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The Ecosystem Services Framework and Natural Capital Conservation

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Abstract Work at the interface of ecology and economics has inspired a major transformation in the way people think about the environment. Increasingly, ecosystems are seen as capital assets, with the potential to generate a stream of vital life-support services meriting careful evaluation and investment. We first present the concepts underpinning the ecosystem services framework (ESF), laying out the scope and limitations of the approach. We then describe the major challenges in making the ESF operational: (i) detailed information at scales relevant to decision-making; (ii) practical know-how in the process of institutional design & implementation; and (iii) compelling models of success in which economic incentives are aligned with conservation. We close with a brief review of pioneering experiments now underway worldwide, which illustrate how these challenges can be overcome.

Keywords Ecosystem services · Natural capital · Cast-Benefit analysis

1 Introduction

Although the term and concept of ecosystem services has received a great deal of attention in the recent academic literature, an operational decision-support system for better biodiversity conservation and environmental change management has been slow to emerge. The ecosystem services framework (ESF) highlights the long-term role that healthy ecosystems play in the sustainable provision of human wellbeing, economic development and poverty alleviation across the globe. Efficient and effective management of ecosystems (living natural capital) can sustain the provision of vital ecosystem services such as climate stabilisation, drinking water supply, flood alleviation, crop pollination, recreation opportunities and amenity and

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cultural assets (Westman 1977; Holdren and Ehrlich 1974; Daily 1997; Balmford et al. 2002; Turner et al. 2003; MEA 2005).

But the human welfare benefits generated by ecosystem goods and services are both private and public goods made available across a range of temporal and spatial scales, and associated with (or hindered by) a variety of property rights and other institutional arrangements. The resource space can be privately owned, publicly owned by the nation, represent common property or be subject to international treaties and agreements. The gainers and losers in any environmental change situation therefore vary depending on the type and scale of ecosystem service provided, the mix of stakeholders involved and the socio-economic characteristics and the socio-cultural context. This complexity ensures that the political economy of ecosystem conservation will encompass not just efficiency and effectiveness criteria, but also equity, justice and legitimacy criteria together with other ethical concerns (Adger et al. 2001; Paavola 2005).

2 The Problem

In the past, nature conservation and protected area policy was justified largely by a combination of separate scientific and ethical "intrinsic value" arguments. But today, with unprecedented and intensifying pressures to deplete natural resources, the traditional arguments in support of ecosystem conservation alone are not sufficient. They do not capture the utter dependence of human well-being on natural capital. Despite growing global-level recognition (via the United Nations' Millennium Development Goals and the Millennium Ecosystem Assessment) that conservation often makes economic sense for society as a whole, decision-makers, from individuals to governments, continue to discount inappropriately when choosing between ecosystem conversion or conservation and seem unwilling or unable to provide sufficient finance/investment to match the conservation rhetoric (Pearce 2007). As a result, wild habitats and populations are declining by an average of 0.5–1% per annum, with losses particularly pronounced in the developing world (Balmford et al. 2003).

Adoption of the complete ecosystem services-based decision support process (see Fig. 1) can help to reverse this trend by analyzing and synthesizing relevant knowledge and capturing the benefits of ecosystem services. The framework encompasses a comprehensive analytical and practical process which begins with a problem/issue identification stage in which ecosystem service provision and the social, economic and politico-cultural contexts are delineated and scaled. The chosen ecosystem and services are then modelled, mapped and valued. The management choices and their opportunity costs can be explored via scenarios of future states of the world and/or policy interventions. The public goods characteristics of some ecosystem services in particular highlight the need for practical and inclusionary payment mechanisms to ensure that sustainable management of resources is a reality. Finally, capacity building, monitoring and re-appraisal efforts should all be adequately invested in to complete the circle.

A number of key constraints, however, need to be overcome:

- Despite growing general awareness of conserved ecosystem benefits, detailed information at scales useful for decision makers on how people benefit from specific services remains deficient. This "information failure" is one reason why conservation investment finance is still too low and sometimes ineffective;
- Another reason is "institutional failure". The beneficiaries of ecosystem service provision are often different and distant from those who gain from ecosystem transformation. Local socio-ecological contexts, including property rights and institutions, are often not

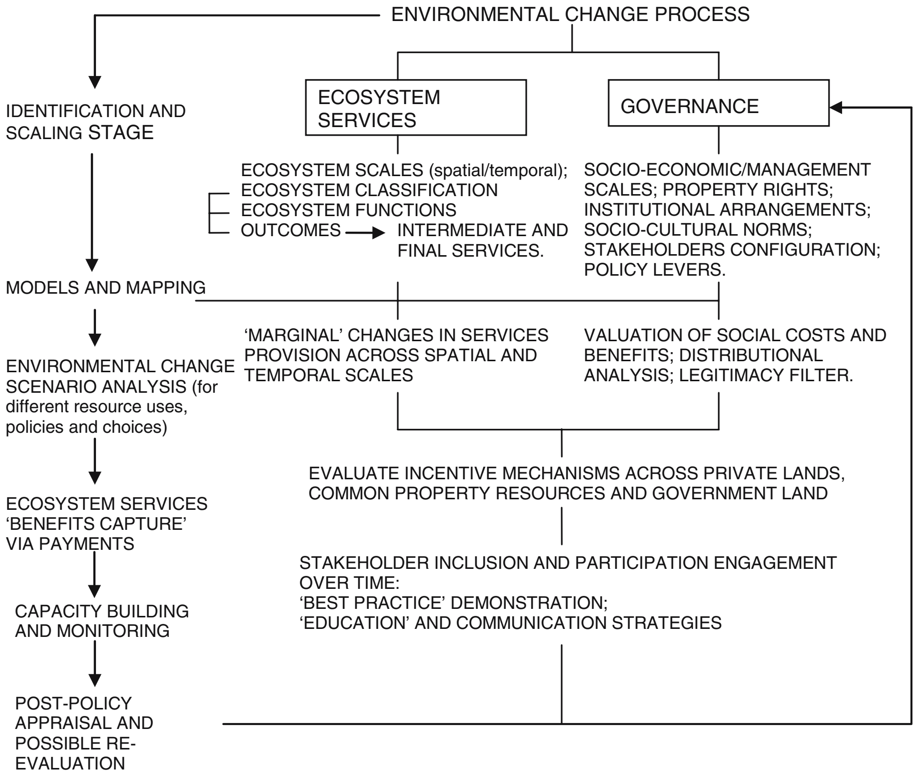

Fig. 1 The Ecosystem Services Framework (ESF)

given sufficient consideration in conservation programmes, so that legitimacy and equity concerns inhibit uptake;

- Thirdly, "market failure" occurs because of the public good characteristics of many benefits and their lack of prices. Markets also typically reward short-term values of natural resources (exaggerating the real opportunity costs of conservation) to the detriment of long-term ecological health and human welfare.

A common and agreed set of definitions and principles for the ESF are needed if the procedures are to be made fully operational. The next section presents our contribution to this debate.

3 ESF Terminology and Systems Principles

We take the case of wetlands in order to provide some detail (Turner et al. 2000). Figure 2 shows the linkages between ecosystem structures and process functioning and consequent outcomes which lead directly or indirectly to valued human welfare benefits (gains or losses). So ecosystem services are the aspects of ecosystems consumed and utilised to yield human well-being.

Ecosystem structure is a service to the extent that it provides the foundation from which ecosystem processes occur. How much structure and process is required to provide a diversity of services in a given context is still an active research question. Some minimum configuration

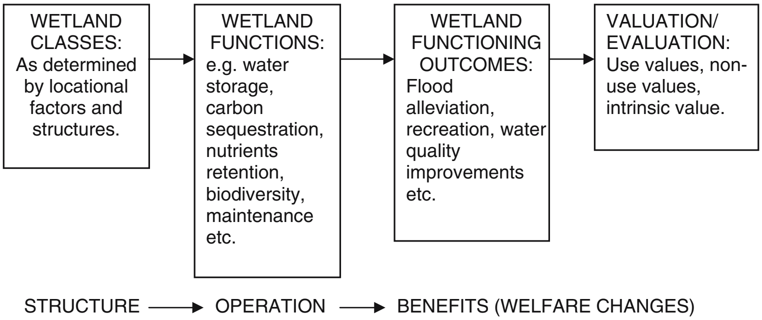

Fig. 2 Ecosystem Services Approach: Wetlands

of structure and process is clearly required for "healthy" functioning and services provision, but the minimum is often uncertain. This "infrastructure" has value in the sense that its prior existence and maintenance is necessary for services provision (Gren et al. 1994; Turner et al. 2003). While "infrastructure" value in nature is a condition for the existence of other values, it does not support an argument for rejecting economic values for ecosystem goods/services as such, but rather it constrains those values to reflect precaution and a stewardship ethic which guarantees endowments/bequests of natural capital to future generations (Crowards 1998).

Some ecosystem processes produce "joint product" outcomes, for example nutrient cycling in a wetland can result in cleaner water. Nutrient cycling is therefore a service indirectly utilised by humans, while provision of clean water is a direct service and a benefit. Recreational activities, such as bird-watching, provided or enhanced by the existence of a wetland and related features are a benefit. Because ecosystems are "systems" with feedbacks, time lags, nested phenomena and other complex dynamics, the value of their services is often consumer-dependent (Boyd and Banzhaf 2007).

Different stakeholders can perceive different benefits (sometimes complementary but also competitive) from the same ecosystem process outcomes. For instance, while rainforest conservation in Madagascar is estimated to yield net benefits to both global and local stakeholders (principally through carbon sequestration and non-timber forest products, respectively), conservation comes at a net cost at the national scale (resulting from lost revenues from industrial logging) (Kremen et al. 2000). Stakeholder perceptions, property rights and institutional arrangements are thus important components of any scheme to capture benefits on a practical and lasting basis. Failure to recognise and accommodate these components invites a lack of trust, accountability and legitimacy.

In line with earlier work (Daily 1997; Costanza et al. 1997), the framework set out in this paper recognises maintenance of biodiversity[1] as both an intermediate and final ecosystem service since biodiversity is a major component of ecosystem structure, processes and services and benefits outcomes. The intermediate service role of biodiversity is set out in the organisation and operation of ecosystems, while the final service role is linked to cultural, spiritual, option and bequest values that are significant human benefits.

[1] Biodiversity is defined as "the variability among living organisms from all sources, including *interalia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (Convention on Biological Diversity, Article 2).

Many goods and services provided by biodiversity have characteristics of public goods. This usually means that individuals acting in their own self-interest will under-provide for its conservation and incentives need to be offered. Property rights owners can be induced to cooperate through flexible incentives that reward stewardship motivations towards species and habitats. In North America, Europe and Australia, a range of compensation schemes have been debated and/or implemented: direct compensation to land owners, conservation banking, tradable habitat rights, insurance schemes and tax relief mechanisms (Shogren 2005). These payment approaches require a willing buyer-seller market arrangement and well-specified property right.

In parts of the developing world, direct and indirect payment schemes have had less certain outcomes because of inequitable or poorly specified land tenure, legal system rigidities or gaps and relatively high opportunity costs of conservation (real and perceived) (Ferraro and Kiss 2002). The international community has provided a number of "payments for ecosystem services" opportunities such as debt-for-Nature swaps, the "incremental cost" model operated by the United Nation's Global Environment Facility and bilateral aid tied to biodiversity conservation. The sums of money involved have been estimated by several analysts (see James et al. 2001; Pearce 2007) and run into billions of dollars.

Despite this effort, biodiversity loss continues and some commentators have questioned the effectiveness of this international effort and/or the real level of concern about ecosystem loss (Hutton and Dickson 2000; Barrett et al. 2003; Pearce 2007). Some have gone as far as to suggest there is a "global deficit of care", for example to mitigate global warming (Pearce 2003). Scaling mismatch is another problem in this context. Species and many habitats are local, which makes the political economy of species and habitat conservation local (Shogren 2005), and political endemism a factor as important as biological endemism (Ceballos and Ehrlich 2002; Ceballos et al. 2005). National to international arrangements seeking to protect and conserve global natural capital mainly through market-based economic incentive mechanisms need to incorporate local social, political, legal and cultural complexities into their design and practice.

Economic incentives on their own are unlikely to transform local cultural, ethical and behavioural traits towards environmental stewardship and citizenship. In the future, the legitimacy and fairness perceptions of such schemes will become increasingly important (Adger et al. 2001; Paavola 2005) and packages of complementary measures will need to be deployed. So what is the current position and what are the future prospects for biodiversity and other ecosystem services conservation?

4 Costs and Benefits of Ecosystem Conservation and the "Duty of Care" Deficit

A growing literature has emerged in which ecologists and economists have undertaken valuation exercises to frame and quantify the social costs and benefits of conserving biodiversity and ecosystems services (e.g. Daily et al. 2000; Balmford et al. 2002; Turner et al. 2003; Naidoo and Adamowicz 2005; Chan et al. 2006; Goldstein et al. 2006). The common result that these studies generate is that, across a wide range of socio-ecological contexts, conserved ecosystems generate net benefits (i.e. benefits of conservation outweigh the costs of conservation management). These studies require estimates of willingness-to-pay for conservation benefits and utilise a number of economic valuation techniques, including survey-based contingent valuation and choice experiment methods.

The role of rainforest in coffee production is an illustrative example, making two striking points: first, how valuable services can be hidden right under our nose (in a steaming cup of coffee); and, second, how conservation can be justified economically even in the midst of prime farmland (where conservation is typically assumed to have little merit and to face prohibitive opportunity costs). Coffee is one of the world's most valuable export commodities, employing over 25 million people (O'Brien and Kinnaird 2003), and virtually anywhere it grows formerly supported rainforest. In a study on a Costa Rican farm, researchers determined that the proximity of two small remnants of rainforest (46 and 111 ha) to the farm (<1 km away) increased yields by 20% and translated into ca. US$60,000 per year, via the supply of bee pollinators dependent on the rainforest (Ricketts et al. 2004). This value is commensurate with competing land uses.

Yet, in a deliberately provocative paper, it has recently been argued that these valuation studies are "optimistic" and may be "biased" in the sense that only studies which yield positive results in terms of environmental protection tend to get published (Pearce 2007). It was further argued that the surveyed willingness-to-pay values (WTP) for ecosystem services benefits are much higher than the sums of money actually spent on biodiversity and other preservation projects and programmes (Pearce 2007). The question is raised that maybe, in terms of real action rather than rhetoric, the global community does not care that much about biodiversity loss and related issues. The pessimistic perspective is strengthened further by the argument that the "easiest" ecosystem conservation options have already been taken and that in the future, contexts in which the economic opportunity costs of conservation will be higher will be the norm.

The pessimistic position can be challenged. While it is the case that WTP estimates have been criticised, within and outside economic circles, much progress has been made on survey designs and understanding of the motivational issues involved (Sugden 2005). Dealing with so-called existence values related to biodiversity along with other "non-use" value estimation problems, remains problematic, but in many cases WTP estimates would have to be "wrong" by a very large margin for the cost-benefit calculation not to come out in efficiency terms in favour of conservation. The publication bias argument is in our view less strong. Rational decision making should be anchored to the available evidence base and if the grey literature contains studies which show ecosystem conservation in a negative economic efficiency light (a priori or *expost*) we are not aware of this. But we would argue in any case that equity, legitimacy and other ethical concerns are important decision-making criteria in this context and may temper the economic efficiency result.

For us, the future prospects turn more on the extent to which the complete ESF can be implemented and, in particular, whether payments for ecosystem services measures can be made more sophisticated and better tailored to local socio-ecological circumstances. While all the "easier" conservation-versus-conversion contexts may not yet have been faced and decided on, the economic opportunity costs in many future choice contexts (judged over the short to medium term) are likely to be moving against conservation options. This makes it imperative that the ESF is fully adopted and that a sufficiently long-run decision horizon is accepted.

In the final section of this paper we review some case study evidence to highlight the issues discussed earlier and to shed some light on the future prospects for ecosystem services conservation and management.

5 Case Studies

Over the past decade, an astonishing number and diversity of efforts to implement the ESF have emerged worldwide. Individually, most of these efforts are small and idiosyncratic.

Yet, collectively, they represent a powerful shift in the focus of conservation organisations toward a more inclusive, integrated and effective set of strategies (Daily and Ellison 2002). Recent reviews of ES efforts by the world's two largest conservation NGOs, The Nature Conservancy and World Wildlife Fund, revealed dozens of major projects implementing the ESF in some way (Yuan-Farrell and Kareiva 2006a, b). Taken together, these efforts span the globe and a target a full suite of ecosystem services, including principally forest-generated services of carbon sequestration, water supply, flood control, biodiversity conservation and enhancement of scenic beauty (and associated recreation/tourism values).

Many ES efforts focus on a single service that stands out as sufficiently important, from economic and political perspectives, to overcome the activation energy required to protect it. Under the institutional umbrella created for the focal service, it is possible that other services may be at least partially protected (Balvanera et al. 2001). Recent, pioneering examples include water purification by New York City, and flood control by Napa, California. To illustrate the range of efforts underway, in contrasting biophysical, economic and institutional environments, we'll first examine these single-service approaches. We will then review a larger-scale investment in natural capital for water flow regulation in China. Finally, we'll present a pioneering case that points to the future, in which a suite of ecosystem services is targeted simultaneously, in Costa Rica.

In 1997, a water pollution crisis led New York City to a bold experiment. Instead of relying on technology, in the form of a new water filtration plant, the city invested in natural capital. The decision saved several billion dollars—and set a global precedent. This investment is restoring the natural purification services of the Catskills–Delaware watershed, the heart of the water purification and delivery system supporting some 10 million water consumers (Chichilnisky and Heal 1998).

Less appreciated is the machinery of the watershed, pumping out as much as 6.8 billion litres of purified water daily and supplying 90% of the needs of New York City's residents. For decades, these urbanites have relied on the 5,000-square-kilometer watershed to purify their drinking water, a product of exceptional quality. The forest provides this valuable service for free, cleansing the water as it sifts through roots and soil. The forest also metes out water gradually, stabilizing drinking supply and mitigating flooding; prevents soil erosion; shelters wildlife; stores carbon, helping to stabilize global climate; and graces the region with stunning beauty.

In the late 1980s, the United States Congress and Environmental Protection Agency had started to react to perceived threats to surface-water supplies—rivers, lakes and reservoirs. Two-thirds of the U.S. population depend on such systems, rather than groundwater. Yet the great majority of them have been degraded, by years of urban sprawl and runoff from second homes, farms and golf courses. In 1991, the EPA ordered New York City to build a water filtration plant—unless the city could prove it could maintain water quality without it. Presented with the budget-breaking costs of that investment—up to US$8 billion, by some estimates, plus at least US$300 million in annual operating costs—city officials took a revolutionary approach, investing instead in restoring the natural asset, the watershed.

Since 1997, the city has invested nearly US$2 billion in land management changes and innovative tactics such as purchasing land around reservoirs to preserve forests and wetlands that buffer against pollution, paying landowners to restore forest along streams, and offering technical aid and infrastructure to farmers and foresters. This summary does not capture either of the on-going experiments—relating to whether these natural capital approaches will "work" (relative to, or in addition to ever-changing technological alternatives; NRC 2000) and to the complex political negotiations inherent in land-use decisions. The bottom line is that, to many, the outcome is a triple-win: urban people getting pure water at lower cost;

rural people being rewarded for good land stewardship; and visitors and rural residents alike enjoying the spectacular landscape, saved from out-of-control urban development (Daily and Ellison 2002). Cities worldwide are attempting to implement similar approaches (e.g. UNFAO 2004).

Over the past decade, pioneering ecosystem approaches to flood control have been emerging as well. Throughout the world, floods are by far the most common "natural disaster", invited by settlements that, throughout human history, sprang up in fertile flood plains. In the U.S. alone, floods cost dozens of lives and US$4 billion in damages in the average year. Worldwide, flooding caused ca. 40,000 deaths and US$29 billion in economic losses in 1999–2000, as estimated by Munich Re (Daily and Ellison 2002).

The revolutionary approach of Napa, California, is now touted as a model of success, though the revolution came only after enduring 28 major floods and well over $500 million in damages, since record-keeping began in 1862 (Brauman 2006; Daily and Ellison 2002). By the late 1990s, some residents proposed a new plan—a "living river" approach to flooding. Instead of investing in physical capital—reinforcing the levees and concrete barriers that had served to control the river's surges in the past—they proposed using the ESF to guide investments in a "living river" approach. This meant moving nine bridges and over 100 buildings and restoring 250 ha of floodplain, instead of a deep, straight, concrete channel, yearly dredging and tall floodwalls. Bridge replacement removed obstacles to high flows, bank terracing reconnected the river to its historic flood plain, and easements and acquisitions removed especially vulnerable structures from harm's way (Brauman 2006).

Interestingly, residents approved of the ecosystem approach even though it was projected to cost more (US$200 million) than the physical capital approach (US$150 million)—and they had to pay for part of it. This choice was made in anticipation of the many benefits that would result from an investment in natural capital under the umbrella of flood control, that were not valued explicitly. These benefits include the restoration of fish, wildlife and scenic beauty, and all the recreation, tourism, and related commerce that residents hoped would follow.

Indeed, as reported in popular magazines, the town was revitalized by this investment, with boating, hiking, fine dining and other amenities unimagined when the town battled the river as an enemy (Boone 2005; Cusumano 2004). The City of Napa's Economic Development Office confirms that a major increase in private investment occurred after the flood plan was approved—amounting to $193 million in private construction from 1999 to 2005 (Brauman 2006).

In total, Napa's plan will mitigate flooding over six of the 55 miles of the Napa River (and one mile of Napa Creek, a tributary in town). Having proven success at the local scale, the success of Napa's efforts now hinge critically on whether upstream management of the river improves (Jeffrey Mount, personal communication 15 February 2007). This still-fragile situation highlights the dependence of local efforts on support at larger scales.

China illustrates work at large scales perhaps better than any other part of the world. Prompted by massive flooding in 1998, at a cost of US$20 billion in damages, the Chinese government enacted a sweeping land-use policy entitled the National Forest Conservation Program (NFCP). The policy is intended to regulate water flow and promote soil retention, primarily by conserving and restoring natural forests while increasing timber production in plantations. A key component of the policy is a logging ban on 30 million ha of natural forests in the upper reaches of the Yangtze River and upper and middle reaches of the Yellow River (SFB 2005). The government has invested US$billions into the programme since its inception, through a wide array of public policy instruments, including training, resettlement, and direct compensation of forest dwellers, and mandatory conversion of marginal farmlands to forest lands (Zhang et al. 2000).

An incentive for water flow regulation, in addition to flood control, is increasing hydropower production efficiency. Forests in the Yangtze River watersheds decrease flow in the wet season and enhance it in the dry season. As a result, researchers estimate that a single hydroelectric power plant (in Gezhouba) increases its annual electricity production by up to 40 million kWh per year, at a net benefit of ca. US$610,000 each year. This value may increase five-fold when the Three Gorges hydroelectric power plant is operational. In total, there are actually about eight major, interlinked policies in China controlling land use and forest cover, with the aim of enhancing the supply of these vital watershed services (SFB 2005).

Coastal zone policy in the UK and Europe is being re-orientated towards a more flexible and adaptable approach, while water catchment management is also being reformulated under the EU's Water Framework Directive. This switch in strategy provides an excellent opportunity to implement the ESF. In the UK coastal policy will encompass the new managed realignment (i.e. setting back of sea flooding and erosion defences) measure. Because managed realignment policy needs to be appraised across a more extensive spatial and temporal scale than has been the case in the traditional scheme-by-scheme coastal management system, the ESF neatly fits the bill. Whole estuaries or multiple coastal cell areas need to be treated as a "single" project covering a number of realignment sites and multiple ecosystem services. A study of the Humber estuary on the east coast of England has shown that if, through realignment, 7494ha of new intertidal area was created, the nutrient sink/storage ecosystem services would increase by 150% for C, 83% for N and 50% for reactive P (Andrews et al. 2006)[2] If the managed realignment option is compared to the traditional "hold the line" coastal defence policy and the relevant economic costs and benefits are appraised over a 50 year to 100 year time horizon, there is a strong economic efficiency case in its favour. The cost-benefit analysis includes ecosystem service benefits (carbon storage and enhanced environmental amenity) and savings in defence maintenance costs within the realignment appraisal (Turner et al. 2007).

For the provision of ecosystem services to be efficient, it is important that investments in particular land covers/uses be strategically oriented around the full suite of desired services and coordinated across landscapes. This is because the spatial configuration of ecosystems greatly influences the production of services (Goldman et al. forthcoming). Under existing incentive programs, this level of coordination is typically neither required nor encouraged. But in Costa Rica, the government does target a broad suite of important services, and the programme is becoming more efficient with respect to landscape coordination as well.

In 1997, Costa Rica launched a nationwide scheme of payments for the provision of ecosystem services, known as Pago por Servicios Ambientales (PSA). The PSA targets carbon sequestration, water quality and quantity (for drinking and irrigation supply, and hydropower), biodiversity conservation and scenic beauty (for ecotourism). Funds from a diversity of sources (private sector, World Bank, a gasoline tax in Costa Rica) are pooled and distributed to voluntary participants at terms of ca. US$50 per ha per year (Pagiola 2002). This programme is seen as a model internationally, and is now being replicated in Mexico.

[2] The history of reclamation and its biochemistry in the Humber is common to many estuaries in northern Europe.

6 Conclusions

The ESF and practical experiments described here highlight the clear and powerful ethic—and increasingly, an economic rationale—for protecting people from unsafe drinking water, flooding and climate change. But how far can ecosystem services approaches be taken to protect biodiversity? This is a subject of considerable concern and attention among conservation scientists, who are assessing the degree to which alternative conservation goals conflict or reinforce one another (Balvanera et al. 2001; Chan et al. 2006). In this line, Stanford University recently joined forces with the world's two largest conservation groups, The Nature Conservancy and World Wildlife Fund. The partnership, which began in November 2006, is called The Natural Capital Project, and is currently at work worldwide. Networked with The Natural Capital Project is a complementary project funded by the UK's Leverhulme Trust and involving a group of English Universities (Cambridge, East Anglia, York, Cranfield and Leeds) which is applying the ESF to the Eastern Arc biodiversity hotspot region of Tanzania. The overall aim is to replicate and scale up the many promising efforts underway that make conservation a compelling choice, on both moral and economic grounds.

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