

Ecosystem Services Economics (ESE)

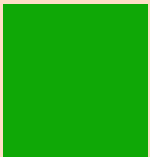
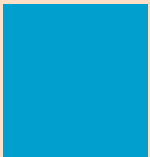
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The “Ecosystem Service Framework”: A Critical Assessment



R. David Simpson

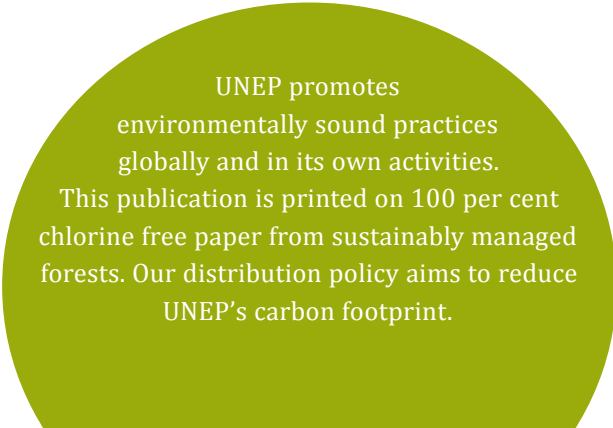


January 2011

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Ecosystem Services Economics

The “Ecosystem Service Framework”: A Critical Assessment

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January 2011



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Preface

In last ten years, the economic analysis of ecosystem services has caught the imaginations of the decision-makers. Economic values of ecosystem services at global level have been in the centre of debate. The Millennium Ecosystem Assessment (MA) and recently concluded, the Economics of Ecosystems and Biodiversity (TEEB) have provided adequate rationale for economic approach to management of ecosystems. However, robust estimation of ecosystem services whether it is for payments for ecosystem services or for national level green accounting are scarce and inadequate. The paper provides a critical angle to the whole debate. The author questions both the justification for adopting an "ecosystem service framework" and the extent to which some prominent examples of valuation of regulating services can withstand detailed scrutiny. The views expressed in the chapter may be seen as controversial by many readers, but they highlight some of the fundamental theoretical and methodological considerations that underpin this field of ecological economics.

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Abstract

Natural ecosystems provide services to society. Conservation advocates have shown considerable enthusiasm in recent years for an "ecosystem service framework". They hope that an appeal to the value of ecosystem services will result in greater funding. Yet it is not unusual to find examples in which an author states at one point that the value of ecosystem services is great, while shortly thereafter stating that values have not yet been quantified.

The values of ecosystem services have not yet been estimated with any generality or precision. While several celebrated studies have addressed valuation, most are problematic. Moreover, there are a number of reasons for which valuing ecosystem services remains a daunting exercise. Foremost among these is a "paradox of valuation": it would be most useful to estimate the value of ecosystem services that are dispersed over a very broad public, but it is precisely under these circumstances that it is most difficult to use the tools of economic value estimation.

How, then, does an "ecosystem service framework" affect conservation strategy when the value of ecosystem services remains uncertain? It certainly motivates more research. Unless and until such research yields more precise and compelling evidence of values, however – and there is little reason to be optimistic that it soon will – the *status quo* will likely prevail in conservation policy. That is, conservation will be largely motivated by the importance of providing global public goods (largely carbon sequestration and biodiversity protection), and will require international transfers from wealthier to poorer countries.

1. Introduction

In recent years considerable enthusiasm has grown for an "Ecosystem service framework" (Daily and Turner 2008) to conservation policy. Gretchen Daily and Pamela Matson write that such an approach has sparked

... a growing feeling of Renaissance in the conservation community. This flows from the promise in reaching, together with a much more diverse and powerful set of leaders than in the past, for new approaches that align economic forces with conservation, and that explicitly link human and environmental well-being. And this promise is flowering thanks to substantial recent advances in key areas of inquiry, such as ecology, economics, and institutions, and their integration. (Daily and Matson, 2006, references omitted)

Many hope that the ecosystem services framework will provide a new and generous source of conservation funding. Heather Tallis and Peter Kareiva (2005) write that "realization of the market worth of ecosystem services has the potential to increase conservation funding by orders of magnitude."

This enthusiasm has sparked an impressive volume of work within the ecosystem service framework (Daily and Turner 2008). Fisher and coauthors (2009) document an exponential increase in the number of published papers employing the terms "ecosystem services" or "ecological services," beginning from essentially none in the early 1980's to more than 250 in 2007, the last year for which they have data. Some rough idea of the currency of the term can be gleaned from the fact that entering "ecosystem services" in the Google search engine returns about 4.7 million entries.¹ The Millennium Ecosystem Assessment, a multi-year, multi-million dollar international undertaking involving over 1,300 scientists from around the world was conducted to assess the consequences of ecosystem change, and consequent alterations in the flow of ecosystem services, for human well-being (MA 2005). This work may continue under a recently proposed "Intergovernmental Platform on Biodiversity and Ecosystem Services" (IPBES), modeled on the Nobel-prize-winning Intergovernmental Panel on Climate Change (IPCC). In November of 2008 representatives of 78 nations and 25 international NGOs met to consider establishment of an

¹ Search conducted 17 April 2009. To get some idea of societal priorities and the zeitgeist, this is about half the number returned when one searches for "Julia Roberts", and slightly more than are returned for "Matthew McConaughey".

IPBES (UNEP 2008a). At the meeting an approximately US\$18.4 million "Programme of Work and Budget" was presented (although not yet adopted; UNEP 2008b).

Organizations around the world are adopting an ecosystem services approach to ecological decision making. Yet the elements of that approach are not as settled as its widespread adoption might make it appear. One often encounters passages such as the following: "Although the societal benefits of native ecosystems are *clearly immense*, they remain *largely unquantified* for all but a few services" (Ricketts, *et al.*, 2004, emphases added; see also Kareiva and Ruffo 2009; Daily *et al.* 2009). But if benefits are "largely unquantified" what is the basis for concluding that they are "clearly immense"? Is there really much evidence supporting the contention that the services supplied by natural ecosystems are of great value and that they are being squandered by unwise land use decisions?

In this paper I suggest that evidence to that effect remains sparse. This is so for several reasons, and I will consider them in turn. The first is that many of contributions to the literature on ecosystem service values would appear to be intended to motivate research on ecosystem services than rather than to document the findings of such research. There are certainly numerous, and in many instances eloquent, statements of the *hypothesis* that natural ecosystems provide valuable services, but fewer careful tests of that hypothesis.

In some studies the interpretation of evidence concerning ecosystem service values is problematic. It is worth underscoring that evidence of ecosystem service values will only motivate different conservation decisions if such values outweigh costs. It is not sufficient simply to note that there is *some* value to conserving what is already in place without comparing that value to other possibilities.

Closely related to the above observation is the economic truism that "value is determined on the margin". The relevant concern is typically not that biodiversity or ecosystem services will perish in their entirety. Any monetary estimate of such a calamity would necessarily be, to borrow Michael Toman's (1998) characterization, a "serious underestimate of infinity". The relevant policy question, then, is whether preserving *specific components* of ecosystems provides benefits in excess of those that would arise from their forgone uses.

If ecosystem services are not assigned their true value in land use decision-making, it is because such services are *public goods*; that is, they are benefits which, when supplied by one person are necessarily accessible to many. Yet the more compact the scale on which such public goods are provided – the fewer the members of the "public" are who benefit from their provision – the less likely it is that ecosystem services will be underprovided. Consequently, we should be most concerned about the provision of ecosystem services whose benefits are very widely dispersed.

This observation, in turn, leads to a couple of other issues. First, there is what I call below a "paradox of valuation". The things we would most like to be able to place an economic value on are those public goods whose benefits are the most widely dispersed. But these are precisely the goods whose value is most difficult to estimate. Second, the most compelling argument for conserving relatively pristine ecosystems may prove to be that they provide the global public goods of carbon sequestration and biodiversity protection. If this is, in fact, the most important argument for conservation, however, the "ecosystem service framework" does not appear to be adding much new to the debate on conservation policy.

This last observation poses the main question motivating this paper. How *does* adopting the "ecosystem service framework" alter the ways in which we think about conservation policy? If the point is simply that we ought to regard natural ecosystems as assets that provide value to society and should, therefore, compete with alternative land uses as we make choices about how to allocate the earth's surface among our wants and needs, the argument is unexceptionable. It is also not novel, however. Underscoring such a fundamental principle is useful, but it alone cannot account for current enthusiasm for the ecosystem service framework.

That enthusiasm derives, rather, from the sense that the ecosystem service framework has already demonstrated, or can soon be expected to demonstrate, the general economic superiority of conservation to alternative land uses. Most of the remainder of this paper is devoted to considering and, generally, rebutting these assertions. As these are controversial points, I should hasten to point out a handful of caveats. I most certainly do not dispute that the services of natural ecosystems are valuable to humanity. Nor do I dispute that the services of some such ecosystems

are *more* valuable than are any alternative uses that might be made of the areas they occupy. Consequently, there are surely instances in which land use could be made more rational and, generally, socially beneficial by undertaking public policy to preserve natural habitats.

What has not been satisfactorily established is the generality of such propositions. In the body of this paper I pursue two major themes. The first is that many well known tracts on ecosystem services do not, in fact, make a general and compelling case for their economic value. The second is that there is a good reason for the first observation: valuing ecosystem services is an extraordinarily difficult undertaking. In the final section of the paper I return to my main question in light of these observations. What *does* an ecosystem service framework imply for conservation policy, and does adopting it move the debate forward?

2. The value of ecosystem services: Some prominent examples

In this section I review some prominent examples of empirical work on the economic valuation of ecosystem services. The point I wish to make with this review is that some of the works on which the perception that ecosystem service values are important and significant are controversial. They do not establish a *prima facie* presumption that the benefits of ecosystem preservation exceed those of alternative use. Nor, it should be pointed out, does the selection of a handful of studies whose methods can be faulted or the generality of whose conclusions can be disputed establish that ecosystem services are *not* valuable or important. It does, however, suggest that we should consider more carefully the empirical evidence for that assertion. This is the inquiry to which I turn after discussing a handful of prominent examples.

The millennium ecosystem assessment

Publications discussing ecosystem services and their value often reference the recently concluded Millennium Ecosystem Assessment ("MA"; see, e.g., MA 2005). Despite the characterizations one sometimes reads of the MA, the reader should appreciate a few things about its purposes and results. First, the Millennium Ecosystem *Assessment* did not conduct and report new research. Rather, as the term I have emphasized suggests, it *assessed* work already in the literature. Second, the MA was, in the final analysis, restrained in its assessment. It did not make as bold claims for ecosystem service values as have appeared elsewhere in the literature.² Finally, some commentators appear to have confused the MA's suggested *definitions* and *classification schemes* for *estimates* of value. The MA proposed a division of ecosystem service values among provisioning, regulating, cultural, and supporting services. While this taxonomy may be useful,³ definition and estimation remain very different undertakings. The reader must, then, look elsewhere for empirical estimates of ecosystem values.

"The Value of the World's Ecosystem Services and Natural Capital"

Perhaps the best-known example of ecosystem service valuation remains a paper published in *Nature* in 1997 by Robert Costanza and twelve coauthors. In *The Value of the World's Ecosystem Services and Natural Capital* Costanza *et al.* suggested that a "minimum estimate" of such values was US \$33 trillion.⁴ I will consider in somewhat more detail the "benefit transfer" approach taken by Costanza *et al.* below. For now, however, I will just note that the

² A summary of the MA's findings is often reported as that 15 of 24 of the ecosystem services studied in the MA are in decline or are being degraded. While perhaps cause for considerable alarm, this is not a statement concerning the value of ecosystem services.

³ Taxonomic and definitional matters are not yet settled, however. No fewer than five definitions of "ecosystem services" have been proposed in the literature (see Costanza, et al., 1997; Daily 1997; MA 2005; Boyd and Banzhaf 2007; Fisher et al. 2007). Moreover, it is not clear to me what the last decade's consideration of "ecosystem services" has added over and above earlier suggestions for the classification of "total economic value" of natural systems as proposed by the late David Pearce and R. Kerry Turner (1992).

study set off a fire storm of criticism, particularly, albeit not exclusively, from economists (see Ayres, 1997; Smith, 1997; Toman, 1998; Freeman, 1998; and Pearce, 1998). If the point of Costanza, *et al.*, were to demonstrate the value of natural ecosystems in sustaining life, it delivered, as Michael Toman (1998) noted, "a serious underestimate of infinity". If, on the other hand, it was intended to provide an economically meaningful measure, it reported a logical impossibility: a *willingness* to pay that exceeded world income which would, in turn, limit the *ability* to pay (Smith 1997). While it might reasonably be argued that this logical oversight was more semantic than substantive,⁵ a more fundamental criticism of the Costanza *et al.* (1997) work is that it confused marginal and total values. In economic theory the value of a good is determined by how much benefit we receive from a little more of that good, phrased in terms of the other goods we might give up to obtain it. Costanza *et al.* (1997) have often been criticized (see, recently, for example, Plummer 2008) for applying values obtained for ecosystems at one place and supposing that such values could be applied to superficially similar but very differently situated systems at other places. Since Costanza *et al.* (1997) continues to be widely cited – and some subsequent work could be subjected to the same criticisms (see, e.g., Mates 2007), readers of work on ecosystem services may be surprised to discover that most economists who have reviewed this work have found it seriously defective.

"Economic Reasons for Conserving Wild Nature"

Motivated in part by the criticisms offered of the work by Costanza *et al.* (1997), some of the same authors joined with Andrew Balmford and others to produce another piece that was published in *Science* in 2002. Balmford *et al.* (2002) attempted carefully to identify studies that reported the marginal value of land retained in natural habitat and compared it to the projected value of the same land if it were converted to more intensive use. This study has also been widely cited, but is also problematic in several respects.

The first concerns coverage and representativeness. Balmford and coauthors reported that they had surveyed over 300 case studies, but ultimately identified only five which compared the benefits of ecological preservation with its opportunity costs and used broad and generally accepted measures of opportunity costs. Of the five studies identified, two were not published in peer-reviewed journals (despite having been written, respectively, eight and four years before Balmford *et al.* 2002). It is open to question whether such a limited sample is in any way representative of broader circumstances in conservation policy.

Moreover, two of the five studies cited relied in part on the ecosystem service of carbon sequestration to generate values in excess of the opportunity cost of habitat preservation. The inclusion of such values is entirely appropriate if the objective is to demonstrate that the *global* benefits of conservation exceed the *local* costs. However, the international community has, by and large, still not developed a consistent set of rules for crediting developing countries for carbon sequestered in forests and other ecosystems.⁶ This observation begs the question of what an emphasis on ecosystem services is intended to accomplish for international conservation. If the objective is to convince local communities to preserve natural habitats because of the benefits they will confer on themselves directly, work such as that by Balmford *et al.* (2002) does not necessarily make the case (see also Kremen, *et al.* 2000, Naidoo and Ricketts 2006). If the objective is to convince global donors that "purchasing" ecosystem services from developing countries would be wise, work such as that by Balmford and his coauthors may be more compelling, but this argument has been made in the conservation literature for many years (see, e.g., Pearce and Moran 1994).

⁴ Costanza *et al.* derived a range of value estimates from \$18 to 54 trillion, but, because of exclusions, suggested that the "average" estimate of \$33 trillion was a lower bound on actual values.

⁵ Willingness to pay is, in economic theory, the amount someone would and can pay for a some good or collection of goods. It is necessarily limited by the resources at one's disposal, i. e., wealth. There is not anything necessarily logically inconsistent about saying that the benefits we receive "for free" from nature are of equal or greater subjective value to us than those we purchase in markets.

⁶ Developing such guidance is the mission of the REDD (Reduced Emissions from Deforestation and Degradation) programme.

"The Catskills Parable"⁷

The next example for consideration is one of the most often-cited pieces of "evidence" for the importance of ecosystem services. In the late 1990's New York City sought to secure the safety of water provided from its reservoir in the Catskills outside the city. In a widely praised (see, e.g., Levin 1999) and cited, albeit brief, piece in *Nature*, economists Geoffrey Heal and Graciela Chichilnisky (1998) argued that the city had avoided the substantial capital and operating costs of a new water treatment plant by devoting a fraction of their expense to the preservation of natural habitat in the vicinity of the reservoir. While this example has often been cited as evidence of the value ecosystem services provide, it is problematic on several counts. First, the city faced the requirement to construct a treatment plant based on a *regulatory* rather than an actual demand. The United States Environmental Protection Agency required that municipalities drawing from surface sources either treat their water or obtain a waiver from the regulatory requirement. There was no evidence that New York City's drinking water was unsafe or required treatment (NRC 2000; Sagoff 2002). While one positive aspect of maintaining more "natural" habitat for wildlife around the reservoir was that it could then support more wildlife and biodiversity that some constituents valued, there was no evidence that citizens were willing to pay extra for water in order to support wildlife conservation, an observation made at the time by some advocates of biodiversity protection (Salzman, *et al.* 2001).

Moreover, the assertion that what happened in the Catskills approximated a situation in which payments were made for ecosystem services is not well supported by the evidence. Mark Sagoff (2002) reports that while the 1997 Memorandum of Agreement between the EPA and the City called on the latter to solicit purchase of 355,000 acres of land in the watershed, by February of 2002 only a little over 19,000 acres had been acquired, most of them in the form of conservation easements restricting use, rather than of fee simple purchase. Environmental organizations expressed dissatisfaction with the pace of acquisitions (Sagoff 2002). On the other hand, however, some efforts to restrict land use without what were regarded by owners and developers as adequate compensation were decried as "thievery" (Daily and Ellison 2002). However one characterizes what happens in the Catskills, it seems reasonable to conclude that it was not what it has often been described as: a set of large-scale transactions entered into voluntarily to realize the preservation of important ecosystem services.

Perhaps the most troubling aspect of the Catskills Parable is the lack of scrutiny to which it has been exposed by those who cite it as evidence of the importance and value of ecosystem services. While the episode is widely cited in the literature on ecosystem services, tracing such citations back to their origin leads one to the 2-page 1998 Chichilnisky and Heal *Nature* article, a piece that lists no references, but would appear to contain significant and material factual misstatements (Sagoff 2002). It is troubling that so slight a piece that has been so influential in the discourse about ecosystem services. This provides a useful cautionary tale. One should be careful in the "evidence" she cites to support the importance of ecosystem services.

Vittel

The next two examples are, I believe, more substantively valid, but each begs an important question. The first is of the Vittel water bottling facility near the Vosges Mountains in northeastern France.⁸ The company bottles water from a spring in the region, which it then markets worldwide.⁹ In recent decades the quality of water from the underground aquifer in the area has been threatened by more intense cattle operations in the region. The presence of the cattle could lead to increased nitrate in the groundwater and, eventually, a decline in the quality of water from the spring. As the process by which the nitrate would enter groundwater is well understood, as is the relationship between nitrate and cattle ranching operations, Vittel readily perceived how it could maintain the value of its franchise by compensating local ranchers for modifying their practices. Consequently, it entered into an arrangement to do so.

⁷ I have borrowed this designation from the title of Mark Sagoff's (2002) paper, on which I have drawn heavily.

⁸ I rely heavily on Perrot-Maitre (2006) for this paragraph.

⁹ One might also pause in passing to speculate as to the aggregate ecological consequences of maintaining the purity of a spring so that its water can continue to be shipped to customers as many as 10,000 kilometers away

The Vittel example has often been cited as an example of the importance of ecosystem services (see, e.g., Ranganathan, *et al.*, 2008; U. S. Forest Service 2007, Perrot-Maître 2006). Indeed, it does seem to be a very apposite case. While it does not speak to the difficulties or methods of valuation *per se*, it is clearly an instance in which improved environmental performance would create large enough gains for the beneficiary to induce it to compensate the party that restricts its options in order to supply such benefits. The only question is whether this is an exception that proves a rule. *Reducing* the intensity of agricultural use is perhaps a step back toward a more "natural" ecosystem, but is a far cry from adopting measures to preserve relatively pristine landscapes. The Vittel example is a good one of the benefits of less intensive use of the landscape. Inasmuch as even this relatively straightforward effort to conclude a transaction in relatively minimal "ecosystem services" was difficult to consummate, it begs the question of what realistically are the prospects for concluding deals for still more difficult, diffuse, and diverse ecosystem services.

Pollination on the Finca Santa Fe

Another case study that briefly (we will see momentarily why interest was transitory) attracted interest among practitioners and advocates occurred on a coffee plantation in Costa Rica. One ecosystem services that has attracted considerable attention has been pollination. Areas of retained natural habitat in the vicinity of agricultural fields provide refuge for insects and birds that may pollinate agricultural crops. In a carefully conducted study Taylor Ricketts and his colleagues (2004) demonstrated that coffee yields were about 20% greater on plants within one kilometer of retained forest areas, presumably because they are better served by native pollinators.

This observation begs a question. All 480 hectares of habitat located within one kilometer of preserved forest habitat in the study by Ricketts, *et al.*, were owned by a single owner. If it were, in fact, the case, that the value of coffee production on the plantation would be higher if 157 hectares of adjoining forest were preserved, what would prevent the owner of the coffee plantation from acquiring the forest and preserving it for his own benefit? In fact, however, careful consideration of the numbers reported in the Ricketts, *et al.* (2004) study suggest that the coffee plantation owner might have been better off if, rather than acquiring the 157 hectares to assure that it remained in forest, he had instead acquired the land, cut the trees, and planted more coffee on it. While yields might have declined elsewhere on his land, the increase in production on the newly cleared land would have more than compensated for it. As it turned out, events have revealed a related phenomenon. Douglas McCauley (2006) reports that the coffee plantation was subsequently uprooted and replaced by a pineapple farm. Pineapple does not require insect pollination at all.

The point of this and the above examples is not to suggest that ecosystem services are not important. They surely are in aggregate, and they may well be "on the margin" in a number of situations of policy interest. The point, rather, is that cases in which such values are most easily demonstrated often lead to unsurprising and not very illuminating policy implications. The value of ecosystem services can be most easily estimated when simple, often private transactions can be structured to assure that they are maximized. Conversely, the cases of greatest policy interest arise precisely when the benefits of ecosystem services are diffuse and most difficult to measure.

3. Impediments to estimating ecosystem service values

In this section I discuss why the value of ecosystem services is so difficult to measure, as well as why some of the published estimates of such values might be taken with a grain of salt.

The production function approach

A number of recent papers and studies have suggested that researchers take a "production function" approach to the estimation of ecosystem service values (see, e.g., Boyd and Banzhaf 2006; Daily, *et al.*, 2009; SAB 2009). Under this approach natural ecosystems would be regarded as an element of "machinery" that can, in combination with

manufactured capital, labor, and other such inputs, produce outputs of value to humanity. Such processes might generate products for sale, such as agricultural commodities, lumber, or fish, or through the process of "household production" they might create goods and services such as pleasant household settings or family health that are valuable even though they are not generally traded in markets.

The economic logic behind the "production function" approach is impeccable. The value of an input to production is, from elementary economic theory, the price¹⁰ of the output in whose production it is employed times the marginal product of the input in the production of the output. The problem arises, however, when we have to impute such a price in the absence of market data. Empirical economists typically cannot observe the marginal product of an input in the production of an output. The thought experiment implied – "What would happen to the production of good Y if we varied the quantity of good X_1 by a small amount *while holding the quantities of goods X_2, X_3, \dots, X_N constant?*" – can rarely be conducted in practice.

The alternative is to specify a production function – positing a functional relationship between the quantities of inputs employed and the quantity of output¹¹ produced – and estimate it employing statistical techniques. This approach has indeed been used, both in the general empirical economic literature and in work on ecosystem services (see, e.g., Acharya and Barbier 2001). Several comments and caveats are in order, however. I begin with some of the more basic.

The first is that estimating production functions has long been regarded as problematic by empirical economists. The reasons for their caution are instructive in the context of ecosystem services. When an economist estimates a production function she must make certain assumptions about the variables that she assumes affect production. Naturally enough, if she wants accurately to estimate the effects of particular inputs on production, she must suppose that she has accounted for all relevant factors. If she has not, it is likely that some of the factors she presumes to account for higher levels of production in one area relative to another are not actually contributing to production but, rather, capturing the effect of some other variables that has not been considered explicitly.

The canonical example of this phenomenon is the application of fertilizer in agriculture.¹² If an econometrician were to estimate the effects of capital, labor, and fertilizer applications on agricultural outputs, he might well find that the effects of fertilizer appear to be negative: farmers who are observed to apply more fertilizer harvest less output than those that use less. Why would this be? Are the farmers making foolish choices, paying to buy more of an expensive input that is actually hurting, rather than helping, them?

Not necessarily. A more plausible explanation is that farmers who apply more fertilizer are attempting to make up for the absence of another factor that has not been measured, and are not fully able to do so. Perhaps farmers who have inherently less fertile soil apply more fertilizer.

I suggested that this example is instructive in the context of ecosystem service value estimation. The simplest solution to this problem of "unobserved" variables would be to make the effort to observe them. Yet this is not the solution typically prescribed by econometricians. Rather, it is more common to employ a bit of theoretical sleight-of-hand to obviate the problem. The problem of unobserved variables arises because some of the explanatory factors that *are* observed are correlated with some that are not, and consequently "pick up" the effect of the unobservable variables. If it were possible to express the production relationship as a function of variables that are uncorrelated with any unobserved factors, the problem would be solved. This is exactly what econometricians try to do. They typically prefer to estimate "cost functions" or, better yet, "profit functions". A cost function relates a producer's cost of production to the *prices* of the inputs he purchases and the *quantity* of output he produces. A profit function

¹⁰ More generally, we might say that the value of an input in utility terms is the *marginal utility* afforded by the output times the marginal product of the input in its production, but a still more fundamental result of economic theory holds that the price of any product is determined by the ratio of the marginal utility it affords to that of some other good denoted as *numeraire*.

¹¹ More generally, we might say positing a relationship between the quantities of inputs employed and the quantities of outputs produced, but I'll keep the example simple for now.

¹² I base this statement on the fact that this was the example my instructor employed when I took my introductory econometrics course some thirty years ago.

relates profit to the prices of inputs *and* output. As such prices are typically not correlated with local production conditions, estimation can proceed.¹³

The point I want to emphasize here is that if including variables such as the effects of ecosystem services on production were easy, empirical economists would have done so a long time ago, as the need to develop more sophisticated methods of estimation would have been obviated. Of course, virtually any worthwhile empirical undertaking will be difficult. So, the relevant question is whether these difficulties have been, or will be, overcome in empirical work.

A paradox of valuation

Even indifferent students of economics are likely to remember the “paradox of value”: why is water, which is so essential to life, so cheap, while diamonds, which have such limited, and largely ornamental, uses so expensive? The answer is that water is (generally) abundant with respect to the uses to which it is put, while diamonds are (generally) scarce relative to the demand for them. The exceptions prove the rule: someone dying of thirst – experiencing an extreme scarcity of water – would surely trade all the diamonds he had for a drink.

One might propose a corollary, a “paradox of valuation”. If something is more or less valuable depending on whether it is more or less scarce, then the only way to place a value on a good that does not have an evident market price is to identify some circumstances under which it is more scarce and others under which it is less so. Now this will be relatively easy to do if the public good whose value we are trying to establish is relatively local; that is, it has a discernable effect on those who are close enough to its source as to enjoy its benefits, but others who are more distant receive only negligible benefits.

If the benefits afforded by such a public good are localized, however, it begs the question as to whether the good in question ought to be considered “public” at all. If the radius of dispersal is compact enough, why would it not make sense for affected parties to merge their interests and “internalize the externality” by placing the source and the beneficiary of the good under common ownership? This is essentially what Vittel did in the example cited above; it “internalized the externality” of cattle ranching in its watershed by purchasing a portion of local ranchers’ land use rights. For ecosystem services dispersed over somewhat broader areas a number of authors have documented instances in which local communities evolve rules for their management (see Ostrom 1991; Balland and Platteau 2002; more generally, Coase 1960 discusses ways in which affected parties might reach agreement for the provision of public goods or private goods with extensive externalities).

There are also instances in the literature in which it would appear that private parties may be behaving optimally already with respect to the provision of local, at least, ecosystem services. This would appear to have been the case in the pollination services example developed by Ricketts, et al., (2004) and discussed above. In a similar study Thorsnes (2002) finds that construction lots abutting retained natural areas command modestly higher prices than those at a greater distance away. It would appear, however, that, absent legal restrictions against doing so, residential developers would make more money by clearing more land for construction than they would lose by reduced sales prices on lots now rendered less attractive by the reduction in adjoining natural areas.

When there is enough spatial variability to facilitate easy measurement of values, then, it seems reasonable to suppose that local people would develop an appreciation for such values and evolve institutions to preserve them. The opposite situation is more problematic from a policy perspective. What if the services provided by natural

¹³ Explaining the reasons for this gets complicated. Basically, principles of “duality” assure that production processes may be expressed equivalently by relating the quantity of output to the quantities of inputs or by expressing the cost (or profit) associated with production to the prices of inputs and the quantity of output, in the case of the cost function, or prices of inputs and output in the case of the profit function. There is also a general preference for estimating profit rather than cost functions, as the quantity of production may also be affected by unobserved factors. For this reason much of the early empirical work on estimating cost functions was conducted on regulated public utilities, on the argument that output was determined by regulatory authorities rather than by factors that might also affect costs.

ecosystems are provided to very large publics? This would be the case if the intensity of service provision does not vary greatly over the landscape. There would, then, be little variation, and little prospect for inferring value from observing differences in the volume of service flows received from different areas. In extreme instances, such as the sequestration of atmospheric carbon dioxide and the preservation of endangered species valued for moral and aesthetic reasons, the public goods are truly global and it would be impossible to value them by observation of cross-sectional variation.

Moreover, even if it were possible to identify *some* variation in the intensity with which ecosystem services are provided across the landscape by different sources, it becomes increasingly difficult to disentangle the effect of any particular wetland, forest, meadow, etc., from that of the multitude of others that would influence production if the benefits of each do not dissipate relatively quickly in the distance from source to receptor areas.

Thus, there is a "paradox of valuation". We would like to know the value of services whose benefits accrue over such large areas of the landscape that efficient arrangements for their private or communal provision are unlikely to evolve. Yet these are exactly the circumstances under which the exercise of valuation will prove most difficult.

Spatial correlation and endogeneity

The estimation of production, cost, or profit functions is often forgone for a related technique: hedonic estimation. A hedonic model is one in which the value of some good or asset – such as a hectare of land – is determined by its attributes.¹⁴ Two other fundamental results from economic theory are, first, that the rental value of an asset, such as a hectare of land, is determined by the profits that can be earned from using that land in its "highest and best use", and second, that the purchase price of such land is determined by the net present value of the stream of rental earnings that could be realized from its ownership. So, the value of ecosystem services should be reflected in the rental and purchase prices of land receiving them.

There is a long history of hedonic studies of land value based on the uses made of nearby lands []. Some of the statistical issues discussed above reappear with a vengeance in this context, however. Since it is often extremely difficult to measure the actual services provided to one parcel of land relative to those provided to another, a more common empirical procedure is to estimate the value of land as a function of (among other factors) proximity to features of the landscape such as wetlands or forests. This raises a difficult issue. Reliable estimation of the empirical relationships between land values and the factors that explain them requires that *all* of the latter be included in the analysis. If they are not, some of the other "explanatory" factors included may not themselves explain variations in value so much as proxy for other factors that are not included. This becomes particularly problematic when land values are determined by similar factors in adjoining or nearby places in the landscape.

The essential points to appreciate here are that 1) the goal of a hedonic pricing exercise is to understand how land *value* at one point in a landscape is determined by land *use* at another; 2) it is more likely that more intensive *use* will be made of land when the features of its location imply that greater value will arise from more intensive use; and 3) the value-determining features of the landscape are likely to be highly correlated for properties that are relatively close to one another. This combination of considerations will likely imply that the benefits of ecosystem services will be *underestimated* in hedonic studies. The reasoning encapsulates the points above. Consider a farm located adjacent to an area of retained natural forest. The value of that farm may well be lower than that of a similar farm situated amidst other farms and far from remaining forest areas. Both the isolated farm and the forest adjoining it are likely situated in areas identified as having been less advantageous for farming.

In recent years econometricians have derived techniques for dealing with the combined effects of correlation in values across space and omitted variables. Such techniques presume the existence of adequate "instrumental variables" –

¹⁴ "Hedonic" is derived from the same Greek root as "hedonism", and, like the latter term, relates to "pleasures". A "hedonic" valuation exercise attempts to estimate the price of a good by the "pleasures" or, more generally, "attributes" its ownership offers.

factors that may explain the decision not to devote nearby land to more intensive use while not affecting the value of land that has been placed into more intensive use. Such variables are difficult to identify, however. While work such as that of Irwin and Bockstael (2002) illustrates the approach, results cannot yet be considered definitive.

Taking the "Con" out of Econometrics¹⁵

Despite the conclusion of the last paragraph, most hedonic studies relating land value to the retention of relatively pristine ecosystems show that proximity to ecosystem services affords positive benefits. Inasmuch as the factors explicated above would seem to suggest that estimates of value would be biased *downwards* by the factors considered, one might suppose that existing studies make the case that ecosystem services are valuable *a fortiori*.

There are, however, other concerns. It is difficult to do controlled experiments in economics on a temporal, spatial, social, and financial scale relevant to the resolution of important policy questions. Thus, rather than having the researcher control the circumstances under which experiments are conducted, as would be the case in laboratory sciences, economists must typically look for "natural experiments" in which circumstances have fortuitously crafted situations in which variables of policy interest – in our case, areas preserved for the provision of ecosystem services – differ while other relevant circumstances remain relatively constant. Perfect natural experiments are vanishingly rare, however, and researchers must instead attempt to correct for the influence of variables they cannot control directly.

This is typically done in practice by supposing that there exists a functional relationship between all of the explanatory variables of interest and the dependent variable (in the case of a hedonic model, the price of a property). The empirical procedure works then by seeing how well the function "fits" the data.

This procedure is problematic, however, as it does not comport well with the statistical theory on which empirical procedure is supposedly based. That theory presumes "hypothesis testing". The researcher forms a hypothesis – for example, that property values increase with the proximity of a property to a wetland – and can test this conjecture by seeing whether or not the estimated effects of proximity are greater than those that might arise simply from random variation in the data.

In testing such a hypothesis, however, researchers inevitably must posit certain "maintained hypotheses": that the set of other explanatory variables they have included is appropriate and complete, that the functional form they have chosen is correct, and that the spatial and, in some instances, temporal coverage of the data is adequate and representative. It is, however, very unusual for a researcher to collect a set of data, perform a single statistical procedure on it (in the parlance of the discipline, a single "regression"), and report the results of that procedure. Rather, it is common for researchers to experiment with a number of different "specifications" – typically, different functional forms, combinations or variables, and/or special techniques – and report only a subset of the results obtained from doing so. To give an example, Robert Costanza and his coauthors (2008) in their study of the value of coastal wetlands in protecting inland areas against hurricane damage, report that they experimented with nine different models, but report results for the one that, in their judgment, best fit the data.¹⁶

There is widespread agreement that this type of "pre-test bias" is prevalent and problematic (see, e.g., Leamer 1978; Kennedy 2002; Glaeser 2007), although there is probably less agreement as to the severity of the problem and the appropriate solution. There are certainly examples from other areas of economics in which researchers have exposed the sensitivity of results to specific assumptions (see, e.g., Easterly, *et al.*, 2004).

¹⁵ The title of this subsection is borrowed from Edward Leamer's 1978 paper (and plea) "Let's Take the 'Con' out of Econometrics".

¹⁶ My point in citing this work is not to criticize the authors so much as to credit them for having been more forthright than are many of their peers in describing their procedures.

The battle of the Bayesians

Even if researchers do violate the principles of statistics in conducting “specification searches”, is there a reason to suppose their results will be biased in predictable ways? There may be. Edward Glaeser writes:

Economists are quick to assume opportunistic behavior in almost every walk of life other than our own. Our empirical methods are based on assumptions of human behavior that would not pass muster in any of our models . . . While economists assiduously apply incentive theory to the outside world, we use research methods that rely on the assumption that social scientists are saintly automatons. (Glaeser 2006)

What are the incentives that might motivate economic researchers? One that has often been suggested (see, e.g., Pearce 2005) is *publication bias*.¹⁷ Academic economists obtain recognition and advancement in their field by publishing in scholarly journals. It is seen as almost a *de facto* requirement for publication that one find a “significant” – in the statistical sense of that term¹⁸ – result. It may then, be that journals disproportionately accept papers which report statistically significant results (suggesting that they may then reject large numbers of papers that do not report such results). Of course, if we *do* apply economic theory to economists, it seems unlikely that large numbers of economists would spend the time and effort required to write up results they knew would likely be rejected when submitted for journal publication. They might instead prescreen research projects, bringing to fruition only those that yield significant results. They might also be tempted to engage in specification searches until they derive results that are (ostensibly, ignoring the pre-testing that may have occurred) statistically significant.

It is too cynical to suppose that researchers slant their results simply to get them published. Another reason for which researchers might find and report strong positive results is that the researchers firmly believe that such results are correct. The process of specification search can be given a Bayesian¹⁹ interpretation. Suppose that I believe that the services performed by a particular forest ecosystem are of considerable value. Suppose also that I amass a set of data, choose what seems to be a plausible functional form, and perform a statistical analysis to test the hypothesis that “the value of ecosystem services provided by this area of forest are no greater than the opportunity cost of its preservation”. Finally, suppose that I cannot reject the hypothesis. I can react to this information in one or a combination of two ways. First, I can say “That’s surprising! I must have been wrong,” and change my mind. Or, I can say “I’m pretty sure I’m right about the value of ecosystem services; since I didn’t get the ‘right’ result, some other aspect of my analysis must be wrong. I’m going to try a different specification and see if I get the ‘right’ result with it.”²⁰

The reader of such work is then faced with her own problem of inference, informed by her own beliefs. If confronted with a claim that ecosystem services are of significantly greater value than the opportunity cost of their preservation, but dogged by the suspicion that such a claim is based on extensive specification searching, does she believe the claim or discount it? It seems that the general answer will be that she will follow her own prior beliefs on the matter.

Such a scenario could lead to what I suggest might be “a battle of Bayesians”. A researcher believing that ecosystem services are important and valuable would credit those formulations of the empirical model that led to the confirmation of his prior beliefs. A skeptical reader could dismiss the researcher’s findings on the presumption that the researcher had continued to test specifications until he found the one that comported best with his prior beliefs.

¹⁷ Publication bias is also perceived as a problem in the pharmaceutical research industry, where some allege that results favorable to a research funder’s interests tend to be published, while those that are not are “stuck in the file drawer”.

¹⁸ A result is said to be “statistically significant at the x% level” if the probability that it would have been found to be as large as it is if it did not, in fact, have any consistent effect, is less than x%.

¹⁹ The Reverend Thomas Bayes (c. 1702 – 1761) was a British mathematician best known for formulating Bayes’ Law, a theorem describing how probabilities are updated in the presence of new information. Bayesian analysis concerns the ways in which “prior distributions” are updated on the basis of new information so as to form “posterior distributions”.

²⁰ I do not believe that this is unusual. I have certainly proceeded in this fashion myself at times. It is not unusual to overhear one colleague ask another “Did that new specification you tried work?” where I presume “to *work*” would be “to produce statistically significant results of the sign and magnitude expected”.

Which is right? In a sense – and despite the contradiction between their positions – both are. Each is equally entitled to her prior beliefs concerning ecosystem services, and there is nothing necessarily irrational or inconsistent in each updating her beliefs as she has. Such divergent positions cannot be reconciled absent agreement on an “experiment” – a set of data on which a hypothesis can be tested – that each agrees will resolve the matter.²¹

An extreme example of a Bayesian standoff: Stated preference studies

Nowhere is the problem of defining agreed-upon procedures for the estimation of value more vexing than with stated-preference studies. Economists often distinguish their discipline from other social science approaches by noting that they base their analyses on what people actually *do* rather than on the subjective thought processes that people *say* motivate their actions. It is, then, not surprising that sharp disputes have broken out between economists citing the above distinction and those who abandon it by *asking* people what they *would* do rather than *observing* what people *have* done.²²

Adherents of the latter procedure conduct “stated preference” (as contrasted to “revealed preference” studies). Examples include “contingent valuation studies”, in which survey respondents are asked how much they would pay for a particular good or service,²³ and “conjoint analysis” and “choice experiments,” in which respondents are asked to choose among different combinations of public and private goods to determine their preference for the former and the consistency of their preferences between them.

Stated preference methods have often been used in the valuation of ecosystem services. Richardson and Loomis (2009; see also Loomis and White 1996) survey a number of stated preference studies that have been conducted on the value of threatened and endangered species. Naidoo and Adamowitz (2005) consider the value of birds to protected areas in Uganda. Many such studies exhibit a troubling feature, however. As Brown and Shogren (1998) commented in reviewing 18 earlier stated preference studies, “the average person was willing to pay about \$1000 to protect 18 different species Many will find these figures suspiciously high.” They are “suspiciously high” because there are so many public goods to whose maintenance a survey respondent might be asked to contribute. More generally, critics have long alleged that respondents to stated preference surveys regard them as opportunities to “purchase moral satisfaction” (Kahneman and Knetsch 1991) – albeit without having to actually pay for it – rather than as considered statements of budget-constrained choices. Peter Diamond (1996) has suggested that relatively simple “adding-up” tests can be conducted to determine the consistency of stated-preference studies, although other authors suggest that Diamond’s conditions are more stringent than need be implied by received theory (Smith and Osborne 1996).

This is precisely the problem, however: different commentators disagree as to what received theory requires, and hence, as to whether stated preference studies provide valid estimates of the social values of things like imperiled ecosystem services. While V. Kerry Smith was writing over a decade ago, it is likely that his assessment remains accurate:

Indeed, there is a curious dichotomy in the research using CV for nonmarket valuation. Environmental economists actively engaged in nonmarket valuation continue to pursue very technical implementation or estimation issues, while the economics profession as a whole seems to regard the

²¹ This standard echoes Karl Popper’s (1959) notion distinguishing science as a realm of inquiry whose propositions must be falsifiable if they are to be meaningful. If we are to credit statements such as “the services provided by a natural ecosystem are more valuable than the opportunity cost of its maintenance,” or the opposite, we must have some agreed-upon set of techniques for measuring values and opportunity costs. My concern in much of the empirical literature on ecosystem services is that one side or the other in the debate can too easily say “you have reported the wrong ‘experiment’”.

²² See, for example, Nordhaus and Kokkelenberg (1999), who wrote: Some techniques such as hedonic-price or travel-cost studies rely on behavioral or market-based estimates; while these estimates are subject to significant measurement problems, they are conceptually appropriate in economic accounts. Other techniques, such as contingent valuation, are not based on actual behavior, are highly controversial, and are subject to potential measurement errors. [emphases added]

²³ I simplify greatly in the interests of brevity. Open-ended questions of the form “How much would you pay for x?” are no longer asked by most state-of-the-art practitioners of stated preference studies. A more likely question would be a variant of “Would you pay \$y for x?” possibly followed by “If you would pay \$y, would you also pay \$y + z for x?” The difference between the answers given to “willingness to pay” and “willingness to accept” questions has also generated great controversy in the literature.

method as seriously flawed when compared with indirect methods. They would no doubt regard this further technical research as foolish in light of what they judge to be serious problems with the method. (Smith, 1997; p. 42)

The profession has, essentially, agreed to disagree as to the validity of the approach.²⁴ Obviously, this state of affairs makes any estimates of ecosystem values based on stated preference approaches problematic.

Benefit transfer

Another approach has been employed in a number of ecosystem service valuation studies. Probably the most prominent of these was the study published in *Nature* in 1997 by Robert Costanza and his colleagues. In that piece the authors combined data from scores of earlier studies reporting the value of services generated by particular ecosystems. They then extrapolated these service values on a per-hectare basis to other ecosystems of similar types around the world. As has already been noted, this procedure is problematic. If, as may often be the case, ecosystems that are unusually valuable because of their proximity to centers of population or sources of pollution are the subjects of study, extrapolation of estimated values to more remote areas will be inaccurate.

The general technique of applying estimates derived in one setting to the valuation of services generated in another is known as "benefit transfer," and has been the subject of considerable research (see, e.g., the survey in Navrud and Ready 2007). Best practice now gravitates to "function" rather than "estimate" transfer, wherein researchers transfer a set of parameters describing monetary benefits as a function of underlying site attributes rather than simply a numerical measure of monetary benefits themselves. Other approaches attempt to "calibrate preferences" by imposing consistency across different methods of estimation and categories of value (see, e.g., Smith, *et al.* 2006).

One troubling aspect of existing benefit transfer studies – and related "meta-analyses"²⁵ – however, is that they often find that the *methods* employed in individual studies may explain the *results* they derive (see, e.g., Richardson and Loomis 2009). At the very least this suggests that some approaches may be more reliable than others. More generally, any approach that aggregates other studies can be no more reliable than the studies on which it is based.

One might counter the above observation by saying that aggregation of individual studies should, by the law of large numbers, smooth the errors found in each. This would be true if the underlying studies themselves were unbiased. If, however, the potential sources of bias discussed above are matters of concern, the biases of the individual studies would be transferred to the study that aggregates them.

4. How does the ecosystem services framework change conservation policy?

Its more enthusiastic advocates have hailed the adoption of an ecosystem services framework as a watershed event in the evolution of conservation policy. Is it likely to prove to be one?

The underlying logic of the ecosystem services framework is impeccable. Natural ecosystems do, of course, provide a host of valuable services to society. There is no denying that the water purification, pollinator habitat, recreational opportunities, erosion protection, soil regeneration, etc. provided to humanity are valuable. Of course, there are

²⁴ "We agree to disagree" is the phrase employed by a presenter at a recent workshop at which I discussed his and other papers using stated preference methods. Much the same message was conveyed by a prominent representative of the other side of the stated preference debate, Peter Diamond, in a personal communication in 2005.

²⁵ A meta-analysis is a "study of studies". It typically takes the estimates of earlier studies as a set of "dependent variables" which it then seeks to explain as functions of the location, methods, or other attributes of the studies.

also questions as to how such values compare to alternative uses, how valuable they are on the margin, and what disamenities natural ecosystems also generate. Some contributors to the literature on ecosystem services note, for example, that there may be rapidly diminishing returns in certain services with respect to the size of the ecosystem providing them (see, e.g., Tallis, 2005),²⁶ and that completely *unnatural* systems might provide equal or superior services in some respects (see, e.g., Turpie, *et al.*, 2003)²⁷ With respect to disamenities, it is instructive to note that many of Europe's wetlands were drained in 18th and 19th century efforts to control malaria (Bate 2002; McCauley 2006).

It should also be remembered that many of the values ascribed to ecosystem services arise precisely because of the benefits they provide to decidedly *unnatural* systems: large areas of monoculture crops benefit from pollination, vulnerable urban structures benefit from flood control, etc. If the benefits of maintaining areas in natural ecosystems depend on the size and location of unnatural areas, there is obviously a balance to be struck between preservation and development. This observation would not come as any revelation to advocates of an ecosystem services approach, but it does underscore the importance of evaluating marginal tradeoffs.

It may be problematic if the general conclusion drawn from an analysis of conservation vs. development tradeoffs is that they are currently drawn with too much emphasis on development. There is an important result in economics known as the "general theory of the 2nd best" (Lipsey and Lancaster 1956). Put in very colloquial terms, it says that "two wrongs may make a right" in some circumstances, or, perhaps put more accurately, "correcting" one problem may not improve welfare when other problems are not addressed. While the situation is certainly very complicated, it is not clear, for example, that taking steps to provide greater ecosystem services for one community would always and necessarily be the best measures to combat other environmental problems, such as global warming. Kareiva and Ruffo (2009), for example, note that "Massive infrastructure projects, monocultures of plantation trees, seawalls and levees, [and] biofuels that ultimately accelerate land conversion" are among the strategies suggested for combating climate change. While noting again that the science is complicated and appears to remain in some dispute, one cannot simply assume that the maintenance of natural ecosystems is necessarily consonant with other ecological or social objectives.

These observations may be particularly trenchant as it would appear that concern for ecosystem services arises largely from conservation advocates' concern for the preservation of biodiversity. There is some discussion in the ecosystem service literature concerning the degree to which biodiversity preservation and ecosystem service provision are, in fact, consistent objectives (see, e.g. Balvanera, *et al.*, 2001; Tallis and Kareiva 2005; Turner, *et al.*, 2007; Nelson 2009).

These concerns are likely to be particularly acute when considering particular strategies for realizing the purported value of ecosystem services. In the absence of clear evidence of the demand for ecosystem services, conservation (and, in some instances, development) donors have often underwritten "integrated conservation and development projects" (ICDPs) in which local communities are subsidized to undertake projects such as the collection of nontimber forest products, "ecotourism", and pharmaceutical research and development based on natural products. Such ICDPs have proved to be very controversial, with both their ecological (see, e.g., Terborgh 1999; Terborgh, *et al.* 2002) and economic (see, e.g. Wells and Brandon 1991; Ferraro 2000; Ferraro and Simpson 2002) benefits in considerable dispute. Many commentators are likely to be extremely disappointed if the current interest in ecosystem services turns out to be only a recycled argument for a conservation strategy that has already been tried and, in the opinion of many, failed dismally, in developing countries.

Yet some authors who have extolled the promise of the ecosystem service framework write hopefully of the possibility of "ecosystems services . . . aligning conservation values and poverty alleviation" (Tallis and Kareiva

²⁶ The author notes that increases in forest cover above 70% of a land area have no demonstrable effect on water purification. See also Kareiva and Ruffo 2009.

²⁷ The authors note that South African mountains might provide better water supply to the cities and farmers below them "if the mountains were covered in concrete" than if they native vegetation were restored.

2005), largely in hopes of tapping into international development funding.²⁸ To those who have followed the evolution of conservation strategy over the past two decades, such suggestions elicit a disheartening sense of *déjà vu*. While the new emphasis on ecosystem services – and the operational mechanisms of payments for their provision – differ in some important ways from the ICDP strategies that have often prevailed in recent decades, they are also similar in one fundamental way. To the extent that the ecosystem services emphasized in recent conservation literature call for the preservation of natural ecosystems in order to provide *local* public goods – services such as water purification, erosion control, pollinator habitat, etc. – they presume that local communities have proved incapable of recognizing and safeguarding these services on their own.

It is almost tautological that poor communities are poor because they fail in some respects to take the actions that would be required to improve their collective welfare. Having stipulated as to this fact, however, it seems disingenuous to suppose that poor communities will be better off if they conserve ecosystems whose preservation would also benefit an international conservation community whose underlying interest is in the preservation of biological diversity, not the welfare of local communities *per se*.

How then, should the ecosystem service framework affect conservation policy? It seems reasonable to suppose that research should continue into the valuation of ecosystem services. If clear and compelling evidence can be assembled to suggest that the local value of preserved ecosystems will compensate the opportunity costs of their preservation, perhaps local development and international biodiversity interests can be aligned.

For the reasons cited above, however, research has not yet established that case. Given the circumstances, we might consider two paradigms in deciding on policy. The first is that provided by the theory of options. When faced with uncertainty and irreversible consequences, it is prudent to exercise an abundance of caution. The second, competing, view is provided by Ockham's razor. While there may be other possibilities, the most parsimonious explanation for why local people do not preserve their ecosystems is that they do not believe it is in their interest to do so.

While some poor communities may be sufficiently forward-looking as to wait for the resolution of uncertainty before making irrevocable choices concerning their ecological assets, it would be surprising if this were the norm. So, whether or not one thinks local communities are making wise choices in liquidating their natural capital,²⁹ they are likely to continue to do so in the absence of international payments for conservation. In the final analysis, then, it seems that the consideration of ecosystem services does not leave conservation planners in a much different position than they were before with respect to practical conservation strategy. Payments will likely be required from the wealthier to the poorer nations of the world on the basis of carbon sequestration in standing forests or intangible values of biodiversity if natural ecosystems are to be preserved. We may choose to call the goods for which such payments are made "ecosystem services", but the designation makes little difference with regard to the practical need to structure international payments for conservation.

²⁸ The passage quoted here follows immediately after that quoted on p. 2 above, in which Tallis and Kareiva suggest that the "realization of the market worth of ecosystem services has the potential to increase conservation funding by orders of magnitude."

²⁹ It is worth remembering in this context that some authors who are sympathetic with the ecosystem services perspective believe such choices are, in fact, wise, in the absence of international payments; see Kremen et al. 2000, Naidoo and Ricketts 2006.

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