techniques might further entrench current inequalities. Equity weighting in the appraisal process may help to address this issue (see Srinivasan et al. 2008 for an example). Economic valuation also requires judgments about how individuals and society understand and attend to risk via discount rates, time horizons and even the scenarios people are asked to value. These choices will likely affect the outcome of any analysis, and therefore they must be made not only transparent, but also include a range of values so that decision-makers and stakeholders can see how sensitive welfare outcomes are to the input assumptions. For all of these reasons there is validity in calling for societal choices, especially in the domain of environmental decision-making, to be made without recourse to valuation (Vatn and Bromley 1994) or with the results of a cost-benefit analysis being a single component in a lager body of evidence (Turner 2007).

There are other limitations of such an ecological-economic modelling approach. The very action of including or excluding certain ecosystem services requires a value judgment that has implications. For example, there is likely bias in such an ecosystem service assessment towards services which more directly affect human welfare and towards those which we have a better ecological understanding. This might leave out services which could be critical in delivering human welfare, but for which we do not have good ecological models, or methods for valuation. Despite this bias, ecosystem services are inherently a way to add breadth to a policy decision by incorporating contributions to human welfare that might not otherwise be considered. While some ecosystem services might be undervalued or ignored the incorporation of any ecosystem services will, at least in theory, lead to a more informed decision. Any services that are underrepresented in a valuation exercise should certainly become part of the wider knowledge-base which should feed into the decision process (Turner 2007).

Our global assessment example was motivated by the Potsdam Initiative, but for any real-life applications at any scale, all relevant stakeholders should be included—in selecting the services to assess, through to developing the scenarios. Stakeholders can be brought into the process by a myriad of ways including participant appraisals, surveys, scenario workshops and citizen juries.

Our framework makes some simple assumptions about the effectiveness of conservation interventions. A more complex process would involve an analysis of the success of past interventions in order to judge the potential effectiveness of the scenario-derived interventions. Implementing the framework on a global scale can likely provide a first idea of the direct impacts of the policy interventions (or non-interventions), but probably not a clear indication of how the secondary and tertiary etc. impacts feedback into dynamic human welfare functions through social and market systems. We are still in early days of this level of integration at the macro-scale, but the results from undertaking a global assessment can give an indication of how various inputs to macroeconomic models may change. Zooming in, although we might be able to get decent spatial resolution on the areas of ecosystem service production, flow, benefits and key indicators of poverty, the framework here would be inadequate to address the impacts of how changes in local ecosystem service provision (e.g. soil retention) affects local or regional market prices for a good (e.g. maize) and therefore the effects on the low income consumers reliant on that market.

Transaction costs represent another limitation of applying such a framework, which might make it difficult to be utilized where funding, expertise and institutional support may be lacking. Another key issue is how to develop a complementary approach that focuses explicitly on services and benefits that conventional economic valuation fails to comprehensively express—such as cultural benefits. Valued landscapes and places with symbolic significance are often expressed as shared-values which diffuse out into social norms over long periods of time. Such values are perhaps best elicited through group- based deliberative processes and methods (Turner 2010). This is likely to become an important avenue of research in rapidly moving field of ecosystem services.

5 Conclusion

Despite these important caveats, we believe than by directly addressing key limitations to previous efforts—by contrasting scenarios, avoiding double-counting, being spatially explicit, and incorporating costs—our framework will help to provide a new and sharper set of tools for making decisions about the conservation of biodiversity and ecosystems. Although the framework can be applied at any spatial scale, its greatest potential, and the greatest challenges, will arise from its global application to multiple ecosystem services, as required by the Potsdam Initiative (2007). The first attempts to do so will seem unsatisfactory and highlight the need for better data and models than currently exist. However, being explicit about the nature of the global decisions that face us, and what we need to know to inform them, offers the best prospect of taking effective and timely action.

Acknowledgments We are indebted to very many people for stimulating discussion providing information during the development of these ideas. For stimulating discussion during the development of these ideas, we are particularly indebted to Ian Bateman and Aude Neuville (who also provided very useful comments on this manuscript) and Patrick Murphy. We are also very grateful to: our project partners Matt Walpole, Patick ten Brink, Marianne Kettunen, Leon Braat and Rudolf de Groot; the members of the Working Group on the Review on the Economics of Biodiversity Loss; and Katie Bolt, Martin Jenkins, Lera Miles and Ben Phalan. James Beresford, Kelly Flower, Antares Hernández, Hannah Peck, and James Waters researched the background literature on which many of the reviews underpinning Table 1 were based. For providing direct expert support to these reviews, we are heavily indebted to: Jackie Alder, Richard Bardgett, Jaap Bloem, Corey Bradshaw, Justin Brashares, Kenneth Cassman, Roland Cochard, Saul Cunningham, Will Darwall, Patrick Dugan, John Fa, Val Kapos, Alexandra-Maria Klein, Claire Kremen, Bernhard Lehner, Peter McIntyre, E.J. Milner-Gulland, Gert-Jan Nabuurs, Luke Parry, Gerardo Perillo, Sandra Postel, Francis Putz, John Robinson, Marcus Rowcliffe, Charles Sheppard, Brian Silliman, Ed Tanner, Jerome Vanclay, Reg Watson, Karen Weinbaum, Robin Welcomme, Sue Wells, Sue White, Neal Williams, Boris Worm, and Peter Zuidema. The following provided additional information for the reviews: Robin Abell, Elizabeth Bennett, Neil Burgess, Robert Costanza, Guy Cowlishaw, Sally Hacker, Belinda Hawkins, John Hopkins, Teresa Mulliken, Robert Naiman, Rashid Sumaila, Daniel Pauly, Pascal Peduzzi, Joan Roughgarden, Wayne Scoting, Charles Vörösmarty, Eric Wolanski, and Stanley Wood. Four anonymous reviewers provided constructive critiques, which served to greatly improve this manuscript. This work was developed as part of Study Contract 070307/2007/486089/ETU/B2 for the European Commission's Environment Directorate-General, under the project "The Economics of Ecosystems and Biodiversity: Scoping the Science". The opinions and views expressed here are those of the authors and do not necessarily reflect those of the European Commission. BF is funded by the "Valuing the Arc" programme of the Leverhulme Trust and by the Program in Science, Technology and Environmental Policy (STEP) at Princeton University.

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