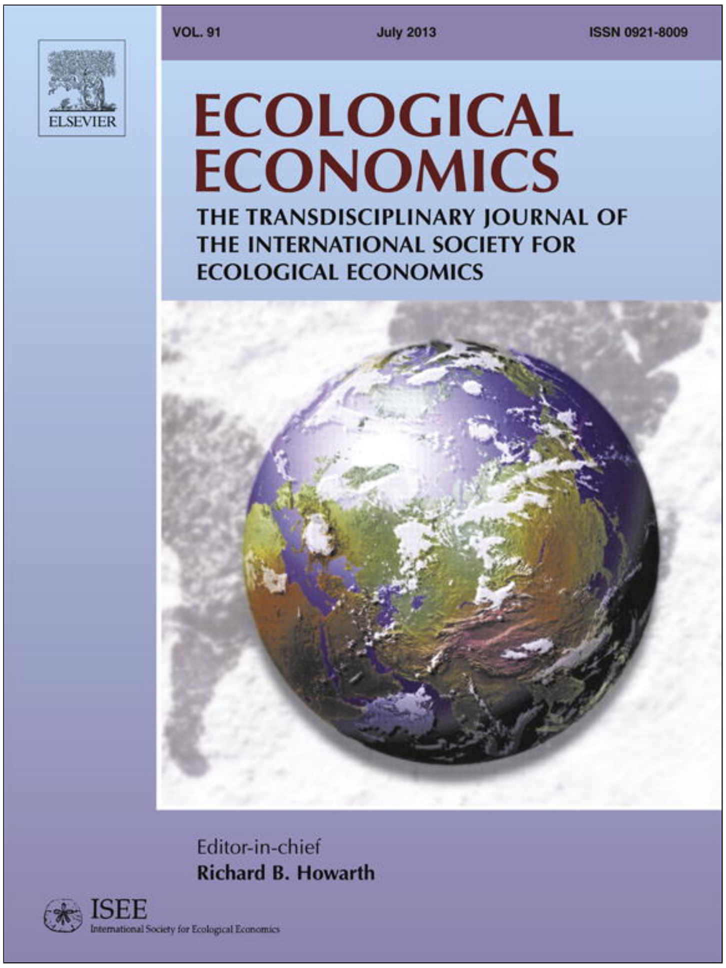
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Methodological and Ideological Options

# Disaggregated economic impact analysis incorporating ecological and social trade-offs and techno-institutional context: A case from the Western Ghats of India

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ABSTRACT

Economic valuation of ecosystem benefits and their aggregation in a benefit–cost analysis (BCA) framework is the norm in mainstream environmental economics. But valuation and BCA have also attracted criticisms. 'Internal' criticisms point to the absence of alternative scenarios in valuation, overlooking of ecological trade-offs and dis-services, and inattention to context. Others criticize aggregation across diverse stakeholders and the problem of non-monetizable benefits, and dismiss BCA as fatally flawed. They suggest approaches such as deliberative decision-making and multi-criteria analysis. We propose a middle path that uses the strengths of economic analysis for decision support while avoiding the pitfalls. We disaggregate economic impacts by stakeholder groups, link ecosystem changes to benefits as well as dis-benefits, and examine how socio-technological context shapes the magnitude of economic impact. We illustrate this approach by studying the impact of creating the Biligiri Rangaswamy Temple wildlife sanctuary in the Western Ghats forests of southern India. Our analysis shows that while some stakeholders are net beneficiaries, others are net losers. Changes in forest rights, irrigation technologies, and ecosystem dynamics influence the magnitude of benefits and sometimes convert gainers into losers. Such disaggregated analysis can provide useful information for deliberative decision-making and important academic insights on how economic value is generated.


## 1. Introduction

Environmental economists have long believed that economic valuation is the best way to estimate the societal importance of an environmental good, that conventional valuation[1] methods can be extended to generate the 'total economic value' (TEV) of ecosystems (Randall, 1987), and that incorporating these values into an extended benefit–cost analysis is the best approach to decision-making (Dixon and Hufschmidt, 1986; Pearce et al., 1988). In recent years, many ecologists have accepted the economic valuation framework for highlighting the importance of 'ecosystem services' and extended benefit–cost analysis as the 'rational' tool for making decisions about conservation versus development (Daily et al., 2000). Indeed, valuation of ecosystem benefits or services has become the single largest activity within the environmental/ecological economics literature in the last two decades. In addition to many micro-level studies, large-scale initiatives such as The Economics of Ecosystems and Biodiversity (TEEB: www.teebweb.org) are emerging.[2] Valuation is being seen by even natural science journals (e.g., Science and PNAS) as the best way to link science with policy.

Despite this popularity in academia and policy making, both environmental (or now ecosystem service) valuation and benefit–cost analysis (BCA) have attracted substantial criticism from many quarters. Some of the critics are 'internal', those who still believe in the ultimate usefulness of these concepts, and they have focused on lacunae in the practice of valuation, particularly the non-specification of alternatives, non-adherence to analysis of marginal changes, and inattention to ecological detail (Arrow et al., 1997; Bockstael et al., 2000; Daily et al., 2000; Hanley, 2001). Many others have, however, criticized the concepts themselves, pointing to inter alia the serious limitations of contingent valuation, the fundamental non-monetizability of certain values (merit goods, human life, biodiversity), the uncertainty, non-linearity and irreversibility of ecological processes, the problems with aggregation across economic classes and generations, and the inappropriateness of individual consumer preferences as a basis for making public policy decisions (Chee, 2004; Niemeyer and Spash, 2001; Sagoff, 1998; Vatn, 2009). They call for various combinations of multi-criteria

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[1] In theory, valuation could be done in different ways. Throughout this paper, however, we use the terms 'valuation' and 'economic valuation' to refer to 'monetary valuation'.

[2] To be precise, the generic idea of 'greening' national accounts, promoted by environmental economists since the early 1990s (Ahmad et al., 1990), is being specifically focused on ecosystem products and services.

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analysis, participatory valuation and deliberative decision-making, with limited or no role for conventional economic analysis.

We seek to explore the middle ground between these two camps: those believing in an 'improved BCA' and those rejecting valuation and BCA altogether. Distinguishing the descriptive role of economic analysis from the prescriptive role of BCA (Pritchard et al., 2000), we argue that while decision-making should happen in a deliberative framework with inputs from multiple sources, rigorously done ecological–economic analysis can provide important input or decision-support to such a decision-making process. This, however, requires that the focus shift from valuation per se to economic impact analysis, with careful attention to a) alternative scenario development, b) ecologically and institutionally generated trade-offs and c) the techno-institutional context within which economic value is generated. Instead of estimating either single numbers for TEV of ecosystems or for the benefit:cost ratio of a project, ecological economists should focus on identifying the winners and losers, estimating tangible economic impacts in the stakeholders' terms, and estimate the impacts of significant technological and institutional changes, not just small shifts in prices or discount rates.

We outline an approach that explicitly addresses these issues, and illustrate it by examining the impacts of converting a production-oriented state forest to a conservation-oriented wildlife sanctuary in the tropical forests of the Western Ghats region in southern India. Drawing upon prior research, our field work, and expert inputs, we identify two different possibilities within the wildlife sanctuary trajectory: a 'normally expected' trajectory and a 'surprise' trajectory resulting from unexpected technological, institutional and ecological shifts. Our results illustrate how conservation may produce net positive or negative economic impacts for different local stakeholders, but more importantly how sensitive these results can be to the way conservation is carried out and the wider techno-institutional context.

We begin the paper by reviewing in detail the major critiques of economic valuation and BCA mentioned above (Section 2), and present an approach that addresses these critiques (Section 3). We then describe the case study site, the stakeholders, scenarios, and methods (Section 4), and the results (Section 5). Finally, we discuss the implications of these findings in terms of what insights such disaggregated economic impact analysis might provide, especially in the context of tropical forests (Section 6).

## 2. Valuation and BCA: Critiques and Usefulness

The concept of BCA emerged in the context of making public decisions about water resource projects in the USA, and was given a theoretical foundation by welfare economists in the 1950s. It is closely linked with the concept of valuation, especially when applied to environmental issues, because many environmental benefits and costs occur in a non-market context and therefore special efforts are required to estimate them. Environmental economists adopted the idea of total economic valuation (Randall, 1987) and devoted substantial energies to figuring out alternative methods for non-market valuation (Smith, 1993). Most also embraced the corresponding idea of 'extended' BCA (Barbier et al., 1990; Dixon et al., 1986), albeit with qualifications (Pearce, 1994). More recently, many ecologists have promoted the concept of 'ecosystem services' and have adopted valuation (and implicitly a full or partial benefit–cost analysis) as the best or only way to communicate to policy-makers the value of ecosystems, which are otherwise assumed to be free or cheap (e.g., Costanza et al., 1997; Daily et al., 2000).[3]

At the same time, there have been critical voices from within and without. Some have pushed for improved methods and practice of valuation and BCA, whereas many others have completely rejected both concepts. We summarize below both types of criticisms, before suggesting a middle ground that might be most useful. We focus on the economic valuation of tropical forests, which figures prominently in the studies and in the critiques because tropical forests are in many ways exemplars of the salience and complexity of the ecology–society linkage.

### 2.1. Double-counting and Mis-counting

In the practice of tropical forest valuation, four common errors have been identified (see reviews by Chomitz and Kumari, 1998; Lele, 2009; Tacconi, 1995; Turner et al., 2003). First, there is often double-counting of benefits by including both ecosystem processes or functions and ecosystem services. For instance, value is assigned to both nutrient cycling and to the timber production that is the result of nutrient cycling. Second, many studies estimate the production in the forest when they should be estimating only what is extracted, i.e., useful production. Alternatively, some try to value stocks when they should be valuing flows. Third, water flows are often counted as a provisioning service of the ecosystem, when in fact water is the result of rainfall and the forested ecosystem only provides regulatory service. Fourth, even this regulatory service is nuanced: increasing forest cover may sometimes lead to decreases in certain flows and flood regulation benefits may lower than commonly assumed.

### 2.2. Valuation in isolation

An issue that goes beyond practice and into the conceptual arena is the tendency to simply estimate the Total Economic Value (TEV) of an ecosystem in (say) \$/ha (e.g., Adger et al., 1995; Furst et al., 2000; Krieger, 2001). Knowing this number, however precisely, helps little when taking decisions about whether to modify (marginally change) or convert (drastically change) the ecosystem. Making such decisions requires specifying what the alternative land-use will be, understanding what its ecological implications are, and (within the BCA framework) estimating the change in TEV due to the proposed change in ecosystem condition.

Presenting the absolute value of an ecosystem implicitly conveys the message that if the ecosystem were destroyed, society would lose that much income. This was also the message in the famous Costanza et al.'s (1997) study. But this assumption does not stand either ecological or economic scrutiny. Ecosystem 'destruction' is a graphic term that sets up an artificial contrast between 'pristine ecosystems' on the one hand and 'no ecosystem' on the other, neither of which exists in reality. Tropical forests may be replaced by coffee plantations or pastures, grasslands by farming, and wetlands by prawn aquaculture, paddy cultivation or even urban sprawl. But in every case, some biota will continue to exist and provide some biodiversity, some photosynthesis, some infiltration and some carbon sequestration. Some kinds of ecosystem benefits might even increase under deforestation (as we shall discuss below). And conventional economic valuation only allows us to estimate economic impact in the context of marginal changes: non-marginal changes on a large scale (such as the global loss of ecosystem services) would require general equilibrium analysis.

This point has been made a number of times (e.g., Chomitz and Kumari, 1998; Lele, 2009; Toman, 1998; Turner et al., 2003) and several studies comparing two well-defined alternative scenarios or 'before' and 'after' situations do exist (e.g., Norton-Griffiths and Southey, 1995; Yaron, 2001). However, the tendency to estimate value in isolation persists (e.g., Croitoru, 2007; Nahuelhual et al., 2007) and, with conservationists taking to valuation of ecosystem services to press their case for biodiversity conservation, this tendency may even be increasing.

[3] For instance, Costanza et al. begin by saying: "Because ecosystem services are not fully 'captured' in commercial markets or adequately quantified in terms comparable with economic services, they are often given too little weight in policy decisions."

### 2.3. Trade-offs: ecological and social

Not only have many studies tended to value in isolation, but also almost all have emphasized only positive services, thereby de-emphasizing 'ecological trade-offs', i.e., the mixed impacts of environmental conservation. By coining the metaphorically useful but scientifically limited concept of 'natural capital', many ecological economists have aided in this glossing over. Whereas financial returns always increase when, ceteris paribus, financial capital increases, this is not the case with the values derived from natural capital: some values increase while others may decrease. This may occur in two ways. First, ecosystems often produce some negative impacts (i.e., 'dis-services') such as harbouring pathogens or pests (malarial parasites, rats or monkeys that raid crops) (Dunn, 2010; Willott, 2004). Second, changes in ecosystem condition often lead to increases in some ecosystem benefits while decreasing others. Contrary to lay perception, not all benefits are maximized under pristine forest conditions. The existence of ecological trade-offs *within* forestry has been qualitatively pointed out (Lélé, 1994), graphically depicted (Lampietti and Dixon, 1995), and empirically analysed for specific cases: e.g., between timber production and biodiversity conservation (Catterall et al., 2005), carbon sequestration and biodiversity (Venter et al., 2009), timber and streamflow (Chomitz and Kumari, 1998; Dixon, 1997; Hamilton, 1983; Lele et al., 2011), and between timber and grass or timber and other non-timber forest products (Arnold and Pérez, 2001). Some attempts have been made recently to characterize these ecological trade-offs more systematically and comprehensively (Rodríguez et al., 2006). But by and large, ecosystem service assessments, if they look at alternative scenarios at all, highlight only the indirect and long-term gains from preservation versus the direct and short-term gains from conversion or heavy utilization of forests (e.g., DeFries et al., 2004; Maass et al., 2005).

In theory, BCA is all about trade-offs, and economists are trained to evaluate them. But economists have by and large tended to focus on the bottom line, the net benefit, so as to recommend one scenario over the other, and have not focused on who gains and who loses, what one might call the 'social' trade-offs. This stems from the overwhelming focus in neoclassical economics on the so-called 'efficiency' objective, and inattention to distributional dimensions (Kerr and Swarup, 1997). Thus, ecological–economic analyses of the impact of ecosystem change on the full set of ecosystem benefits (and dis-benefits) and its winners (and losers) are rare (Turner et al., 2003), especially in a tropical forest context.

The few cases where ecological–economic analysis has carefully examined trade-offs are worth noting. Ecological trade-offs have been clearly highlighted in economic terms in two studies. Shahwahid et al. (1999) analysed the ecological impacts and economic consequences of different logging systems in a Malaysian forested watershed on downstream hydropower generation and water quality effects for users further downstream, and found that the TEV under reduced-impact logging might be higher than that under both strict conservation and conventional logging. Nelson et al. (2009) used an integrated and spatially distributed ecological model called InVEST to highlight the trade-off between agriculture, timber and housing values on the one hand and water quality, soil conservation, storm peak regulation, carbon sequestration and biodiversity on the other in the context of a river basin in Oregon state, USA.

In terms of social trade-offs, Hein et al. (2006) highlight the trade-off between local communities that derive both use and conservation values and global conservation beneficiaries. A more comprehensive analysis is the study of an Indonesian National Park by van Beukering et al. (2003). They analysed the implications of three scenarios, viz., conservation, selective use and deforestation for a wide range of benefits (water supply, fisheries, flood and drought prevention, agriculture and plantations, hydropower, tourism, biodiversity existence value, carbon sequestration, non-timber forest products (NTFPs) and timber).[4] They disaggregated their estimates by regions (provinces) and also by type of stakeholder (local community, local government, elite industry, national government, and international community), and thereby argued that the mismatch between the distribution of the benefits of conservation and that of political power makes it difficult to get policy support for conservation to materialize.

### 2.4. The role of techno-institutional context

Past reviews, even while adopting a neoclassical approach, have recognized that environmental "*benefit levels are highly location specific and scale dependent*" (Chomitz and Kumari, 1998, p.14). However, as the idea of ecosystem service valuation has gained momentum, there appears to be an increasing tendency amongst analysts to ignore the societal context, focusing only on the biophysical links. For instance, Naidoo et al. (2008) attempt to generate a 'global map' of ecosystem services, where the value of services emerging from each pixel is estimated with little attention to whether or not there are users for those services. And even though Costanza et al. (1997) were criticized for extrapolating data from a few contexts to the whole globe, discussions continue on how to carry out 'benefits transfer', a peculiar euphemism to say the least (see Spash and Vatn, 2006 for a detailed critique). In some cases researchers have used the global estimates from Costanza et al. (themselves based upon much extrapolation) to generate location-specific estimates (e.g., Seidl and Moraes, 2000).

The relevance of context goes beyond just the need for accuracy in choice of price or productivity coefficients. It alerts us to the fact that 'services' or 'value' or 'benefits' are social constructs, emerging from the interaction between human labour, institutions, capital and the environment. Without taking an extreme social constructivist approach, one may say that, while ecosystems exist independent of human presence or perceptions, the 'value' of ecosystems (whether measured in economic terms or otherwise) is always 'co-produced' through specific human interactions with them. Therefore it is pointless and misleading to talk of valuation without the context (Pritchard et al., 2000; Vatn, 2005).

### 2.5. The fundamental problem with BCA: aggregation

BCA is focused on giving the 'right' answer to decision-makers. This requires making several assumptions: that all values are monetizable, that all monetized values can be added and subtracted to come up with 'net change in economic welfare', and that 'economic welfare' as defined in BCA should be the criterion for societal decision-making. Critics of BCA have highlighted the problems with all three assumptions (see Ackerman, 2005 for a useful summary; see also Sagoff, 1998; Vatn, 2009). First, it may be not just impossible but ethically quite objectionable to put a monetary value on things that have intrinsic value, such as human life or the lives of other organisms. 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).

Second, adding up all benefits and costs across different people involves using the Kaldor–Hicks compensation criterion to get around the problem of non-comparability of individual utilities. But this criterion has been the subject of much criticism, to say the least, as it depends upon hypothetical, not real, compensation. Equally important, the aggregation of benefits derived by persons with very different levels of wealth or income assumes that the marginal social (or individual) utility of income for each person is identical regardless of

[4] Their treatment of ecological trade-offs, however, appears weak: in their analysis, water supply, flood prevention and hydropower benefits account for almost half of these benefits. This invites scepticism in face of the forest hydrology literature mentioned earlier and recent valuations of hydrological services based on primary data (Aylward and Echeverria, 2001; Lele et al., 2011).

their level of income. This is clearly untenable, and does not pass the 'laugh test' (Farrow, 1998).[5] In the context of tropical forests, where the stakeholders range from local firewood collectors to high-income eco-tourists to global beneficiaries of climate change mitigation, income inequalities are particularly severe. One solution that has been recommended frequently is to carry out sensitivity analysis using different distributional weights (e.g., Dasgupta et al., 1972). But with rare exceptions (Azar and Sterner, 1996; Murty and Menkhaus, 1998), this has not been practised.

Similarly, for long-term environmental impacts, BCA involves aggregating benefits and costs across generations, and the act of discounting with a positive discount rate de-emphasizes the preferences of future generations. Sensitivity analysis using different discount rates is common, but rarely with a zero discount rate, although this idea is gaining some acceptability in neoclassical economics in recent times (e.g., Dasgupta, 2008).

A third, and deeper, critique of BCA is that decisions about public policy should not be based upon the mechanical aggregation of individualized economic preferences (even after carrying out 'extended BCA' to incorporate environmental aspects) precisely because individual preferences are relevant only in the context of decisions made by individualistic consumers about commodities in a market, whereas decisions about environmental matters are made by people as citizens about public and common-pool goods that have merit good attributes and ethical dimensions. Thus, an increasing number of ecological economists (Vatn, 2005), supported by political philosophers (Sagoff, 1998; Taylor, 1992) and others (Jacobs, 1997) recommend rejecting BCA as a basis for making public policy decisions. Instead, they recommend the use of deliberative decision-making approaches of various kinds, including participatory multi-criteria assessments (Rauschmayer and Wittmer, 2006) or other ways of structuring the deliberative process (see Niemeyer and Spash, 2001 for a review).

### 2.6. Is there a baby in the bathwater?

In light of the fundamental limitations of valuation and BCA that go beyond simply 'errors in practice', what should be one's stance towards these techniques? Should one simply abandon the entire exercise and shift to deliberative valuation and decision-making methods? Or should one still hope for 'better BCA'? Or is it that there is no simple either-or?

To begin with, we note that *purely* deliberative valuation methods, where both impact criteria and the change in their magnitudes are estimated only through deliberative processes, have their own limitations. For instance, in many multi-criteria evaluation exercises there is no explicit model linking interventions to outcomes. All variables, process or outcome-related, are called 'criteria' and then the participants are asked to identify possible links between them (e.g., Proctor and Drechsler, 2006). Without denying the importance of local ecological and other knowledge, we would argue that relationships derived from empirical investigation (provided they are expressed transparently) are more useful than those obtained only through deliberation. Even staunch supporters of deliberative processes such as Vatn (2009) acknowledge that "science has a very important role to play in deliberation over complex environmental resources".[6]

Second, it is important to make a distinction between the strongly normative exercise of BCA, and economic analysis as a descriptive exercise a la Pritchard et al. (2000). BCA is clearly too reductionist and normatively loaded to be acceptable.[7] As Bromley (1990) argues, the role of analysis is not to reduce everything to a 'bottom-line', but rather to "attempt to understand who the gainers and losers are, how they regard their new situation in their own terms, and what this means for the full array of beneficial and harmful effects." Economic impact analysis across different scenarios can do the latter and provide useful decision-support, without usurping decision-making.

Third, even if decisions must not be based purely on economistic considerations, many people will experience some of the impacts of public decisions on their individual lives in direct economic terms. In other words, although distinguishing between individualized economic preferences and citizenship-based thinking is useful, in reality, there is no neat separation between individual and social rationality or between privatizable and common-pool goods. There is still a significant value in carefully tracing the impacts of ecological change on household incomes, and offering this as *one* piece of information in the deliberative decision-making process.

## 3. An Alternative Approach: Disaggregated Economic Impact Analysis, With Techno-institutional Context

Our starting point is that valuation used as economic analysis can be useful as a descriptive exercise even if BCA as a prescriptive exercise is not. But for such analysis to be useful, it must be based on an acceptance of both ecological synergies and trade-offs, sensitivity to distributional consequences, and an understanding of how value emerges from the interaction between technology, institutions, labour and the ecosystem. We propose an approach that explicitly embraces these features. The key elements of this approach are:

Step 1: Identify the main *benefits and costs* associated with a particular ecosystem in its current condition. Identify those benefits that are *monetizable*, as against others that are more in the nature of merit goods or non-monetizable for ethical reasons.

Step 2: Identify the *stakeholders* best associated with each benefit and their socio-economic position (such as income class). *Disaggregate* them into socio-economically homogeneous groups.

Step 3: Identify the *process* through which a particular economic benefit flows to (or is obtained by) the stakeholder group, including the *technology* (of harvesting, collecting, or processing the product or service), and the *institutions* (property rights, market characteristics, governance systems) on which the benefit flow is contingent.

Step 4: Identify clearly the *alternative land-use scenarios* that are being considered or may be realistically considered, and what changes in technology, investments and institutions are associated with them.

Step 5: Estimate how changes in land-use might change the biophysical magnitude of each benefit, and what kind of *ecological and social trade-offs* it might generate.[8]

Step 6: Estimate the economic value of the benefits and costs under the alternative scenarios, but calculate only *stakeholder-wise net change*, and examine how sensitive these changes in benefits and costs are to changes in the ecological or techno-institutional context/assumptions.

## 4. Study Area, Scenarios, and Methods

To illustrate the above approach and to highlight the insights it provides, we carried out an economic analysis of impacts of changing the governance regime for a tropical forest ecosystem in southern India. The setting, scenarios and methods used for estimating the economic impacts are described in this section and the results in the following one.

[5] Lay people simply laugh at the idea that one dollar more to a rich person is as important as a dollar more to a poor person.

[6] Vatn calls for a structured interaction between the purveyors of scientific information ('experts') and lay people with 'normative and practical competencies'.

[7] Even if we ourselves practised it in the past (Lele et al., 1988).

[8] Note that steps 4 and 5 overlap and may require iteration.

### 4.1. Location and ecological characteristics

The Biligiri Rangaswamy Temple (BRT) Wildlife Sanctuary, located in Chamarajnagar district of Karnataka state, India (see Fig. 1) lies at the confluence of the Western and Eastern Ghats mountainous regions, making it particularly rich in biota. Its 540 $km^2$ of forests is spread over an undulating terrain ranging from 600 m above MSL to 1800 m, and contains five broad vegetation types such as scrub thorn forest (28% of total area), deciduous forest (61%), evergreen forest (7%), high altitude grassland (3%) and high altitude stunted cloud forest (*Shola*: 1%) (Ramesh and Menon, 1997). The sanctuary harbours 36 species of large mammals (including elephants, tigers, leopards, Indian bison, sloth bears, spotted deer, sambar deer and barking deer), 245 species of birds including several endangered species (Aravind et al., 2001), 145 species of butterflies (N.A. Aravind, pers. comm.) and ~1000 species of higher plants.

### 4.2. Brief socio-ecological history[9]

Human presence in and dependence on these forests dates back at least a thousand years and has taken various forms. The Soligas, a community recognized as a Scheduled Tribe in the Indian constitution, are the oldest inhabitants of these forests, and they historically practised shifting cultivation, trapping and gathering forest produce. But by the 1970s, shifting cultivation had been stopped, and Soligas were forced to settle in specific locations. They became increasingly dependent on collection and sale of non-timber forest products (NTFPs) to supplement their livelihoods. Today, about 4700 Soligas live inside and 16,300 on the fringes of the forest.

Other communities settled at the BR Temple in certain pockets in the hills starting in the early 1800s, but the plains around the BRT have been predominantly populated by non-Soligas for several centuries, and have been under settled agriculture (both rainfed and irrigated) and livestock-based livelihoods. Their dependence on the forest is primarily for firewood, small timber, and grazing of livestock. Streams that originate in the BRT forests have been dammed to create small and medium-sized reservoirs that irrigate agriculture. A small area (~6 $km^2$) in the heart of the forests was also leased out for coffee cultivation in the 1890s.

The forests were declared as Reserve Forest (RF) in the 1930s and managed primarily for timber and bamboo by the state, with some concessions for NTFP collection. A significant shift in management objectives happened in 1974–76, when about 300 $km^2$ of the BRT forests was declared a Wild Life Sanctuary (WLS), which was expanded to 540 $km^2$ in 1992. The forests now occupy an important place in the list of wildlife tourism spots in Karnataka and as a landscape of significant biodiversity value. At the same time, wildlife imposes significant costs on villages in the periphery through crop damage and occasional livestock and human deaths.

### 4.3. Stakes and stakeholders

From the above, one can see that the BRT forest ecosystem provides multiple benefits and some dis-benefits. Some of these are clearly direct use benefits (firewood, grazing, timber), others are indirect use benefits or services (watershed regulation, carbon sequestration, crop damage) and still others are non-use benefits (wildlife and biodiversity). Of these, the benefits and dis-benefits that are monetizable and significant are listed in the columns of Table 1. Those left out because they were considered insignificant are forest-based foods, micro-climatic benefits for agriculture, pollination services and pest control for agriculture. Also left out due to non-monetizability are the religious importance of these forests, the existence value of the wildlife, and the human deaths caused occasionally by elephants.

When identifying stakeholders, to balance between socio-economic detail and analytical tractability, we aggregate them into five relatively homogeneous groups: 1) Soligas living within and on the fringes of the forest, 2) poor, non-Soliga peasants living mainly in fringe villages, 3) rich non-Soliga landowners in fringe villages, 4) the rest of India, and 5) the global community (see rows of Table 1).[10] The point to be noted here is the enormous socio-range in the socio-economic condition of these stakeholders: from local forest-dwelling Soligas and non-Soliga landless and marginal farmers often below the Indian poverty line to those in developed countries at the other extreme of income and wealth. This is very different from the contexts in which BCA evolved, viz., water resource projects in the USA.

### 4.4. Institutions and technologies related to forest ecosystems

The key manager of the forest as of now is the Karnataka Forest Department (KFD). It regulates forest access by local communities and by tourists, and carries out all silvicultural and protection activities. Soligas and other local communities have customary rights, but these were not properly recognized in the law till very recently and are still heavily regulated. After the conversion of RF to WLS in 1976, KFD continued to permit NTFP collection by Soligas but officially banned firewood collection and grazing by fringe communities. It also officially stopped timber logging but continued to extract and sell dead and fallen trees and logs from plantations. NTFP collection and marketing by Soligas are done through a cooperative set-up by the state ostensibly for Soliga welfare with some help from non-governmental organizations (Lélé et al., 1998). Markets for NTFPs are regional, while those for firewood are highly local. WLS management is done solely by KFD with no local consultation, no sharing of revenues from tourism. KFD's WLS management does not involve major replanting of degraded lands or proactive measures for NTFP regeneration, is backed by limited ecological monitoring, and focuses primarily on prevention of poaching and forest fires.

Lands outside the KFD's jurisdiction are managed by the Revenue Department, while the irrigation reservoirs fed by the streams from BRT are managed by the Minor Irrigation Department of the state government. There is limited coordination between these departments. Soil and water conservation measures are occasionally taken up in the fringes, but the Major Irrigation Department is independently working on expanding canal irrigation. Electrification and road connectivity are expanding rapidly. Agriculture is a mix of commercial and subsistence crops and this mix is changing in response to increasing market penetration.

### 4.5. Scenarios, trajectories and trade-offs

The scenarios we compared were the management of this landscape as an RF focused on producing timber and other products, and its management as WLS focused on wildlife conservation.[11] Using what is known about the pre-1976 (RF) and post-1976 (WLS) scenario, we constructed a narrative of how the forest condition, access and thereby the flow of forest benefits would differ in the post-1976 period under WLS management as compared to RF management. To do so, we examined official forest department documents and past studies (cited below), held interviews with a number of current and past forest officials, ecologists and local community members, surveyed

[9] Based on Rajan (1983), Lélé et al. (1998) and Bawa et al. (2007).

[10] Admittedly, there is some simplification involved here. E.g., the averted damages from increased carbon sequestration accrue to everybody in the globe. Nevertheless, given the disparities in income across the globe, all estimates of averted damages are heavily biased towards damages experienced in the developed world and hence the ‘global stakeholder’ here is largely the developed world.

[11] We use 1976 as the starting date because that is when the WLS policies became fully effective, not 1974.

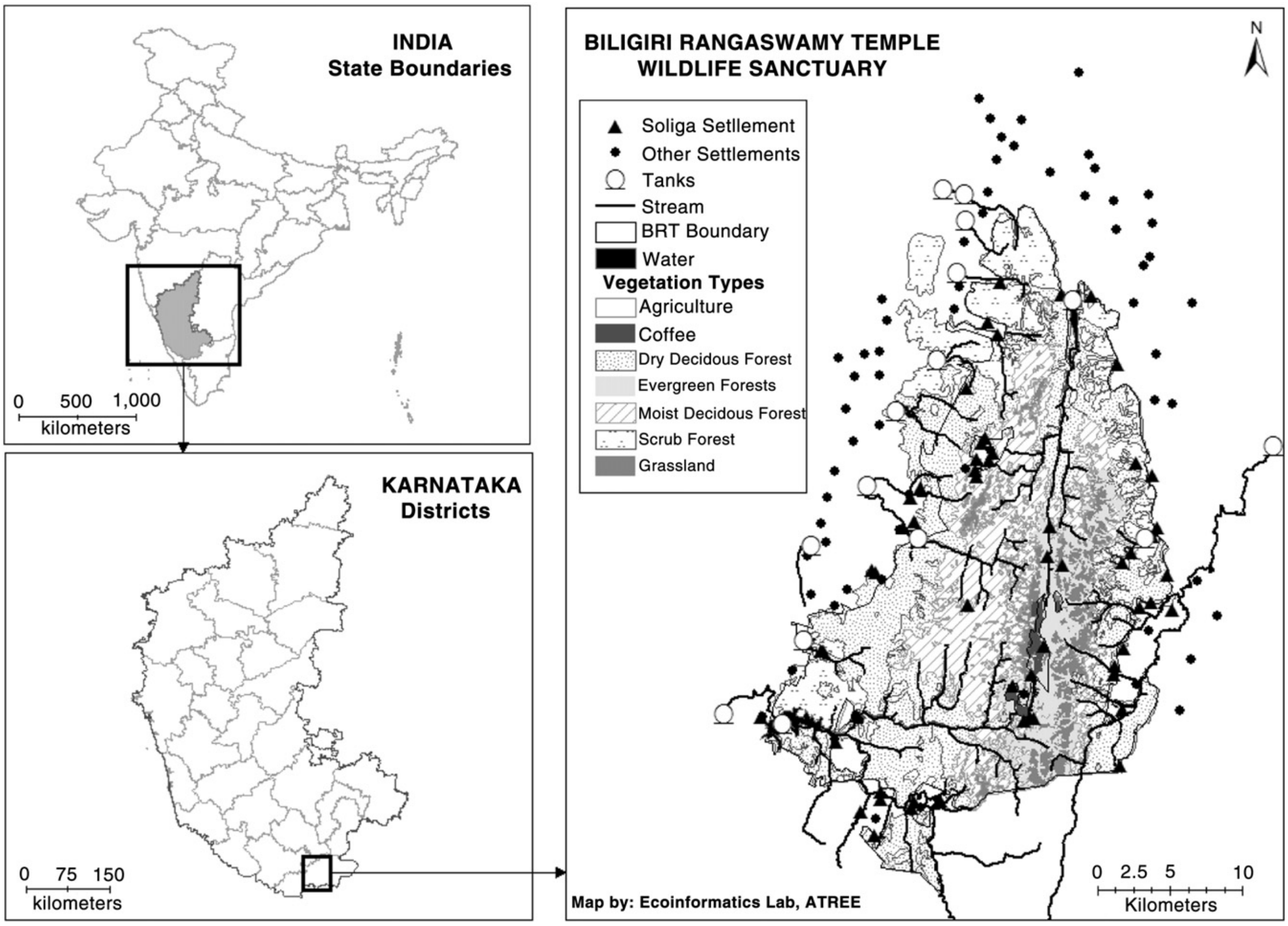


**Fig. 1.** Map showing location of Karnataka state in India, BRT wildlife sanctuary in Karnataka state, the forests of BRT wildlife sanctuary and the location of human settlements, streams and reservoirs inside and on the fringes.
Courtesy: Ecoinformatics Lab, ATREE.

the prevailing condition of the forest vegetation, and took cognizance of a detailed analysis of Landsat MSS, TM, IRS, and AVHRR data for the period 1973 to 1999 for BRT WLS and neighbouring areas that continued to be RFs after 1976 (Krishnaswamy et al., n.d.). The pathways through which the impacts of this change in management objectives would be transmitted ecologically and experienced economically are given in Fig. 2.

The 'normally expected' trajectory for a forest that is declared a WLS is that of increased protection and reduced local access as compared to RF, resulting in forest regeneration and increases in biodiversity and carbon sequestration, as compared to the RF scenario. This is what is depicted in Fig. 2. While the first two decades after 1976 showed these trends, there have been several shifts and 'surprises' after 2001. We therefore developed two versions of the WLS scenario. (Although the analysis is only partly ex-post, we use the present tense for the sake of convenience.)

a) *The 'normally expected' trajectory (WLS_1)*:

1. Once BRT forests are declared a WLS, logging ceases, and firewood collection and grazing are officially banned. Significant resources are allocated to patrolling, in terms of vehicles, jeeps and guards to deter illegal firewood collection and grazing. Increased efforts are also made for soil and water conservation, control of poaching and prevention of forest conversion for settled cultivation. These additional resources come from the state budget, i.e., the national taxpayer.
2. The net result is that the forest regenerates and biodiversity improves in the interior parts of the sanctuary, and the rate of forest degradation slows down in the parts that are still subject to heavy human use, i.e., the northern fringes.
3. The conservation of wildlife leads to the emergence of eco-tourism as a significant economic benefit, with visitors from both urban India and abroad. At the same time, it also leads to increases in wildlife damage to crops and attacks on humans.
4. The human population living in the forest grows more slowly than might have been the case in the RF scenario because the conversion of forest to agriculture is prevented.
5. NTFP collection and sale, and deadwood collection for self-consumption, by Soligas living within the sanctuary boundaries continue to be permitted.[12] NTFP yields increase due to forest regeneration. Firewood and fodder availability to Soligas also increase marginally.
6. For fringe populations, firewood collection is officially banned, but significant illegal extraction continues. The population dependent on the forest for firewood grows more slowly as compared to the RF scenario. Forest degradation reduces firewood availability in the RF scenario.
7. Similarly, grazing is only partially controlled. Livestock held by the interior population is officially permitted to graze in the forest. Grazing access to the fringe population is reduced, and manifests itself in a lower growth rate of the forest-dependent livestock population compared to the RF scenario.

[12] The Indian Wildlife Protection Act 1972 allows the Chief Wildlife Warden to permit NTFP collection in WLSs. Ecological research on the impact of NTFP collection in BRT WLS has shown that it can be sustainable and need not have major adverse biodiversity impacts (Ganesan and Setty, 2004; Lélé et al., 2004).

**Table 1**
Significant economic benefits (and dis-benefits) and relevant beneficiaries linked to the BRT forest ecosystem, with likely direction of change when moving from Reserve Forest to Wildlife Sanctuary status.

| Stakeholder | NTFP benefits | Firewood benefits | Grazing benefits | Soil conservation benefits | Eco-tourism benefits | Carbon seq. benefits | Timber & bamboo benefits | Wildlife related dis-benefits | Protection costs |
|---|---|---|---|---|---|---|---|---|---|
| Soligas | + | +/− | +/− | | | | | + | |
| NS local poor | | − | − | +/− | | | | + | |
| NS local rich | | | − | + | | | | + | |
| Rest of India | | | | | + | | − | | + |
| Global | | | | | + | + | | | |

Note: "+/−" represents cases where the direction of mixed or unclear at the outset. "NS" refers to non-Soliga (i.e., non-tribal) communities.

8. Forest regeneration reduces soil erosion, thereby reducing the rate of siltation of reservoirs fed by the streams running off the hills. The beneficiaries are the richer farmers who owned land in the irrigation command. Groundwater recharge, however, does not increase significantly.[13]
9. Forest regeneration increases the standing biomass and therefore carbon sequestered, benefitting the global community.

b) *The 'surprise' trajectory (WLS_2)*:
The trajectory is based on several key events that actually took place post 2001. These events changed the relationship between stakeholders and the forest ecosystem.

1. *Loss of rights to NTFP collection and sale*: An important feature of WLS_1 was the continuation of NTFP-based livelihoods for the Soligas. However, in 2004, the Forest Department cancelled the permission given to the Soligas to collect NTFPs, citing a Supreme Court interpretation of the Wildlife Act. NTFP collection for sale virtually ceased by 2006 (Setty et al., 2008), dramatically affecting Soliga livelihoods (Sandemose, 2009). In WLS_2, although NTFP species continue to grow and even flourish in the forest, they do not provide any direct use benefit to any stakeholder in society after 2006.[14]
2. *Changes in irrigation technologies downstream*: In WLS_1, increased soil conservation translated into slower reductions in irrigation benefits. However, in 2001, a new 'external' irrigation system was commissioned, that brings water in a canal from a major dam 70 km away. The command area of this canal overlaps partially or fully with that of at least six of the local irrigation reservoirs (Harish, 2010; Purushothaman et al., 2009). In WLS_2, the contribution of the local reservoirs to irrigation declines after 2001, rendering the soil conservation function provided by the forest cover less valuable.
3. *Ecological dynamics*: In the WLS_1 scenario, invasive exotic plants such as *Lantana camara* and *Chromelina odorata* are present but, following Murali and Setty (2001), they were assumed to not affect regeneration of the natural forest vegetation. Later studies, however, found that *Lantana* had spread rapidly in the post-2001 period in spite of some weed eradication efforts. *Lantana* incidence in sample plots increased from 41% in 1997 to 81% in 2008, making it the most commonly occurring plant in BRT and appearing to suppressing the regeneration of other species (Sundaram, 2011; Sundaram and Hiremath, 2012). Regional climatic shifts may also be making the BRT forests drier than before (Krishnaswamy et al., n.d.), further affecting plant growth. Thus, in WLS_2 tree productivity and hence the net carbon sequestration rate is significantly lower than WLS_1.

[13] The nature of the hydro-geology of the region, with steep rock slopes for the valleys and relatively thin soils, implies that groundwater recharge happens largely through flooding of the plains, and infiltration in the hills may play a lesser role. This was corroborated by a forest hydrology study conducted on one of the BRT streams (Lélé et al., 2007, chap.5).

[14] In 2011, after a long struggle, the Soligas were granted legal rights (as against temporary leases) to NTFP collection under the recent Forest Rights Act 2006. It remains to be seen whether this translates into actual and long-term livelihood gains to the Soligas.

The direction of anticipated change in economic benefits or costs derived from different products or services by different stakeholders is depicted qualitatively in the cells in Table 1 as arrows of change. Where WLS_2 differs from WLS_1, the WLS_2 change is indicated in brackets.

### 4.6. Ecological and economic methods[15]

Broadly speaking, the schematic of Fig. 2 was implemented by estimating the trajectories of forest cover change in the different scenarios, and then tracing the implications of these changes for each of the benefits being valued for the corresponding beneficiaries. We used a combination of primary and secondary data, results from a number of studies, and educated guesses based upon discussions with experts, researchers working in BRT, and local communities. We briefly summarize the methods and assumptions below; more details are available with the authors. Except when specifically mentioned, the methods and assumptions are identical for WLS_1 and WLS_2. In general, extrapolations for WLS_1 were based on trends in time-series data up to 2001, and deviations under WLS_2 were estimated based on post-2001 data.

A vegetation map prepared by Ramesh and Menon (1997) provided the basic vegetation categories and areas for 1995. The trajectories of vegetation change were estimated using sources mentioned in Section 4.4. The implications for the production of firewood, grass, NTFP, timber and carbon sequestration were deduced from this trajectory, but in estimating the eventual benefits we also had to factor in the changes in demand, access rights and the level of enforcement.

For the economic valuation, where product markets were well developed (firewood, timber, NTFPs, tourism), gross returns were estimated from market prices. In the case of carbon sequestration, the absence of a market meant we had to use the marginal value of averted social damage. In the case of non-marketed benefits (grazing and irrigation), the value was imputed from returns on the final marketed products (dung, milk, draught power, livestock and agricultural crops). Details of methods and assumptions are given below. The cost of production was generally the opportunity cost of time, which was taken to be 50% of the market wage rate, following patterns of employment and unemployment observed in an earlier unpublished study that involved year-long monitoring of 114 Soliga households (hereinafter HHMON), and following Yaron (2001). Two different wage rates were assumed: one for the peak growing season from July–October and a lower one for the rest of the year. The entire analysis was done in constant 2010 prices, with no product-specific inflation except in the case of carbon sequestration, where the fluctuating dollar-rupee exchange rate had to be accounted for.

#### 4.6.1. Estimating local forest-dependent populations

Villages adjacent to the WLS boundary were identified from maps in the district census handbooks. Additional lists of Soliga settlements were obtained from surveys conducted by ATREE's Community

[15] Only a brief outline of data and methods is given here. For details, please contact the authors.

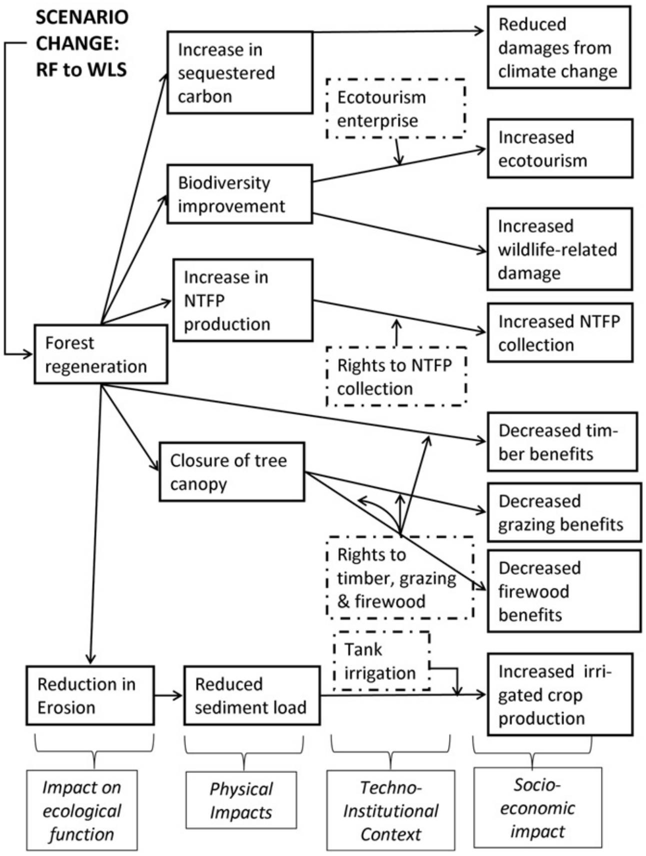


**Fig. 2.** Schematic depiction of the various pathways through which the change in forest management from RF to WLS will generate physical and socio-economic impacts, and the influence of the techno-institutional context.

Conservation Centre in the WLS. Population data and trends came from the official decadal censuses between 1971 and 2001, as well as censuses carried out by social workers working with the Soligas. Census data on agricultural labourers and survey data from HHMON provided indirect estimates of the fraction of poor households.

*4.6.2. Firewood benefits*

Estimates of per capita consumption of firewood and its variation by location (fringe versus interior) came from Shankar et al. (1998). This study and our investigations showed that firewood was the sole source of fuel for interior populations and its use did not change under the WLS scenario. But fringe populations depended only to the extent of 40% in the RF scenario (the rest coming from agricultural residues) and this declined significantly in the WLS scenario to about 15%, being limited to the poorer sections of that population.

*4.6.3. Grazing benefits*

Population density and growth rates for livestock (cattle, goats and sheep) were obtained from the village-wise sheets of the state Livestock Census and extrapolated following discussions with local experts. The number of livestock dependent on the forest was assumed to be 40% higher at the end of the RF scenario as compared to the WLS scenario, based on data from neighbouring RFs. Labour involved in livestock management was estimated from a combination of HHMON and village-level rapid surveys and discussions in the fringe areas. Availability of alternative sources of fodder was derived from a study of fodder benefits in nearby irrigated farms (Purushothaman et al., 2009). Benefits of grazing include manure, milk, draught power and sale value of livestock. Prices and quantities of manure, milk, use/rental of draught power and sale of cows, goat and sheep were obtained through discussions with fringe villagers.

*4.6.4. NTFP collection benefits*

Data on quantity and price for each NTFP product were obtained from the three local NTFP marketing co-operatives (and also Bawa et al., 2007; Lélé and Rao, 1996) and their trends were extrapolated. Labour involved in collection and its opportunity costs were estimated from HHMON. NTFP royalties charged by the KFD were subtracted as costs from the Soliga income but included as income to the state exchequer.

*4.6.5. Timber*

Timber logging and bamboo extraction continues as before in the RF scenario but goes to zero in the WLS scenario (FD Working Plan documents). The FD's royalties from the auctioning of timber and bamboo extraction rights minus costs incurred by KFD in developing the timber and bamboo plantations are assumed to represent the net profit from these products. Data for these costs were extrapolated from the neighbouring Kollegal Forest Division (FD Annual Administrative Reports).

*4.6.6. Soil conservation benefits*

Irrigation reservoirs fed by streams from BRT were identified from topographic maps. Soil erosion rates were estimated on the basis of the Revised Universal Soil Loss Equation (RUSLE: Renard et al., 1997). For this, catchment boundaries and slope data were derived from publicly available Shuttle Radar Telemetry (SRTM) images. These were overlaid on a soil map from the National Bureau of Soil Survey and the vegetation map mentioned above to obtain the parameters required for applying the RUSLE to estimate soil erosion (as per Jain et al., 2001). The reservoir capacities and areas irrigated were obtained from the Minor Irrigation Department's records. For most reservoirs (those located in flatter terrain), the dead storage capacities were insignificant and so the area irrigated was assumed to shrink in proportion to the fraction of reservoir capacity lost by siltation. For two reservoirs where the dead storage was a significant portion of the total storage, it was assumed that irrigated area would begin to decline only after the dead storage had been filled up with silt. It was assumed that in the absence of reservoir irrigation, farmers would carry out rainfed cultivation. The difference in economic returns from irrigated and rainfed farming was obtained from Purushothaman et al. (2009).

*4.6.7. Eco-tourism benefits*

Data on profits from and current trends in eco-tourism under WLS scenarios came from annual reports of the state-controlled company Jungle Lodges and Resorts (JLR). To this we added revenue from daily visitors using data from FD records. Under the RF scenario, tourism was assumed to be non-existent. In practice, a part of JLR profits goes as royalty and licence fee to the state. But given that JLR is also a state-owned enterprise, both the profits and the royalties constitute benefits to the rest of India.

*4.6.8. Wildlife damage costs and protection costs*

Data on compensation paid to farmers for crop and livestock damage caused by wildlife were available from FD records. Field visits and experience elsewhere indicated, however, that this is an underestimate, as many claims were either not filed, rejected on technicalities or under-compensated. Based on field discussions, the total estimated damage was taken as double the compensation paid out, and paid out compensation was a dis-benefit to the rest of India, while the unpaid compensation was counted as a dis-benefit to local farmers. Data on the additional protection and management costs incurred under the WLS scenarios were obtained from FD records.

*4.6.9. Carbon sequestration benefits*

Areas of different vegetation types in the French Institute vegetation map were converted to basal area using data on plant species, tree density and girth were available for 134 plots (Murali et al., 1998). These were converted to standing biomass using available allometric equations for the dominant species and then to mean annual increment using other studies in the Western Ghats (e.g., Lélé, 1994). Under the WLS scenario, some of the vegetation is assumed to transit from discontinuous to dense, and the mean annual increment in standing biomass is assumed to taper off (from an initial value of 2% per year, to 1% and then to 0.5%) to reflect density-dependence of growth rates.

There is a large literature on valuing carbon sequestration benefits. Carbon credits in sequestration projects in Costa Rica work out to ~$10/tonne in 1994 prices (Pagiola, 2008). But carbon markets are thin, since there is no global treaty with meaningful emission caps and defined trading regimes, and therefore hardly reflect the marginal value of averted damage from carbon sequestration. Estimates of the latter range from $20/tonne (1994 prices: Pearce and Moran, 1994, quoting work by Frankhauser) to $42 (1992 prices: Polasky et al., 2011; see also review by Tol, 2009). Given the increasing evidence of global warming, we have chosen a value for avoided damages close to the higher end of this range, i.e., $42 in 2001 prices.[16]

#### 4.6.10. Discount rates and time horizon

The choice of discount rates and time horizons is as sensitive a topic as distributional weights and much more heavily debated. A review by Howarth (2005) indicates that a real rate of 6% is used in monetary BCA, but it should be much lower in environmental BCA, and there is a large literature arguing for 0% (see review in Dasgupta, 2008). So we analyse using 3% and 0%.[17]

The choice of time horizon is also a complicated issue. In conventional 'development' projects, the effect of the project may last for a limited number of years, such as the life of a hydropower dam or mine. Some of the effects of a 'conservation' project may also disappear after some years: e.g., the effect of reduced access to forest products may diminish as people seek out other resources. But the biodiversity conserved may have undiminished or even increasing value over time, as biodiversity may become scarcer over time. This poses a problem for setting the time horizon. Nevertheless, we set a horizon of 50 years, on the assumption that a) local users will have adjusted to the reduced access in various ways, b) future generations are at some point likely to review the conservation decision afresh, and c) the uncertainty associated with all projections beyond 50 years becomes very high, making all estimates doubtful. This time horizon encompasses at least two generations. Sensitivity analysis showed that the results do not change substantively with a longer time horizon.

## 5. Results

The estimated changes in the present value (PV) of the economic benefits obtained by different stakeholders because of the shift from RF to WLS are given in Tables 2–4. In all cases, we aggregate only across all benefits for a particular stakeholder group, not across stakeholders (i.e., across columns but not across rows). Focusing initially on the RF to WLS_1 case, we examine the general trends in individual benefits and the trade-offs faced by stakeholder groups using calculations based on a 3% discount rate (Table 2) and then look at the effect of aggregation over time by comparing with results for a 0% discount rate (Table 3). We then examine the effect of 'surprises' caused by techno-institutional changes under the WLS_2 scenario based on a 3% discount rate (Table 4).

### 5.1. Changes in Individual Benefits

Changes in individual cells, without reference to other cells, provide a few insights. The direction of change (declines in firewood, grazing and timber benefits and increases in irrigation benefits due to soil conservation) can in most cases be anticipated from the ecological dynamics outlined in Section 4.5 and captured in Table 1. Firewood and grazing impacts for the Soligas were initially ambiguous, because a regenerating forest would increase biomass availability while stricter conservation rules would mean reduced access. The net effect turns out to be negative in this particular case.

Some of the percentage changes relative to the benefits in the RF scenario are notable. The 75% increase in NTFP benefits to Soligas reflects how the ecological potential of the forests is realized under a regime that permits NTFP harvesting. On the other hand, the decline in grazing benefits to the non-Soliga communities (poor and rich) is fairly sharp (−22%), showing how curtailment of access hurts these forest-dependent (even if not tribal) communities significantly.

Surprisingly, soil conservation benefits do not increase significantly when the forest regenerates (+1% or +2%), indicating a weak link between forest cover change, soil erosion and irrigated area. There are several reasons for this. First, most of the benefits come from 3 large reservoirs, and two of these have significant 'dead storage' capacity that buffers them against siltation. Second, the other large irrigation reservoir has a catchment that was already covered with intact forest, and so it does not benefit from the shift in forest management. Third, the vegetation changes we have projected are less dramatic than typical eco-restoration (or deforestation) narratives. Fourth, erosion rates vary significantly within each catchment, and so improvements in vegetation do not uniformly translate into reductions in soil erosion. We believe our results are realistic representations of the forest cover-soil erosion link and discuss their significance in Section 6 below.

Finally, it may be noted that the increase in the value of sequestered carbon is only 9% in PV terms (or 15% in undiscounted terms), indicating that these forests may have a limited capacity to add carbon to their biomass. This is because the regeneration takes place mostly on the fringes of the WLS, where the floristic type is scrub thorn, which does not have a high standing biomass under the best of circumstances.

### 5.2. Trade-offs Faced By Each Stakeholder

As discussed in Section 2, if a group is relatively homogeneous in its interests and income levels, there is some basis for converting all benefit flows for that group into economic units and estimating net changes. These net changes are given in the rightmost column of Tables 2 and 3. We find that different stakeholders face different trade-offs and stand to lose or gain in different ways. Specifically:

a) In spite of some declines in firewood and grazing benefits, the Soligas on the whole would benefit significantly under the WLS_1 scenario relative to the RF scenario, with a net economic benefit of 2.68 million US$ (PV at 3%). This is because the increased benefits from NTFP collection and sale far outweigh the losses in firewood and grazing.[18]
b) In contrast, the poor non-Soliga households (mostly located on the periphery of BRT) are big losers, as they face substantial reductions in firewood (−13%) and even more in grazing benefits (−19%), and only marginal gains from soil conservation, resulting in a net loss of 3.06 million US$ (PV at 3%). Thus, an increase in natural

[16] Note that this value already suffers from a serious aggregation problem, since global damage estimation studies do not adjust for the large differences in incomes between poor and rich nations or communities.

[17] Much of the literature that recommends higher discount rates ignores the distinction between real and nominal rates. E.g., a nominal rate of 12%, often recommended by the Planning Commission of India, would be equivalent to a real rate of 3%–6%, given current rates of inflation in India.

[18] Note that the suppression of shifting cultivation and the forced re-settlement of many Soliga settlements from the interior to the periphery of the forest are assumed to be common to both scenarios, i.e., would have occurred regardless. Hence, the economic and cultural losses due to such forced re-settlement, although significant, are not factored into this net benefit.

**Table 2**
Present value of changes in economic benefits and costs for the 'normally expected trajectory (RF to WLS_1) with a 3% discount rate over 50 years.

| Stakeholders | NTFP benefits | Firewood benefits | Grazing benefits | Soil conservation benefits | Ecotourism benefits | Carbon seq. benefits | Timber & bamboo benefits | Wildlife related dis-benefits | Protection costs | Net economic impact |
|---|---|---|---|---|---|---|---|---|---|---|
| Soligas | +2.68 (+75%) | −0.03 (−7%) | −0.28 (−7%) | | | | | | | +2.38 |
| NS local poor | | −0.66 (−13%) | −2.73 (−19%) | +0.33 (+2%) | | | | | | −3.06 |
| NS local rich | | | −4.10 (−19%) | +0.97 (+1%) | | | | +0.48 (NA) | | −3.60 |
| Rest of India | | | | | +0.07 (NA) | | −1.61 (−100%) | +0.48 (NA) | +5.17 (+69%) | −7.18 |
| Global | | | | | +0.02 (NA) | +15.49 (+9%) | | | | +15.51 |

Notes: 1. Units are million US$ in 2010 prices (1$ = Rs.44).
2. Figures in brackets are % change with respect to the value in the RF scenario. When the benefit was completely absent in the RF scenario, % change could not be calculated and is shown as (NA).
3. Protection costs and wildlife-related dis-benefits are subtracted from benefits.

capital does not bring about a net improvement in their economic well-being.[19]

c) The richer group of non-Soliga households in the periphery villages are even bigger losers (3.60 million US$ PV at 3%). The substantial loss they incur due to curtailed grazing is aggravated by wildlife-induced crop damage, and the gains from reduced soil conservation are too small to offset these losses.

d) Rest of India, in the form of the Indian state, naturally incurs higher costs of protection in WLS scenarios, and these are not offset by profits and royalties from eco-tourism, thus requiring a net expenditure of 7.18 million US$ (PV at 3%). Since the primary reason that the state has initiated conservation is the admittedly non-monetizable benefits from biodiversity conservation, this may not be a matter of concern, but it is useful to note how little tourism seems to contribute to the increased sanctuary upkeep.

e) The tangible benefit to global stakeholders is in terms of averted climate change damages. On the face of it, this figure seems enormous: 15.49 million US$ (PV at 3%), which, in a conventional BCA, would outweigh all other costs and benefits in Table 2. But this figure is highly sensitive to the assumptions about marginal damages from climate change and so needs to be treated with caution.

Thus, increased forest protection or conservation does not generate uniformly positive outcomes for all stakeholders. Different stakeholder groups face different trade-offs, because some ecosystem benefits increase while others decrease, some dis-benefits increase, and different stakeholders have differing dependences on these benefits.

It is therefore not surprising that some stakeholders have resisted the conversion of RF to WLS (pers. observ. since 1994). This does not imply that they do not have any concern for the non-monetizable values generated by such conservation projects, but highlights the acuteness of material loss that they face under increased conservation.

### 5.3. Effect of time discounting

Our use of a 3% real discount rate was in keeping with conventional benefit–cost analysis of using a positive discount rate. Given our use of a finite time horizon, we are able to compute the net benefits with a 0% discount rate. We find that although the absolute values changed, there is no significant change in the distribution of benefits and costs, or in who is a net gainer or net loser. Further sensitivity analysis showed that increasing the discount rate to 6% also did not change the situation in qualitative terms. This is because there are no major temporal dynamics in the flow of ecosystem benefits and dis-benefits, and also because we have used a low discount rate. The assumptions that lead to low temporal dynamism are also assumptions about no dramatic shifts in preferences from one generation to another or thresholds that may push one generation into a qualitatively different situation from another. However, the normative question of whether discounting benefits accruing to future generations remains to be addressed.

### 5.4. WLS_2: technological, institutional and ecological 'Surprises'

The 'surprise' scenario leads to significantly different trajectories and magnitudes of benefits. The NPVs for each benefit–beneficiary combination calculated using a 3% discount rate are presented in Table 4.

Several differences from scenario WLS1 are notable. First, the loss of NTFP collection rights mid-way through the time horizon means that the Soligas also become net losers from the shift to conservation. Thus, the entire local community (Soligas and non-Soligas, poor and rich) now becomes a loser, even as the forests next door to them regenerate.

Second, there is about a 33% decline in soil conservation benefits to both the poor and rich beneficiaries. This is because the advent of canal irrigation renders many of the irrigation reservoirs irrelevant to agriculture and thereby reduces the significance of the soil conservation impacts of forest cover change.[20] However, given that the magnitude of soil conservation benefits and changes in them due to forest cover change is small, this does not affect the overall calculus of either community significantly. Nevertheless, the coming of canal irrigation is likely to reduce farmers' interest in forest regeneration (which is already at low ebb due to the loss of grazing rights).

Third, the ecological change due to the rapid spread of *Lantana* results in (among other impacts) a ~20% drop in the net gains from sequestered carbon as compared to WLS_1 (in the discounted case). Clearly, the gains from increased conservation effort are highly sensitive to assumptions about ecosystem dynamics.

## 6. Discussion

We shall now discuss our findings in terms of the issues surrounding valuation and BCA that we raised in Section 2.

[19] It could be argued that this negative impact is because their pattern of use under the RF scenario is not sustainable. But only a part of the relative loss is due to the high but unsustainable use in the RF scenario. The main reason for the loss is that they are excluded from accessing the resource, as the management shifts towards other objectives (wildlife conservation).

[20] To be precise, the aggregation of the fringe community into only two categories (landless or marginal farmers and larger farmers) no longer holds—a sharp divide emerges between those who continue to depend upon the local irrigation reservoirs (and therefore indirectly on the forests) and those who depend upon 'external' canal water.

**Table 3**
Present value of changes in economic benefits and costs for the 'normally expected trajectory (RF to WLS_1) with a 0% discount rate over 50 years.

| Stakeholders | NTFP benefits | Firewood benefits | Grazing benefits | Soil conservation benefits | Ecotourism benefits | Carbon seq. benefits | Timber & bamboo benefits | Wildlife related dis-benefits | Protection costs | Net economic impact |
|---|---|---|---|---|---|---|---|---|---|---|
| Soligas | +8.00 (+103%) | −0.06 (−8%) | −0.64 (−8%) | | | | | | | +7.30 |
| NS local poor | | −1.70 (−16%) | −6.34 (−22%) | +0.68 (+1%) | | | | | | −7.35 |
| NS local rich | | | −9.51 (−22%) | +2.05 (+1%) | | | | +0.82 (NA) | | −8.28 |
| Rest of India | | | | | +0.21 (NA) | | −2.53 (−100%) | +0.82 (NA) | +10.82 (+144%) | −13.96 |
| Global | | | | | +0.04 (NA) | +27.41 (+15%) | | | | +27.45 |

Notes: 1. Units are million US$ in 2010 prices (1$ = Rs.44).
2. Figures in brackets are % change with respect to the value in the RF scenario. When the benefit was completely absent in the RF scenario, % change could not be calculated and is shown as (NA).
3. Protection costs and wildlife-related dis-benefits are subtracted from benefits.

### 6.1. Double-counting or Mis-counting?

Ours seems to be one of the first studies to show significant ecological trade-offs and social trade-offs within local stakeholders themselves. It is instructive here to compare our results with those of the very similar study on Leuser National Park, Indonesia (van Beukering et al., 2003, hereinafter VB). VB found that a shift from logging to conservation (similar to our shift from RF to WLS) would

a) increase *all* benefits except those from timber and agricultural expansion, and the NPV of the total economic value is always higher for the conservation scenario
b) the local communities would be the biggest beneficiaries of this shift, and
c) aggregating across benefits and beneficiaries, and without applying any income-sensitive weights, the largest gains would be from increased watery supply, hydro-power generation and flood prevention that would supposedly result from forest regeneration.

Our results, on the other hand, show that not only timber benefits (which were not substantial to begin with) but several other local benefits also decline, that some local communities gain while others lose significantly, and that in non-income-weighted terms the 'global benefits' of conservation would far outweigh the local benefits or costs of conservation.

While the biophysical and social contexts are not identical, and the $ value of carbon sequestration used by us is higher by a factor of almost 10, we believe that the big difference between the two studies also highlights the problem of over-counting/double-counting/mis-counting highlighted in Section 2.1. VB appear to make highly optimistic assumptions about the relationship between forest regeneration and hydrological change that are prevalent in the environmental literature (Lele, 2009, for a critique) and use the additive method of benefit estimation, wherein each benefit stream and its change is estimated independently and added, resulting in competition, overlaps or trade-offs not being identified. This suggests that a spatially explicit model that incorporates the location of both ecological processes and social resource use patterns is needed for any meaningful estimation. This approach was partially adopted by us, and the ecological dimension is more fully developed in models such as InVEST (Kareiva et al., 2011), albeit without providing for dis-services.

### 6.2. Trade-offs and scenarios

Ecological economists came up with the metaphor of 'natural capital' to highlight the importance of the biophysical environment in the economic process. However, when more than metaphorically, it implicitly suggests that ecosystem benefits always increase with increasing conservation (just as financial returns increase with increases in financial capital). But the above results highlight the fact that ecosystems relate in different ways to human well-being, and to the well-being of different human beings. The act of forest and wildlife conservation creates complex trade-offs between different ecosystem benefits and between beneficiaries. In such a situation, the concept of aggregate natural capital is not analytically useful.

Our results also highlight the dangers of valuation in isolation, i.e., estimating TEV of an ecosystem in a given condition without specifying the alternative scenario. Not specifying the alternative unconsciously implies that 'if the ecosystem were to disappear' all values would go to zero. This is an absurd proposition for many reasons. First, the alternative is rarely complete destruction. Indeed, complete destruction is hard to define. For instance, many earth system processes such as rainfall and runoff would persist even in a barren world. Second, the complete destruction scenario would be so radically different from the existing one that assumptions of marginal change would simply not hold for most benefit streams. Projecting a realistic alternative scenario (or counter-factual) is a difficult task that requires much greater attention than has hitherto been given (Caplow et al., 2011). In our study, the actual trade-offs and synergies became apparent only when two alternative scenarios were compared, as even in the RF scenario the forests generated *some* benefits.

In theory, the problem of missing alternative scenarios occurs only in valuation studies, not in BCA studies, as BCA is precisely about estimating the difference between the presence and absence of a project. But in practice even BCA studies or studies where changes in value are estimated tend not to pay too much attention to the socio-ecological detail and dynamics of the situation. The focus tends to be on the economic estimates (see, e.g., van Beukering et al., 2003, Section 2.3). Equally under-emphasized are the social and technological details. E.g., how farmers irrigate their lands is crucial to determining whether and how they would benefit from upstream soil conservation or hydrological change, but this information is often missing from econometrically rigorous impact analyses (e.g., Pattanayak and Kramer, 2001).

### 6.3. Importance of socio-technical context and the Co-production of value

Our study highlights how sensitive the estimated value of an ecosystem is to the socio-technical context. Forests may regenerate but social institutions of property rights might prevent the enjoyment of some benefits, as when NTFP harvesting is banned, while technological change might render some ecosystem processes irrelevant, as in the case of new irrigation systems. And clearly technology and institutions shape not only which processes translate into benefits but also who gets to enjoy them and how much. This point is further

**Table 4**
Present value of changes in economic benefits and costs for the 'surprise trajectory' (RF to WLS_2) with a 3% discount rate over 50 years.

| Stakeholders | NTFP benefits | Firewood benefits | Grazing benefits | Soil conservation benefits | Ecotourism benefits | Carbon seq. benefits | Timber & bamboo benefits | Wildlife related dis-benefits | Protection costs | Net economic impact |
|---|---|---|---|---|---|---|---|---|---|---|
| Soligas | −1.09 (−28%) | −0.03 (−7%) | −0.28 (−7%) | | | | | | | −1.39 |
| NS local poor | | −0.66 (−13%) | −2.73 (−19%) | +0.23 (+1%) | | | | | | −3.89 |
| NS local rich | | | −4.10 (−19%) | +0.68 (+1%) | | | | +0.48 (NA) | | −3.17 |
| Rest of India | | | | | +0.08 (NA) | | −1.72 (−100%) | +0.48 (NA) | +5.17 (+69%) | −7.29 |
| Global | | | | | +0.02 (NA) | +12.34 (+7%) | | | | +12.36 |

Notes: 1. Units are million US$ in 2010 prices (1$ = Rs.44).
2. Figures in brackets are % change with respect to the value in the RF scenario. When the benefit was completely absent in the RF scenario, % change could not be calculated and is shown as (NA).
3. Protection costs and wildlife-related dis-benefits are subtracted from benefits.

strengthened by nuances that could not be incorporated into our analysis. In particular, till 2001 the KFD was extracting a royalty from the NTFP cooperatives of the Soligas in BRT, in effect re-distributing some NTFP value to a larger society. In 2001 the state government decided to abolish this royalty system, effectively transferring this income to the Soliga cooperatives. Simultaneously, rampant mismanagement in state-controlled cooperatives meant that NTFP collectors generally got only a partial share in the prices obtained by the cooperatives (Lélé and Rao, 1996). Conversely, interventions by ATREE and local activists in one cooperative forced improvement in management, competitive auctions and value-added processing, resulting in increases of as much as 50% in the returns to collectors (Bawa et al., 2007; Lélé et al., 2004). Thus, value and its distribution seem almost inseparable from the institutional context.

What does this context-specificity and socio-technical sensitivity imply for economic analysis of ecosystem change? First, our case typifies the situation prevailing in tropical forest contexts, viz., ill-defined property rights, thin, fragmented and non-competitive markets, and information asymmetries. For other services such as soil conservation, markets do not even exist, nor do other kinds of institutions that would better link upstream and downstream ecosystem groups (in this case hill- and forest-dwelling Soligas and downstream non-Soliga farmers). Therefore, an analysis of who gets what economic return (or bears the costs) from the ecosystem under what conditions requires a thorough understanding of the institutional, social and technical contexts in every case. While estimates of specific ecological or social parameters (productivity of a species or wage rates) may be transferrable across similar sites, the 'economic value of an ecosystem' is so integrally related to the manner in which different segments of society perceive and are connected to different aspects of that particular ecosystem that any generalization is probably meaningless. Studies that extrapolate benefit estimates from one site to another (so-called 'benefits transfer') are therefore likely to introduce major uncertainties. And attempts to estimate ecosystem values in $/ha across global landscapes using mostly remote sensing data and minimal social information (see, e.g., Naidoo et al., 2008; Sutton and Costanza, 2002) are highly problematic.

Second, at a deeper level, there is no value without a social context precisely because value is a social construct. When ecologists seek to map ecosystem value by focusing on biophysical attributes, the implicit assumption in the latter studies is that value inheres in nature. The message from this study is that value only emerges when human beings interact with nature, be it for consumptive or non-consumptive use. Value is produced by the action of human labour, technology and capital on the landscape. Ecosystems simply 'are' and processes simply occur. It is human beings who either adapt or mal-adapt to these situations and phenomena, and who are in a complex and continuously shifting relationship with them. This perspective also helps avoid the tendency to essentialize nature, a tendency that leads to ignoring situations where certain groups also suffer dis-benefits from certain ecosystem processes.[21]

Third, conventional economic analysis has focused on correctly estimating the contribution of labour and financial capital and deducting it from the final price of the product or service, and conventional sensitivity analysis has focused on varying assumptions about prices and discount rates. But perhaps more important are the institutional and technical assumptions that underpin alternative scenarios. Our WLS_2 scenario, which is based on real changes that took place after 2001, shows how the return from the harvesting and sale of NTFPs is highly contingent on property rights and the functioning of markets. Rather than 'assume away' the problem, one may have to delve into these complexities to produce more plausible estimates and scenarios. In this case, setting up the problem as simply 'sanctuary versus no sanctuary' was inadequate: the details of the WLS scenario, particularly whether Soligas rights to NTFP collection were recognized or not' was critical in determining the nature of impacts.

Fourth, if social, technological and ecological dimensions of the scenarios are as uncertain as the difference between WLS_1 and WLS_2 highlights, the case for more deliberative and participatory approaches to both economic analysis and the actual management of the ecosystem is strengthened. This is perhaps particularly acute in a tropical developing country context where on the one hand modern knowledge systems have produced limited insights (Guha and Gadgil, 1989) and on the other hand local communities such as the Soligas have been shown to be rich repositories of ecological knowledge (Rist et al., 2010).

### 6.4. Aggregation and Win–Win

Once our economic estimates are displayed in the tabular forms above, there is a strong temptation to take the final step of conventional BCA, viz., aggregating the benefits and costs to estimate net change in economic welfare of society due to increase conservation, and conclude that society is 'better off on the whole' with the conservation project. Alternatively, market-oriented economists point to the huge difference between magnitude of gains for global stakeholders and magnitude of loss for local stakeholders, thereby highlighting the potential for win–win from payment schemes (PES, Simpson, 2004). But our analysis actually cautions against quickly moving to either conclusion.

The BCA approach of simply adding benefits and costs across all stakeholders is particularly problematic in the context of tropical

[21] The problem with the essentialization of ecosystem services is discussed at length in Lele et al. (in press).

forests, given the enormous diversity of stakeholders and consequent disparity in the levels of income between local, regional and global stakeholders: in this case, approximately 1: 4: 40 in 1999 US$. Consequently, the marginal utilities of income would be vastly different across these groups. While conventional BCA would suggest that the RF to WLS_1 shift has substantial positive benefits for society (+4.3 million US$ in PV at 3%), using income-sensitive weights (as we did in a previous version of this paper: Lélé et al., 2001) indicates a net loss (−12.7 million US$). This exercise is useful to demonstrate just how sensitive the conclusion about 'net gains to society' is to assumptions about weights. But it does not take us any closer to what the 'real' net benefit might be, because the impossibility of monetizing several merit goods, such as the existence value of biodiversity or the value of eradicating poverty, makes any concept of net benefits questionable. Moreover, to use conventional aggregation to determine whether a project is societally beneficial and then to do stakeholder-wise analysis to understand why certain groups may oppose such a beneficial project (as done by van Beukering et al., 2003) is misleading. It pre-judges the distributional weights and then casts opposition to conservation projects in only negative terms, whereas, as we have shown, the opposition may be justified on the grounds of loss of legitimate livelihood opportunities, especially for disadvantaged communities. BCA thus hides much more than it reveals.[22]

Our socio-institutional analysis also cautions against jumping from estimates of economic impact to market-based solutions in the form of PES schemes. We found that even for NTFPs, which are physically tangible goods sold and transported to regional markets, the return to the NTFP collector is highly uncertain and strongly influenced by the institutional setup. Markets for indirect ecosystem services such as carbon sequestration or intangible ones such existence value involve linking remote buyers across the globe to sellers in remote tropical locations. These would then be much more risky, involving far higher transaction costs (Cacho et al., 2005) and, in the absence of secure local rights and democratic local governance of forests, will likely lead to adverse impacts on poor communities (Phelps et al., 2010), as in our WLS_2 when the Soligas were denied NTFP collection rights in the name of stronger conservation. PES is thus not just difficult to implement but problematic in its underlying assumptions about the allocation and security of rights and the governance mechanism.[23]

Several limitations remain in our analysis. First, by limiting the time horizon to two generations, we are effectively excluding subsequent generations from the analysis. However, analysing change over a longer time horizon, if it is to be meaningful, will require far better understanding of ecological and social processes than what exists right now. Second, there is clearly a need for more spatially explicit and integrated ecological model (such as the InVEST model in Polasky et al., 2011). However, such an approach runs the risk of being de-coupled from the social dimension–stakeholders and stakeholder-relevant ecological variables, etc.–that we emphasized. More work on integrating the social, the economic and the ecological dimensions in micro-economic analysis of this kind is required. Third, there is room for the deliberative valuation (not decision-making) process to be extended into the empirical analysis (Wilson and Howarth, 2002). For instance, the categorization of stakeholders is necessarily subjective, and this could be done with more consultation. Similarly, given that one can never examine all impacts (see Section 4.3), the decision about which impacts to focus on could be taken more deliberatively.

[22] We acknowledge that desisting from this last step in BCA does not completely remove the problem of aggregation. Our so-called homogeneous stakeholder groups can never be completely homogeneous. This is true of all analysis (Lélé and Norgaard, 1996): subjective choice of level of detail and scale always remain. But the stark forms of aggregation are avoided, and the focus on stakeholders makes it possible to explore the implications of different groupings easily.

[23] Apart from normative objections to the tradability of merit goods (Vatn, 2010) or commodification of nature (McAfee, 1999).

## 7. Summary and concluding remarks

We argued that, especially in the context of tropical forest ecosystems, the criticisms of BCA are sufficient to make its use unacceptable in decision-making. We proposed an approach of disaggregated stakeholder-wise analysis of monetizable or economically tractable impacts. This approach explicitly focuses attention on ecological and social trade-offs and dis-services and on the contextual factors shaping these outcomes. Our case study of the impact of switching from production to conservation forestry in a location in the Western Ghats of India has shown the following:

a) That increasing the 'forest conservation' effort has both positive and negative impacts on local communities, thus creating complex trade-offs for individual stakeholders and across stakeholder groups.
b) That different conservation trajectories based upon different protection strategies can have significantly different economic impacts on local stakeholders.
c) That the economic impacts and their distribution (trade-offs) change significantly depending not only on the discount rate, but also on key assumptions made about property rights, technology and ecological dynamics. Thus, analyses based on benefit-transfer seem inadvisable, and conventional sensitivity analysis should be supplemented by grounded sensitivity analysis that examines and tests key assumptions about larger contextual factors.

In sum, the earlier idea of TEV of ecosystems and its new avatar of ecosystem service valuation have made an important contribution to our understanding of the relationship between ecosystems and societal welfare, particularly by highlighting the role of indirect and intangible benefits from the environment (Lele et al., in press). However, attempting to come up with one TEV estimate or estimating 'net societal benefits' using BCA, and then arguing that this is but one input to political decision-making processes (Daily et al., 2000) is questionable. At the same time, purely deliberative valuation and decision-making also have limitations. A stakeholder-wise economic impact analysis that is self-aware of the limits of monetization and aggregation, and that is more socially and ecologically grounded and aware of trade-offs may provide a useful middle path.

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