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Review

Ecosystem Services: Origins, Contributions, Pitfalls, and Alternatives

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Abstract

The concept of ecosystem services (ES) has taken the environmental science and policy literature by storm, and has become almost *the* approach to thinking about and assessing the nature-society relationship. In this review, we ask whether and in what way the ES concept is a useful way of organising research on the nature-society relationship. We trace the evolution of the different versions of the concept and identify key points of convergence and divergence. The essence of the concept nevertheless is that the contribution of biotic nature to human well-being is unrecognised and undervalued, which results in destruction of ecosystems. We discuss why this formulation has attracted ecologists and summarise the resultant contributions to research, particularly to the understanding of indirect or regulating services. We then outline three sets of weaknesses in the ES framework: confusion over ecosystem functions and biodiversity, omission of dis-services, trade-offs and abiotic nature, and the use of an economic valuation framework to measure and aggregate human well-being. Underlying these weaknesses is a narrow problem frame that is unidimensional in its environmental concern and techno-economic in its explanation of environmental degradation. We argue that an alternative framing that embraces broader concerns and incorporates multiple explanations would be more useful, and outline how this approach to understanding the nature-society relationship may be implemented.

Keywords: ecosystem services, biodiversity, economic valuation, trade-offs, dis-services, political ecology

INTRODUCTION

The idea that human society benefits from the environment or nature in various ways, both directly and indirectly, is certainly not a new one, and can be traced back several millennia. But the modern-day concept emerged in the 1970s as 'environmental services' (Wilson and Matthews 1970), was re-named 'ecosystem services' in the mid-1980s (Ehrlich and Mooney 1983), and really gained momentum from 1997 onwards (Costanza et al. 1997; Daily 1997b; for histories, see Fisher et al. 2009; Gómez-Baggethun et al. 2010). The most popular current definition of ecosystem services (ES) is "the functions and products of ecosystems that benefit humans, or yield welfare to society" (MA 2005). This concept, originally intended as a metaphor (Norgaard 2010), has now become the basis for a large and rapidly expanding literature that seeks variously to measure, assess, and value aspects of societal dependence on nature. It can also claim to have triggered policy shifts of two kinds. Policy makers are asking for economic assessments or valuations of how biodiversity and ecosystem service loss might be translating into welfare

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loss [such as the TEEB (The Economics of Ecosystems and Biodiversity) study commissioned by the European Union (Sukhdev 2008)], and 90 governments agreed to set up an Inter-governmental Platform on Biodiversity and Ecosystem Services (www.ipbes.net). Simultaneously, a number of 'payments for ecosystem services' (PES) schemes have been launched, spanning watershed services, biodiversity conservation, and of course now carbon sequestration. Not surprisingly, an editorial in *Nature* suggested that "ecosystem services (have entered) into mainstream scientific and political thinking" (Anonymous 2009).

The ES concept has also attracted extensive debate and critical comment. Much of the criticism is 'internal', i.e., from those who believe in the usefulness of the concept and seek improvements in terminologies (e.g., Boyd and Banzhaf 2007; Fisher et al. 2009) and methods (e.g., Barkmann et al. 2008). A more critical stream has emerged around the specific translation of ES as an analytical concept into PES as a policy instrument for solving environmental problems. Concerns about commodification of nature have been central to this stream (Kosoy and Corbera 2010; McAfee and Shapiro 2010). Some analysts have cautioned that the ES concept needs to be more carefully unpacked and that "there are risks as well as benefits in the ecosystem services approach" (Redford and Adams 2009).

The question that motivates this review is whether the ecosystem services concept is a useful way of organising *research* on the nature-society relationship. We critically review the literature on ES as an analytical concept, expressed in both biophysical and economic terms, but not so much the specific methods and estimates, nor the policy-oriented literature on PES. We begin by tracing the concept's origins and evolution, identifying major strands, and the extent and nature of convergence achieved thus far. We indicate why this formulation has attracted researchers, and summarise the resulting contributions that ES research has made to our understanding of the society-nature relationship. We then discuss the areas of confusion and other weaknesses in the concept, and trace these to a narrow framing of the environmental problem. We conclude by suggesting steps for moving towards a broader framing that might make ES research a more self-reflective and analytical exercise.

VERSIONS, CONVERGENCE, AND DIVERGENCE

The modern-day concept of ES has multiple origins and strands. While some convergence has emerged, there are important variations as well. Several reviews have focused on the confusing semantics in general (e.g., Fisher et al. 2009); we focus here on substantively distinct usages.

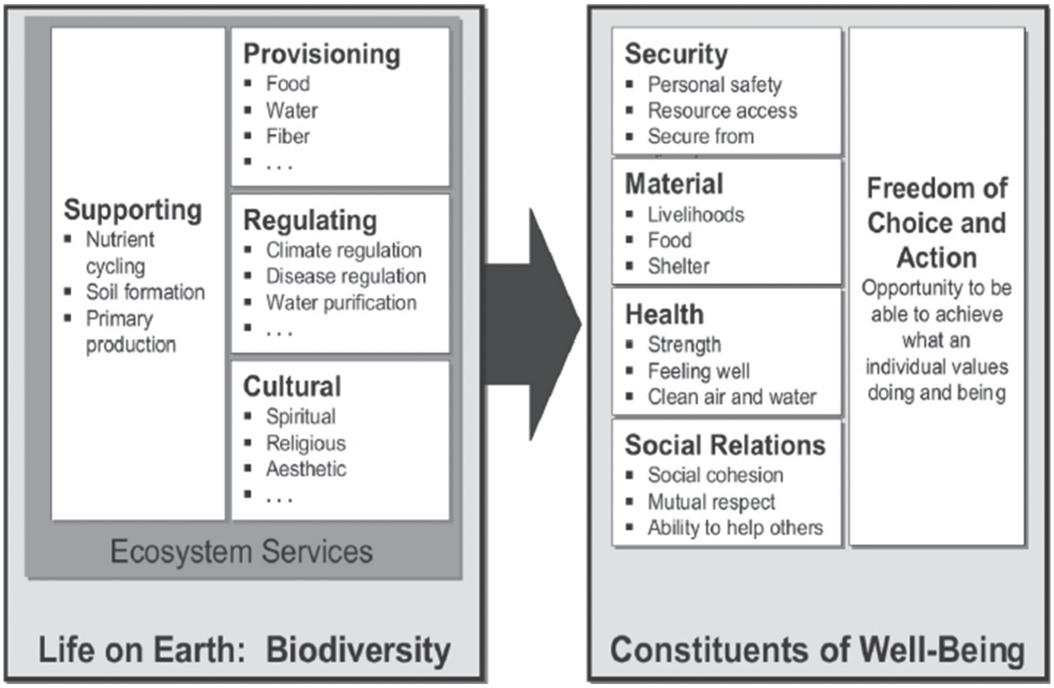
One version, developed by biologists, focused initially on 'life-support services', i.e., those features of the biotic environment that are seen as essential for the very survival of human beings on Earth (Ehrlich and Mooney 1983), the organisms one might take on a spaceship to create life-support systems on a lifeless planet (Daily 1997a). This approach then expanded to embrace all *indirect* benefits that human beings get from the functioning of ecosystems: soil conservation, water purification, waste assimilation, pollination, hydrological regulation, and so on [also called 'nature's services' (Westman 1977)]. Importantly, in this approach, ES-related benefits are seen as distinct from and *in addition to* the value of biodiversity conservation for its own sake (Balvanera et al. 2001). We label this the 'conservation biology approach.'[1]

A broader version, which developed in parallel, included all aspects of human dependence on the environment, and was driven by the concern that human actions leading to 'resource depletion, pollution, and extinction' could have significant negative consequences for human well-being (de Groot 1987). The idea of 'natural capital' (NC) emerged here and was developed by a group of environmental economists such as David Pearce and Ed Barbier and ecological economists such as Robert Costanza and Rudolf de Groot. In this version, NC is the stock that generates different kinds of benefit flows: products or goods, indirect benefits or services, and pure conservation (existence or aesthetic) values. We call this the 'environmental economics' approach.

An alternative to ES is 'environmental' services. This usage is ambiguous, sometimes used synonymously with ES, sometimes meant to emphasise the human contribution to ES (Pesche et al. 2012), but most often meant to highlight the abiotic elements in nature. For instance, rainfall, although originally classified as an ES (Swinton and Lupi 2005), is now re-cast as an environmental service (Scott Swinton pers. comm. 2011). An extreme usage is 'geosystem services' (and 'geodiversity'), said to represent the benefits from geological deposits (Gray 2011; Mace and Bateman 2011: p.6). We believe that this latter formulation is easily subsumed under the environmental economics approach.

From these variants, the Millennium Ecosystem Assessment (MA) crafted a framework that is close to the environmental economics approach, except that 1) it broadened the term 'services' to include products and existence values as 'provisioning services' and 'cultural services' respectively; 2) it limited the concept of natural capital to 'life on Earth' or 'biodiversity', thereby excluding purely abiotic resources such as minerals or abiotic energy sources; and 3) it introduced a confusing category called 'supporting services' that covers what were hitherto seen as functions (see Figure 1 below, reproduced from MA 2005). Subsequent assessments, most notably the UK National Ecosystem Assessment framework (Mace and Bateman 2011), have modified this framework somewhat while retaining the above core elements.

Common to all these approaches is an acceptance of the importance of economic valuation. Although the MA framework in Figure 1 does not necessarily imply an economic valuation of human well-being, the vast majority of social scientists associated with the MA are economists (Daily et al. 2009) and most of the assessment work being done under the banner of ES is economic in its practice (e.g., Sukhdev 2008) or at least in its stated intent (e.g., http://www.naturalcapitalproject.org). The ES concept has therefore become almost indistinguishable



Source: MA 2005

Figure 1
The Millennium Ecosystem Assessment (MA) framework

from the valuation of ecosystem services (VES).[2] The core premise is that ecosystems degrade because society knows neither the 'true extent' of these benefits (because some of them are indirect and thus ignored) nor their 'true value' (because some of them are not priced correctly).

KEY ATTRACTION AND CONTRIBUTIONS OF THE ES CONCEPT

Attraction

The idea that conventional economics undervalues the benefits provided by environmental processes has been around for a while. The concepts of 'ecosystem benefits,' 'environmental valuation,' and 'total economic value' (TEV) were developed as a response to this critique (Randall 1987) and have been in vogue in the environmental economics literature and even in the policy literature [e.g., greening of national accounts (Harrison 1989)] for several decades now. So what explains the explosion of interest in ES that one has witnessed in the past decade?

We believe the key attraction of ES lies in implications of the simultaneous broadening and narrowing that is best characterised by the MA framework. First, the suffixes 'services' and 'capital' come from economics and help ecologists to communicate with those conditioned to an economistic way of thinking. The suffixes shift the debate from a negative tone of economic development being bad for wildlife to a positive one of conservation being good for humans. Whereas ecologists were traditionally suspicious of economics (Vedeld 1994), the idea that economic valuation of ES can strengthen the case for biodiversity conservation, and payment systems can make conservation happen, has offered new hope in an era of market-based thinking (Norgaard 2010). Ecologists are now warming up to neoclassical environmental economics as never before.

Simultaneously, by characterising 'supporting services' as the basis for all other services (Figure 1) and by labelling some of these services as 'life-supporting', the ES approach takes a 'strong sustainability' position, i.e., it implicitly rejects the standard neoclassical economics argument that human-made capital can indefinitely substitute for natural capital. Indeed, the MA framework in Figure 1 does not mention human-made capital at all, although the later UK National Ecosystem Assessment (NEA) framework indicates 'other capital inputs' as complementing ES to co-produce benefits for people (Mace and Bateman 2011: 7). This represents an important shift in the discourse: the idea that life on Earth is fundamental to human well-being, and hence not to be compromised, is being accepted for the first time in wider circles. Furthermore, the MA framework equates 'life on Earth' with 'biodiversity' and suggests that biodiversity underpins all ES. This is particularly appealing to those who care about biodiversity conservation, which includes most ecologists.

Thus, a simultaneous 'economisation' of the conservation argument and an 'ecologisation' of environmental economics are the key attractions of this concept.

Main contributions

In making conservation biologists and ecologists think pragmatically in terms of the material benefits of conservation, the ES concept has generated a large body of ecological research on aspects that were hitherto underdeveloped, viz., indirect use values of ecosystems. In this lies its first major contribution. In the past, field ecology overlapped significantly with evolutionary biology and other streams, and the 'ultimate' variables of research interest emerged from within the discipline. Studies then focused on (say) net primary productivity (NPP) as a whole, nutrient cycling, patterns of diversity and community dynamics, or evolution of traits. Conservation biologists took the importance of species diversity (or endemism, etc.) as a given, and focused exclusively on studying its ebb and flow.

Ecologists are, however, now more willing to focus on variables that may be of direct relevance to human beings. While the literature from the more applied disciplines (forestry, fishery, range management, etc.) had always focused on the tangible goods harvested from ecosystems (provisioning services), the ES literature has contributed substantially in our understanding of regulating services. These include pollination, storm protection, nursery role, and pest control. In doing so, they have broken the logjam created by concepts such as ecosystem health that were rather self-referential, and focused on variables relevant to internal ecosystem processes, not to society (Georgina Mace pers. comm. 2012). We present a summary of this new literature in Table 1. Though not comprehensive, the table serves to indicate both the substantial interest in this area and also some of the complexities that have emerged.[3]

The second major contribution of the ES literature has been to expand the scale of analysis from local studies of individual ES to regional-scale models integrating all major ES. This has typically been done by integrating ecological models derived from individual studies into a spatial framework. A prominent example is the InVEST project (Kareiva et al. 2011). This approach differs from that of global studies such as Costanza et al. (1997) or Sutton and Costanza (2002) in that it is applied at a meso-scale whereby the biophysical models can be much more site-specific and can incorporate interactions between services.[4]

The third contribution has been increased collaboration between ecologists and economists. This has forced ecologists to think more carefully about which variables are socially relevant, even though the problem of treating ecological variables as inherently important has not gone away (as we argue below). It is also perhaps encouraging economists to engage with ecological complexity and to abandon their weak sustainability position, although the evidence of this happening is unclear.

These significant contributions notwithstanding, there are questions about the usefulness of the ES framework in promoting an improved understanding of the society-nature relationship. A closer examination of the concept as articulated reveals significant internal inconsistencies as well as substantive omissions and over-simplifications, which we now turn to.

Table 1
New literature on regulating services: an overview

Regulating service	Nature of service hypothesised	Important recent empirical studies/reviews	Qualifying remarks
Pollination	Forest islands provide habitat for insects that pollinate neighbouring agricultural crops	Klein et al. 2003; De Marco and Coelho 2004; Ricketts 2004; Ricketts et al. 2004; Olschewski et al. 2006; Ricketts et al. 2008: review article; Otieno et al. 2011	1. Estimating non-marginal impacts, i.e., complete disappearance of pollinators (e.g., Losey and Vaughan 2006) is unreliable 2. Risk of global pollination crisis might be exaggerated (Ghazoul 2005; but see also Kremen et al. 2008)
Pest control	Natural pest control is enhanced in complex patchy landscapes with a significant non-crop habitat	Bianchi et al. 2006: meta-analysis; Cleveland et al. 2006: insect-eating bats	1. Non-crop habitat may also harbour crop pests (Zhang et al. 2007; Otieno et al. 2011) 2. Pest control service from surrounding vegetation is not the same as benefits of on-farm integrated pest management (Macfadyen et al. 2009)
Storm protection	Mangrove/coastal vegetation provides protection against cyclonic storms and tsunamis	Das 2009; Bayas et al. 2011	1. Nature of vegetation may have less impact than its position and coverage (Bayas et al. 2011) 2. Vegetation may protect against storm surges, not against inundation, which requires different approaches (Feagin et al. 2010)
Nursery function	Coastal mangroves, coral reefs, and sea grass may act as nurseries for fish, thereby enhancing fish catch in the seas	Wilkinson et al. 1999; McClanahan et al. 2002; Heck et al. 2003; Manson et al. 2005	1. Nursery function is much more ambiguous than earlier economic valuations assumed 2. Declines in fish catch may be more due to overharvest than coral reef loss

CONFUSION: ECOSYSTEM FUNCTION AND BIODIVERSITY

There is significant confusion in the ES discourse about the idea of supporting services and inconsistency in how biodiversity is viewed.

Are all ecosystem processes of service?

There is a persistent tendency in the ES literature to treat processes internal to ecosystems synonymously with ecosystem 'functions' and ecosystem 'services'. This is clearly problematic, because it leads to either double counting or the counting of and comparison between variables at different levels. For instance, nutrient cycling is not a service; it is only a process that contributes to (say) timber production service. Valuing nutrient cycling in addition to timber would then lead to double counting (as in Maass et al. 2005). Similarly, pollination of forest plants, including that of economically useful plants, is a process that goes on within the forest ecosystem, but once the useful products have been valued, one should not value the pollination again. And studying the trade-off between an internal process such as litter decomposition and a benefit such as income (Steffan-Dewenter et al. 2007) is misleading, because the process underpins the benefit.

Is the mis-naming of internal processes as services simply a slip? If so, the slip has been pointed out and criticised several times (mostly by economists: Chomitz and Kumari 1998; Boyd and Banzhaf 2007) and should have been jettisoned long ago. But many researchers (mostly ecologists) continue to treat internal processes on par with final processes (e.g., nutrient cycling in Costanza et al. 1997; Balmford et al. 2002; Zhao et al. 2004; and pollination of natural flowering plants in Memmott et al. 2004). Even a recent review paper (Cardinale et al. 2012) lists soil organic matter and nutrient remineralisation as a regulating service. Although several senior ecologists agree that ecosystem functions are not services (Hal Mooney pers. comm. 2012), the MA institutionalised the confusion by terming these internal processes as 'supporting services.' The UK NEA did not entirely shed this confusion, as it invented an ambiguous usage 'ecosystem processes/intermediate services' (Mace and Bateman 2011: 6), apparently to retain a strategic link with the MA (Georgina Mace pers. comm. 2012).

We believe that this persistent confusion is a reflection of a deeper problem: the tendency to attribute purpose and therefore value to nature and all its processes as such. The word 'service' itself represents a subtle shift in thinking: instead of benefits that human beings *derive* from nature, we are now asked to think in terms of nature *providing* a service to human beings. This casts nature in the role of an active, purposeful agent.

But can we attribute such a purpose to nature, and would that purpose be enhancing human well-being? As philosopher John Searle has pointed out, in saying that the function of an animal's heart is to pump blood, biologists assume that the ultimate purpose of all the body's parts is the survival and reproduction of that animal (Searle 1995). Ecologists have extrapolated this idea to the scale of whole ecosystems. Thus, decomposition is seen as serving the purpose of making nutrients available for the next round of production in the ecosystem and predators as serving the purpose of keeping prey populations from exploding. Even though extreme ideas of grand design in nature or 'strong Gaia' are not popular amongst scientists (Kirchner 2002), the idea that each ecosystem component and process is likely to have some role to play and hence is essential to the survival of the whole is strongly held. But neither is extreme functionalism supported by evidence, nor is the survival of the whole biosphere necessarily equivalent to sustaining human well-being.

The ES concept, by explicitly linking ecosystem processes to an external goal, viz., human well-being, has the potential to go beyond the circular thinking[5] that bedevilled earlier literature on ecosystem health and ecosystem integrity. And as our brief review in the '*Main contributions*' section above indicated, this potential is being realised. But for this to happen consistently, the concept of supporting/intermediate services must be explicitly dropped. It is better to think of ecosystem services as only those stock or flow variables that are socially valuable and treating the rest as ecosystem processes having no intrinsic value. While the grouping of ecosystem processes into 'functional' categories (production, predation, decomposition, etc.) may still be useful, the relevant categories may change when social valuations of the ecosystem change: natural predators may be seen as pests or regulators depending upon the context.

Is biodiversity an ecosystem service?

The role of biodiversity is another confusing aspect of the current ES discourse. Is it the foundation of all ES, as the MA framework suggests? Or is it one of the ES itself, something that directly adds to human well-being? Or should it be seen as both an enabler of ES as well as having 'intrinsic value' beyond human well-being (Díaz et al. 2006)? But then isn't intrinsic value the same as existence value and thereby included in cultural services under the expanded notion of service created by the MA (Mace et al. 2012)? If so, why do most studies still talk of 'biodiversity *and* ecosystem services' as if they were two distinct (even if operationally overlapping) goals of environmental management (see Balvanera et al. 2001; Daily 2001; Singh 2002; Chan et al. 2006; Mertz et al. 2007; Martínez et al. 2009)? One may dismiss this as another semantic inconsistency in a still-evolving discourse. But we believe these differences reflect deeper confusion that needs to be resolved if the ES concept is to be useful as a scientific framework.

At the outset, it is important to distinguish between the scientific debate on the relationship *between* biodiversity and ES, and the conceptual question of whether 'biodiversity *and* ecosystem services' are distinct goals of environmental management. The former debate was long hampered by confusion between ecosystem functioning (EF) and ES, including in review papers such as Hooper et al. (2005).[6]

As mentioned earlier, EF is measured in terms of ecosystem processes such as rate of capture of energy, nutrients, and water in mostly natural ecosystems that ecologists consider key for natural ecosystem survival, and although the relationship of EF with biodiversity is generally positive (but see Ghilarov 2000), the usefulness of this information for environmental management is clearly limited. The literature that focuses explicitly on the relationship between biodiversity and provisioning and regulating services acknowledges a more complicated and mixed relationship (Cardinale et al. 2012; see also comments in Table 1). This seems rather obvious in hindsight (see the section '*Omissions…*' below), but in any case it raises questions about the term's scientific usefulness: if biodiversity as an ecosystem attribute term is too gross, maybe it should be jettisoned in favour of nuanced and precise terms: agrodiversity versus wild diversity, productive plant diversity versus weed diversity, and presence/absence of particular species rather than diversity per se.

The reason, however, biodiversity continues to be the preferred term is the normative attraction it holds, which relates to the conceptual debate. The treatment of 'biodiversity' and 'ecosystem services' as parallel goals occurs in the conservation biology strand of ES. It stems from the normative position that most conservation biologists and many ecologists hold, viz., that nature has intrinsic value (Vedeld 1994), that the conservation of all biodiversity is an ethical imperative not really compatible with the anthropocentric language of services (McCauley 2006). While they may use the ES metaphor to show (anthropocentric) policy-makers that biodiversity conservation can also generate some material benefits, thereby strengthening the possibility of conservation, they have not abandoned their original (bio-centric) intrinsic value position. This is clear from statements like "it is going to be a long haul for biodiversity for its own sake… Ecosystem services is a strategy to buy time as well as getting buy-in" (Gretchen Daily, quoted in Marris 2009). Or, as Skroch and López-Hoffman (2010) openly stated, "although we understand ecosystem services provide a larger audience and more resources for conservation, it is now our responsibility to ensure that these new tools are used in ways that we intended; namely, to protect the diversity of life on Earth."

The problem here is that if the outcome to be achieved, viz., protecting all diversity for its own sake, is already a given, ES assessments become advocacy tools, not scientific exercises. Instead of an open-ended exploration about what consequences a particular decision to convert or modify an ecosystem might have for multiple societal values (in which biodiversity is one), it becomes a pre-determined and instrumental use of utilitarian arguments to increase support for biodiversity conserving outcomes, which are considered the 'right' outcomes anyway. While all applied science has to speak to societal values and in that sense can never be value-free, reducing what is valuable to a single goal is going too far, especially when it is the researcher's personal value rather than the only value society holds. Such unselfconscious practice will often produce bad science.[7] Moreover, this approach can lead to double standards: if it turns out that the material benefits from biodiversity conservation are lower than those from its destruction (say for a mine), then conservationists are tempted to invoke the argument that biodiversity is not valuable only for the (anthropocentric) services it produces, but also for its own sake, and this intrinsic value is immeasurable.[8]

The environmental economics approach, also adopted in the MA, is a more consistent one. The intrinsic value of biodiversity is subsumed under 'cultural services,' which includes cultural, religious, and aesthetic values provided by ecosystems. This follows the earlier economic formulation, wherein TEV included non-use values such as option, bequest, and existence values (Randall 1991).[9] In other words, all the different ways in which society values the environment is included in the idea of value, now recast as service or well-being. Biodiversity does not sit outside the pale. In theory, if human well-being (after factoring in all ES) were somehow found to be higher in a situation where biodiversity declines, that situation would be preferred.

Not surprisingly, conservation biologists and conservationists in general have great difficulty in accepting this approach. Bringing intrinsic value within the ambit of cultural services would mean that, even if not traded-off in monetary terms, conservation goals would have to be negotiated with other societal goals, other dimensions of well-being. While advocacy groups may legitimately have their priorities, the problem becomes serious when conservation-minded biologists and ecologists shy away from accepting this as the way society should work, and look for other forms of resolution.

The pragmatic resolution is not tenable: that conservation of key taxa is congruent with management that sustains all material services is not convincing (Mace et al. 2012). Many ecosystem services may require a reduction in biodiversity (Reyers et al. 2012) and many kinds of diversity will never have any utilitarian value (Gadgil 1998), not to mention the negative value (pestilence and disease) generated by many of them. And the philosophical resolution is still unpalatable. Reyers et al. (2012) argue that even though existence value is still a type of instrumental value, it comes close enough to intrinsic value that conservationists need not worry about the outcomes of conservation policies motivated by the MA-type (apparently) instrumental thinking. Logically, however, all values are anthropogenic, i.e., products of the human mind, even if they are not anthropocentric, i.e., narrowly focused on the material needs of human beings alone (Hayward 1998). If intrinsic value draws upon some ethical principle, then it is not substantively different from religious or existence value, nor on a different plane than environmental equity and social justice, sustainability of future generations, or other ethical concerns, or even material values, since all are social constructs.

We suggest that accepting all values are anthropogenic provides a starting point for a more relevant, culturally sensitive, and open-ended scientific analysis. Instead of putting biodiversity (because of its supposed intrinsic value) on a pedestal, conservation biologists would do well to unpack the real values embedded in 'biodiversity conservation' as a goal.

They may then find that the biodiversity ordinary people care about is charismatic taxa or mega-fauna, or charismatic places, not the entire spectrum of life on Earth (Mace et al. 2012). But instead of treating this as a sign of public ignorance about the importance of all taxa, they should treat this as one way in which society cares about the environment, to be understood and added to other values.

OMISSIONS: THE RELATIONSHIP BETWEEN NATURE AND WELL-BEING

At another level, the ES framework contains a series of oversimplifications and omissions about the relationship between nature and human well-being. The way the nature-society relationship is depicted in Figure 1 suggests a simple positive relationship: more natural capital always leads to more ecosystem services and thereby to more human well-being. Moreover, since natural capital is now equated with 'life on Earth,' the implication is that purely abiotic resources (those that are not generally part of ecosystem processes: fossil fuels, ores, nuclear energy) do not contribute to well-being. Finally, since no human inputs are shown, it also suggests that nature automatically provides these services. But all these omissions are highly problematic.

Dis-services

The relationship between nature and society is not all positive. Nature imposes several kinds of hardships on human beings. Biotically, pests and diseases are obvious examples. Abiotically, rain brings both life-giving water but also life-threatening floods. Human history is a history of a constant struggle to adapt to this munificent-cum-hostile nature. But by focusing on services, a term with positive connotations, the ES discourse automatically conceals the hostile side of the relationship. Some ecologists have begun to question this omission, pointing to 'dis-services', such as pathogens (Willott 2004; O'Farrell et al. 2007; Dunn 2010). Another example would be wildlife, which affects local human populations negatively through crop damage by large herbivores, predation on livestock by carnivores, and direct injury to and loss of human lives. Data on dis-services from tigers, elephants, and snakes in India, given in Table 2, indicate very significant figures of damages and deaths. Clearly, increases in natural capital do not always result in increases in well-being.

It may be argued that the MA framework does not put any sign on the arrows in Figure 1, and so it allows for a positive or negative relationship. But neither the MA report nor any of the hundreds of ES assessments that followed contain any mention of negative relationships. The problem precedes the MA or the ES literature: the literature on TEV also mentions only positive benefits. As the literature on problem framing points out, particular terms predispose our thinking in particular ways (Bardwell 1991), and the positive connotation of the words 'service' and 'benefit' predisposes the ES discourse towards focusing on positive relationships only.

Trade-offs between services

In addition to dis-services, there are trade-offs between services themselves. Natural capital is not in fact a homogeneous entity as the metaphor might suggest. The same ecosystem may relate to human well-being in multiple ways, and in a given context, some services may increase at the cost of others. For instance, a tropical forest may provide timber, firewood, and fodder as well as regulatory services of carbon sequestration or soil conservation, and cultural services such as wildlife. But there are clear trade-offs: increasing carbon sequestration may result

Table 2
Wildlife-related dis-services to neighbouring human populations in India

Dis-service	Study area	Impact (economic losses or number of people affected)	Time unit	Reference
Crop damage due to wildlife	Four southern states of India	INR 6.5 million	1981–1983	Sukumar 1989
	Sariska Tiger Reserve, Rajasthan	INR 3,300/household (average)	Annually, between 1996–1997	Sekhar 1998
Loss of livestock	Kibber Wildlife Sanctuary, Himachal Pradesh	18% of the total livestock of families around sanctuary; economic loss of 12% of income	1995	Mishra 1997
Loss of lives to elephant attacks	South India	30–50 persons	Annually	Sukumar 1991
	West Bengal, Uttar Pradesh, and Assam	115–160 persons	Annually	Sukumar 1991
	India	300 persons	Annually	Bist 2002
Loss of lives to tiger attacks	Sundarbans National Park, West Bengal	57 persons (average)	Annually, between 1975–1984	Khan 1987; Sanyal 1987
Loss of lives to snake bites	Asia	100,000 persons	Annually	Chippaux 1998; Sharma et al. 2004; Kasturiratne et al. 2008;
	India	15,000–50,000 persons	Annually	Meenatchisundaram and Michael 2009

in lower biodiversity as well as reduced harvest of timber, and maximising timber production will reduce the available fodder, firewood, and biodiversity. Thus, the relationship between forest ecosystem states and the flows of different benefits is better represented in the form of a matrix, such as in Table 3.

Again, these trade-offs are well known. For instance, a significant part of the literature in tropical forestry has been about conflicts generated by competing systems of forest management (e.g., Guha 1985), even if the term trade-offs was not used. But the ES discourse has only emphasised 'win-win' situations. If at all, only extreme trade-offs are mentioned, as between commercial agricultural production and biodiversity or water quality (e.g., Nelson et al. 2009). Only recently have ecologists begun to examine the whole range of trade-offs (Rodríguez et al. 2006; Swallow et al. 2009; Raudsepp-Hearne et al. 2010). We argue that any reasonable framing of the society-nature relationship would a priori assume that dis-services, trade-offs, and synergies all exist, and would focus on understanding their nature and extent in specific circumstances. In other words, the simple arrows in the middle of Figure 1 should be replaced by something like the matrix of Table 3, including columns for dis-services.

The role of abiotic resources

Another omission is that of abiotic resources. Early formulations of natural capital also included mineral and water stocks—entities whose presence is only mildly influenced by biota. The 'life on Earth' formulation of natural capital as per the MA would then exclude such abiotic stocks and flows from the ambit of services, but as mentioned earlier in the section '*Versions, convergence, and divergence*' some analysts include rainfall and mineral deposits in the category of environmental services. In either case, the real issue is that the relationship between abiotic resources and ecosystems is not always additive, but often highly competitive.

A core feature of the nature-society relationship is that human beings, through technological innovation, have figured out a number of ways of using abiotic nature to *replace* many benefits provided by biotic nature. From the use of hydropower, petroleum, coal or nuclear energy as replacements for firewood, to using iron, aluminium, and cement to replace timber, nylon for clothing, and petrochemical-based fertilisers to replace organic manure, abiotic resources have rapidly expanded and *directly replaced* ES derived from biota. Statements such as "human societies have been built on biodiversity" (Díaz et al. 2006) do not tell the whole story. Modern societies are disinterested in biotic nature because they see a much smaller dependence on it than earlier societies did.

Moreover, the use of abiotic resources does not simply reduce the use of biotic resources (and thereby the incentive to conserve them), but often actually undermines other biotic services or creates dis-services. Mining destroys forests, petroleum spills destroy fisheries and marine life, industrial manufacturing generates pollutants that impair human health, and of course the burning of fossil fuels leads to global warming that threatens many aspects of well-being. The current framework does not enable us to engage with the dis-services of climate change through fossil fuel use.

It is therefore necessary to include minerals and other abiotic processes in the framework, but it is equally important not to club them with biotic processes. They could be shown as additional land-uses (rows) in Table 3, along with the addition of columns such as pollution. The trade-offs between the services derived from biotic and abiotic nature and the possibly short-term nature of abiotic resources are in fact the central challenges in planning for conservation and sustainability.

Co-production

Obtaining benefits from ecosystem processes usually requires the investment of human labour and human-made capital for harnessing the 'service'. Plants or animals do not automatically generate provisioning services; they (or their parts) have to be

Table 3

Trade-offs between different benefits (and beneficiaries) of forest ecosystem services under different land-use scenarios

Land-use type	Product, service or benefit							
	Local beneficiaries			Regional beneficiaries			Global beneficiaries	
	Fuelwood	Fodder	NTFPs	Timber	Hydrological regulation	Soil conservation	Biodiversity	Standing carbon
Forest								
Dense 'natural' forest	++	0	+++	0	+++	+++	+++	+++
Dense lopped forest	+++	+	+++	+	++?	++	++	++
Open tree savanna	++	++	+	0	+?	++	+	+
Pure grassland	0	+++	0	0	+++?	++	+?	+
Timber plantation	+	0	0	+++	+/-?	+	0	++
Non-forest								
Coffee plantation	+	0	0	+	++?	++?	++	++
Terraced paddy	0	++	0	0	+?	+?	?	0
Slope (dry crop) cultivation	0	+	0	0	0?	-	?	0
Barren land	0	0	0	0	0	-	0	0

Source: Lélé 1994; Notes: Plus/minus signs indicate extent of positive/negative benefits. The signs represent physical benefits, and so are comparable only within a column, not across columns. Question marks indicate significant uncertainty about direction or magnitude of benefits; NTFPs=Non-timber forest products gathered, harvested, or hunted through human labour. Water flows become useful only when the water is lifted, diverted, or stored using various structures or technologies. Rain-fed agriculture requires field bunds to capture rainwater and planting the right crops at the right time to benefit from the rain. Even 'existence' values are not really obtained without someone keeping us informed about the status of the species we cherish! Indeed, the same ecosystem process (soil erosion by streams) can generate a dis-service (siltation of dams) or a service (fertilisation of the floodplain) (Lele 2009), suggesting that ecosystem processes get value (positive or negative) only within specific human contexts and engagements. The omission of human agency in the form of labour and capital from the MA diagram is as problematic as the omission of energy, material, and ecosystem service flows from the circular relationship between economic goods and services depicted in conventional economics textbooks. Even the UK NEA (Mace and Bateman 2011: 7) introduces 'other capital inputs' only at the link between food production and cereals, when in fact they are essential to production itself.[10]

A more accurate picture of the relationship would then depict human capital and labour as co-producers of benefits from ecosystem processes. That is, the expanded matrix of Table 3 would be embedded in a socio-technical context. The focus of ES assessments would have to expand from simply estimating the number in the cell to investigating how labour, technologies, financial capital, and institutions interact with ecosystem processes to produce those numbers. The results of such expansion could be interesting. For instance, a recent study of hydrological regulation service of forests found that even the *sign* of the impact of forest cover change on irrigated agriculture depended upon the nature of the irrigation technology (Lélé et al. 2008; Lele et al. 2011). Similarly, the physical magnitude and economic value of provisioning services fluctuated significantly depending upon the rights and institutions for their marketing (Lele and Srinivasan 2012).

Furthermore, if the value of an ecosystem service cannot be separated from its socio-technical context, then the usefulness of mapping ecosystem services (e.g., Naidoo et al. 2008) and validity of benefit transfer (a euphemism for extrapolating values from one location to another) is seriously in question. A much more place-based approach is called for, than a focus on 'global' assessments.

THE MISUSE AND LIMITS OF ECONOMIC VALUATION

Much of the ES literature tacitly or explicitly accepts an economic valuation framework for assessing human well-being. This raises two kinds of concerns. First, possibly because the ES literature has been driven by conservation biologists more than economists, there have been some glaring mis-applications of conventional (neoclassical) economic methods. Second, all the limitations of the neoclassical idea of economic valuation have been carried over uncritically into this literature.

The mis-applications of conventional economic valuation in the ES literature have been pointed out in many reviews (Bockstael et al. 2000; Turner et al. 2003; Lele 2009). The major ones are: estimating absolute value rather than marginal change in value, generating global scale estimates from local scale studies, and double-counting between supporting and final services. We shall, however, focus here on the more fundamental critiques of the concept of economic valuation and benefit-cost analysis.[11]

First, it may be not just impossible but ethically quite objectionable to put a monetary value on things that have intrinsic value. The problem is not that the price put on these things may be too low, it is simply that one is putting a price (McCauley 2006). Of course, this is true not just for the intrinsic value of non-humans, but also of human lives, and possibly of basic human needs, social justice, or other ethical goals.

Second, the conventional approach of estimating (changes in) economic welfare involves simply adding up benefits and costs across all individuals, regardless of the difference in their wealth. But simple aggregation does not pass the 'laugh test' (Farrow 1998), i.e., laypersons find it laughable that one dollar more to a rich person is considered as important as a dollar more to a poor person. The problem is particularly acute in the ES context, because trade-offs between services are eventually trade-offs between different beneficiary groups (see the column groupings in Table 3), and these groups are often dramatically different in their wealth status. For instance, carbon sequestration benefits from tree planting accrue to the whole world, but the opportunity costs imposed by such carbon plantations may be borne by poor firewood collectors in the tropics. The well-being box in the MA framework is separated into different types of well-being (material, health, security, etc.) but remains an 'aggregate' human well-being, without even a simple poor-rich or stakeholder-wise distinction. Consequently, an analysis of the distribution of gains and even possible trade-offs between such strata due to changes in ecosystem management is entirely missing from the ES literature (Daw et al. 2011).[12]

Third, the conventional approach also aggregates across generations, and does so in a biased manner by the use of a positive (and usually significant) discount rate by which the well-being of future generations is given much lesser weight. Again, environmental impacts of development projects may have long-term implications compared to the short-term material gains from them, and so the use of positive discount rates becomes particularly problematic (see Sáez and Requena 2007 for a comprehensive review).

Finally, several ecological economists (Vatn 2005) and political philosophers (Taylor 1992; Sagoff 1998) have argued that public decisions about environmental problems are qualitatively different from choices made by individualistic consumers about commodities, precisely because environmental goods have the characteristics of common-pool goods or merit goods. Therefore even extended benefit-cost analysis (BCA) is an inappropriate tool for decision-making as it still adopts a utilitarian ethic and reduces the answer to a single number. Instead, deliberative decision-making approaches are recommended. The UK NEA has taken a step in this direction by allowing value to be represented in multiple ways: monetary, rank, and qualitative (Mace and Bateman 2011).

THE UNDERLYING PROBLEM FRAME: TECHNO-ECONOMIC OR POLITICO-CULTURAL?

Taken together, the above lacunae suggest that there is a fundamental limitation to the way the environmental problem has been framed by the proponents of ES. The problem is cast as a case of the Earth's life-support systems being in jeopardy, which in turn jeopardises all human well-being on spaceship Earth, and this is said to be caused by decision-makers under-valuing the contribution of ecosystems in providing human well-being (Daily 1997a: 6; MA 2005). But this formulation is quite limited.

First, the characterisation of the environmental crisis and therefore the ethical underpinnings of the framework are narrow. For the conservation biologist, the crisis is that of declining biodiversity, an ethical violation. For the economist, it is of declining *aggregate* human well-being, however conceived; in other words, a sustainability[13] crisis. The alliance between the two has produced a confusing framework where, as indicated above, sustaining aggregate human well-being is the ostensible goal, but biodiversity conservation looms large in the sub-text.

But environmental problems may also arise due to actions of one group of human beings diminishing the well-being of another group of human beings right away, what economists call 'negative externality.' Similarly, communities or individuals suffer environmental problems in the form of denial of access to key environmental resources—water, land, pastures, or forests—on which their survival may depend. The ethical concern in both cases (downstream impacts or unequal access) is not rights of other species or sustainability of human well-being but intra-generational equity and environmental justice for humans. Similarly, the developmental problem is not simply one of increasing aggregate well-being, but also of increasing fairness and social justice.

Second, the analysis of *why* these multiple problems exist needs to be expanded. The ES framing emphasises under-valuation or non-recognition in decision-making circles, thereby privileging economics and ecology as the solutions. But a broader framing of the problem also leads us to a wider set of potential causes, drawing upon other theoretical perspectives (Robbins et al. 2010). First, political economy and political ecology tell us that in a world where externalities abound and power differentials (of class, caste, race, and gender) are often the norm, the problem may not be that benevolent or objective policy makers are not *aware of* environmental degradation; often, they are captive to or pressurised by powerful interests that are the polluters or the resource appropriators (e.g., Agarwal 1985; Forsyth 2003; Sabatier 2007). Pollution externalities continue to be imposed not because their impacts are under-valued but because the pollutees are powerless.[14] Second, psychological and cultural analysis tells us that people may not in fact hold environmental values (Dietz et al. 2005) and so their preferences, revealed through any number of valuation exercises, may not result in a high enough figure of willingness-to-pay to outweigh the economic gains from ecosystem destruction. Third, people may in fact discount the future heavily for different reasons—desperation due to poverty or conversely individualistic greed and selfishness.

Fourth, institutional failures may result in open-access situations that lead to resource degradation. Fifth, the power of reductionist science and high technology is such that it indeed reduces dependence on ecosystem services, by increasingly substituting them with abiotic resources, and creating at least a short-term increase in human well-being, at least for those who can afford the technology. And these causes may interact. For instance, capitalist structures also play a role in fostering cultures of consumption and selfishness, and in creating myths about technology, and the technologies also lead to accumulation of power and challenge institutions that seek to promote the common good. The idea that environmental problems have multiple causes is not new (Petak 1980), although there has been a tendency for different social science disciplines or perspectives to privilege their own explanation to the neglect of others (Vayda and Walters 1999; Lélé 2008).

In this situation, information on the 'aggregate value of ecosystem services' is likely to play a limited and even distorted role, given the problems and biases in estimating aggregate value mentioned above. This approach tends towards a techno-economic expert-centric process of social change. This also explains why many social science disciplines or perspectives related to the environment, such as political ecology, environmental sociology, ecological anthropology, or human geography have not engaged with the ecosystem services concept; the literature has been largely dominated by economists.

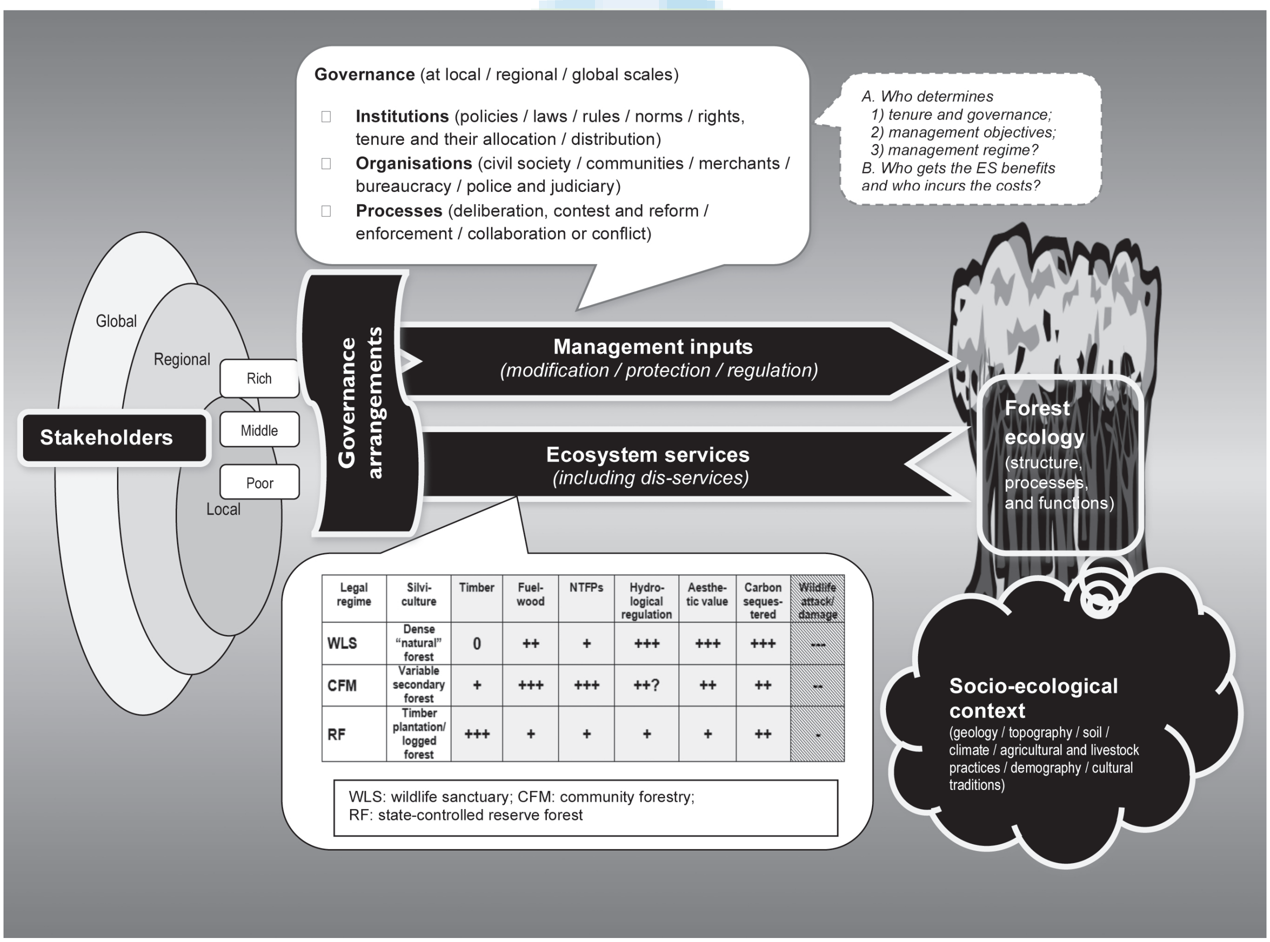
TOWARDS AN ALTERNATIVE APPROACH

The central question in this review has been whether and to what extent the ES concept, as currently framed, is a useful way of framing research into the society-nature relationship. We have shown that a privileging of biotic nature and a tactical use of economic valuation in response to a neo-liberal policy climate have attracted more ecologists to the cause. Significant insights, particularly about indirect or regulating services, have emerged as a result. Nevertheless, we find the framework lacking on multiple dimensions. First, the pragmatic use of a utilitarian ethic does not sit well with a deeper allegiance to bio-centrism. Second, a keenness to make a positive case for biotic nature results in a series of omissions and oversimplifications that threaten the credibility of the science. Third, an economic valuation framework results in highly reductionist analysis about changes in societal well-being. These lacunae seem to be based upon a narrow characterisation of the environmental problem and its causes. Where does one go from here? Limitations of space prevent us from articulating the alternative in detail; instead, we indicate the initial steps required and then briefly describe a framework we developed specifically for an ongoing field study on tropical forest ecosystems.

Moving towards an alternative approach involves several steps. First, it requires greater self-reflection in handling questions of values in applied environmental research (Lélé and Norgaard 1996). On the one hand, the conservation biology strand needs to abandon their deeply held belief that the intrinsic value of biodiversity is non-negotiable and accept that society may value different aspects of biotic nature for different reasons that all need to be understood and heard in the decision-making process. On the other hand, the environmental economics stream needs to reject the absolutism of valuation and BCA. Although "in a democratic society, values used in social decision-making ought to be derived from those held by its individual citizens" (Daily et al. 2000), it does not follow that the values need to be expressed in monetary terms alone nor that BCA needs to be the aggregation procedure: democratic decision-making has a different ethos.

This is not to be mistaken as a call for either an artificial separation of the science-policy (or fact-value) domains nor a retreat into extreme cultural relativism about values. Instead, we suggest that a systematic analysis of the normative underpinnings of different shades of environmental and developmental concerns articulated in real-world situations will show a broad consensus around a multi-dimensional notion of societal well-being that goes beyond the MA framework in important ways: quality of life, sustainability, equity, inter- and intra-general justice, justice to other species, democracy, and so on (e.g., Brechin et al. 2002; Joy et al. 2006; Menon et al. 2007: 21).

Second, ecosystem service analysts must move away from thinking of ES assessment as a decision-making tool and treat it more as a framework for understanding and analysing the nature-society relationship. The task of the analyst is to 'analyse', not to aggregate and give *the* answer (Bromley 1990). For instance, instead of putting aggregate monetary values on the pest control service provided by forest fragments to adjacent farms, it would be useful to know who the adjacent farmers are, why they have not converted the forest fragments to agriculture, how pest control benefits are distributed across different social categories, how knowledge of pest control services is distributed across social categories, how or which farming practices actually enhance the service, what forces prevent farmers from adopting them, and so on. This may also lead to questions of what is driving agrarian change, and influencing crop choice and forest



Legal regime	Silvi-culture	Timber	Fuel-wood	NTFPs	Hydro-logical regulation	Aesthe-tic value	Carbon seques-tered	Wildlife attack/ damage
WLS	Dense "natural" forest	0	++	+	+++	+++	+++	---
CFM	Variable secondary forest	+	+++	+++	++?	++	++	--
RF	Timber plantation/ logged forest	+++	+	+	+	+	++	-

Figure 2
Extended framework for ES analysis: applied to forests in eastern India

clearing decisions (which may often be unconnected to a specific service). Thus, a far more useful investigation is possible if one gets out of the straitjacket of assessment and valuation. Exploring the causes of environmental change from multiple disciplinary perspectives will further enrich the investigation.

Third, when carrying out such investigation, more reflection is needed about what is included or excluded. Again, completely objective models are impossible (Lélé and Norgaard 1996). But while we (the authors) also believe that some elements of biotic nature are essential for long-term human well-being and that a big part of the environmental crisis is the result of a misplaced faith in abiotic technologies, the issues are complex enough that both biotic and abiotic resource use need to be represented and both services and dis-services of either use need to be considered, as well as trade-offs between services. Scenarios considered must realistically include what uses society is considering, not just those that the analyst considers.

As a brief illustration, we present in Figure 2 a framework we are currently using to investigate how governance systems influence the nature, magnitude, and distribution of ES in eastern Indian forests. We have selected multiple services, including one dis-service in the form of damage by wildlife to crops and human life. Stakeholders are identified at multiple scales and in terms of multiple economic and social classes. Existence value of biota is sought to be captured through various measures of visitors, time spent, and sacredness ranking for special sites. We then ask how variations in forest rights and governance might shape the distribution of benefits from different ecosystem services as also the long-term sustainability of the forest. Investigating the structure of markets for non-timber forest products and the distribution of returns along the value chain helps us understand the role of other factors such as social organisation and knowledge in shaping livelihoods of forest-dependent communities.

Several limitations persist, of course. For instance, the alternative scenarios we consider (the rows in the matrix at the bottom of Figure 2) are all 'biotic' as we do not consider mining as a likely land-use option in our study area. We are also assuming that the alternative silvicultural options are all potentially sustainable in their own way, and that future generations will continue to want the tangible forest products. Nevertheless, this kind of framework enables us to analyse how different stakeholders might perceive their relationship with forests, how benefits and costs are distributed, and how they may influence forest governance and its ecological outcomes.

In conclusion, the idea of ES was coined as a way of combating a perceived blindness of policy-makers to the importance of biotic nature. But convincing policy makers to change presumes that one knows what change is required. This makes ES a policy advocacy tool. Conceptual completeness and consistency are then not critical. If, however, it is to be a framework for scientific enquiry, then it has to be much more consistent in its philosophical framework, inclusive in its normative concerns, open-minded about how biotic and abiotic stocks and processes may or may not produce well-being, and inclusive in the social science perspectives it invokes. We believe that such an approach, albeit more challenging to implement, will yield dividends in the long run in terms of more nuanced insights and more usable knowledge.

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NOTES

1. While this is a broad label, and while several individuals associated with this approach early on have moved towards broader frameworks, we will show below that some core distinctions remain.
2. For instance, even when governance is mentioned, it is as an intermediate variable that influences economic value (see Figure 1 in Turner and Daily 2008).
3. We have specifically omitted the literature on hydrological services from this table, as its contribution is more ambiguous (Lele 2009).
4. Of course, to the extent that, in many parts of the world, knowledge of individual ESs and how they respond to ecological and social change is largely missing, the importance given to meso-scale modeling may have to be tempered.
5. Natural ecosystems are healthy by definition because health is measured in terms almost inseparable from naturalness (Lackey 2001).
6. We thank Georgina Mace for pointing this out.
7. E.g., Olschewski et al. (2006) found that even after factoring in the pollination service provided by forest fragments to surrounding coffee plantations, it would be more economically rational for the farmer to convert the forest plot to agriculture or pasture. Nevertheless, they continued to look for ways in which the forest fragments would remain unconverted, rather than bow to their own economic logic. More examples of unconscious value judgements impairing the science are given in Lélé and Norgaard (1996).
8. Based on discussions with representatives of conservation groups in a workshop in Cambridge University, April 2012.
9. We disagree with Turner et al. (1994) and Brown (1994) who assert that existence value is anthropocentric and quantifiable whereas intrinsic value is not.
10. Johnston and Russell (2011) insist that only those outputs of ecosystems "*prior* to any combination with human labor, capital or technology" should be called services (author emphasis). We would argue that no physical process becomes important without human beings providing the context. To use their own example, if harvested fish are a benefit and fish in the lake are the service, the fish in the lake do not become a service if there are no fisherman going around trying to catch them.
11. See Wegner and Pascual (2011) for a more detailed exposition.
12. Some exceptions are: Lélé et al. (2001) and van Beukering et al. (2003).
13. Strictly speaking, a problem of inter-temporal inefficiency, which is a very weak version of sustainability (Kerr and Swarup 1997).

14. Here, power is used not just in material terms but also to frame decision-making in terms of one-dollar-one-vote through economic valuation rather than a more democratic one-person-one-vote.

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