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ECOSYSTEM SERVICES AND ECONOMIC THEORY: INTEGRATION FOR POLICY-RELEVANT RESEARCH

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Abstract. It has become essential in policy and decision-making circles to think about the economic benefits (in addition to moral and scientific motivations) humans derive from well-functioning ecosystems. The concept of ecosystem services has been developed to address this link between ecosystems and human welfare. Since policy decisions are often evaluated through cost–benefit assessments, an economic analysis can help make ecosystem service research operational. In this paper we provide some simple economic analyses to discuss key concepts involved in formalizing ecosystem service research. These include the distinction between services and benefits, understanding the importance of marginal ecosystem changes, formalizing the idea of a safe minimum standard for ecosystem service provision, and discussing how to capture the public benefits of ecosystem services. We discuss how the integration of economic concepts and ecosystem services can provide policy and decision makers with a fuller spectrum of information for making conservation–conversion trade-offs. We include the results from a survey of the literature and a questionnaire of researchers regarding how ecosystem service research can be integrated into the policy process. We feel this discussion of economic concepts will be a practical aid for ecosystem service research to become more immediately policy relevant.

Key words: benefits capture; conservation; ecological economics; ecosystem services; marginal analysis; payments for ecosystem services; safe minimum standard.

INTRODUCTION

Integrating economic and ecological sciences into an operational decision support system has been noted to be a key step for global conservation and sustainability (Millennium Ecosystem Assessment 2005). The concept of ecosystem services has been developed in support of this agenda. Couching ecosystem service research within economic theory gives us one way to move to a more structured engagement between biophysical science, social science research, and policy. The significance of a growing ecosystem service research agenda is evidenced by several major global initiatives. The World Bank (2004), World Wildlife Fund, The Nature Conservancy, and Conservation International have all initiated conservation programs based on ecosystem services, and the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005), based on over 1300 scientists' input, was structured explicitly around the concept. In the peer-reviewed literature, a keyword search for ecosystem services revealed over 1165 papers, of which more than 60% of them have appeared since 2003 (Web of Science key word search on 25 September 2007: ecosystem services, or ecological services, or environmental services).

In this paper we first define ecosystem services. Then we use a simple, theoretical, economic framework modified from Pearce (2007), to discuss three key issues that ecosystem service research must address to be

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PLATE 1. Humans rely on ecosystems for a wide range of services and benefits including soil formation, soil retention, nutrient cycling, and primary production. Here a rural farmer is tending plants in the Eastern Arc Mountains of Tanzania. Much of the world's rural population relies directly on ecosystem services and benefits for daily livelihoods and subsistence. Photo credit: B. Fisher.

operational as a decision support system. These are marginality, infrastructure or insurance value, and the ability to capture nonmarket ecosystem services through some institutional arrangement. We explore these issues in detail, examining difficulties in formalizing them in a way that is useful for policy development. We close with the results from a survey of the literature and a questionnaire of researchers regarding how ecosystem service research can be integrated into the policy process. We feel that, through incorporating a simple economic framework into ecosystem services research, future work in this field can be more immediately policy relevant. In our world of rapid global environmental change, we feel this is essential.

Ecosystem Services Defined

Ecosystem services are intrinsically anthropocentric, and resulting arguments for conserving nature to ensure their continued delivery are in addition to, not in place of, ethical and scientific ones (Costanza et al. 1997, Daily 1997, Turner et al. 2000, Millennium Ecosystem Assessment 2005, Costanza 2006; see Plate 1). There seems to be a consensus on a general meaning of ecosystem services. A few definitions in the literature are repeatedly cited (Costanza et al. 1997, Daily 1997, Millennium Ecosystem Assessment 2005). The Millennium Ecosystem Assessment (2005) defines ecosystem services as "the benefits people obtain from ecosystems." This definition is general by design, and while it provides a context for discussion, it falls short as an operational definition for use such as accounting (Boyd and Banzhaf 2007), landscape management (Wallace 2007), or valuation (Fisher and Turner 2008). The Millennium Ecosystem Assessment divides ecosystem services into supporting, regulating, provisioning, and cultural services. While this typology is useful as a heuristic tool, it can lead to confusion when trying to assign economic values to ecosystem services. For example, in the Millennium Ecosystem Assessment, nutrient cycling is a supporting service, water flow regulation is a regulating service, and recreation is a cultural service. However, we see the first two as providing the same service, usable water, and the third (e.g., recreation on a clean, navigable river) turning the usable water into a human benefit (i.e., the endpoint that has a direct impact on human welfare). If all three Millennium Ecosystem Assessment services were to be individually valued and added to a cost–benefit analysis, we would commit the error of double counting, as the intermediate services are by default included in the value of the final service.

Despite the proliferation of interest in ecosystem services there have been relatively few attempts to define the concept clearly to make it operational (De Groot et al. 2002, Boyd and Banzhaf 2007, Wallace 2007). Drawing largely on Boyd and Banzhaf (2007), we propose that ecosystem services are the aspects of ecosystems utilized (actively or passively) to produce human well-being. (Boyd and Banzhaf [2007] advocate that ecosystem services are only the *end products* of benefit to human welfare such as a bass population or surface waters.)

Defined this way, ecosystem services include ecosystem organization (structure), operation (process), and outflows, if they are consumed or utilized by humanity either directly or indirectly. At the same time, delineating between direct and indirect consumption of ecosys-

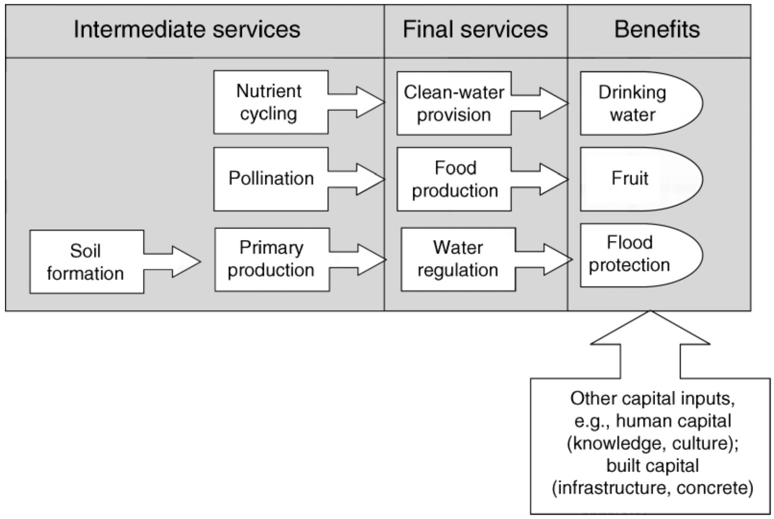

FIG. 1. Stylized relationships among representative intermediate services, final services, and benefits (the complexity and interactions are simplified for heuristic value).

tem services will be helpful for natural capital accounting systems (Boyd 2007) or economic valuation. We therefore recommend the qualifiers *intermediate* and *final* as used in conventional economic accounting systems. For example, food provision is a final service, whereas pollination is an intermediate service. The benefit here is food for consumption. In short, ecosystem services are the ecological phenomena, and the benefit is the thing that has direct impact on human welfare. Benefits are typically generated by ecosystem services *in combination with* other forms of capital like people, knowledge, or equipment, e.g., hydroelectric power utilizes water regulation services of nature but also needs human engineering, concrete, etc. Fig. 1 provides a few examples of this delineation.

By separating ecosystem services into intermediate and final services and benefits, we explicitly understand that in accounting and valuation exercises only the benefits generated by the final services can be aggregated, and hence, avoid double counting. Note that the same service can generate multiple benefits (e.g., flood prevention, drinking water, and recreation), and these can be added together.

As well argued by Boyd and Banzhaf (2007), this delineation between intermediate service, final services, and benefits is not strict. Services are often a function of beneficiary's perspective. For example, water regulation services provided by a vegetated landscape might be valued as a final service to someone interested in a steady water supply, but valued as an intermediate service to someone interested in a final service of usable water for recreation purposes like boating. Our inclusive consideration of ecosystem services also allows us to consider that final services and benefits flow from ecosystem functions or processes that are also intermediate services. This is important because it frames an important part of human welfare through services provided by ecosystems throughout the system, not just as the final step. Making this link explicit throughout socio-ecological systems was the important step made in Daily's (1997) Nature's Services.

ECONOMIC FRAMEWORK ECOSYSTEM SERVICES: WHAT CAN WE DO?

Economics is essentially the study of how humanity provides for itself (Heilbroner 1968), and humanity largely provides for itself by standing on the shoulders of natural systems. Therefore, an economic framework for ecosystem service research is logical. In Fig. 2, we adapted a conceptual framework from Pearce (2007) that links ecosystem services to human welfare with a simple supply-and-demand relationship. The x-axis represents the level of ecosystem service provision, aggregated here across services for a particular area. The y-axis measures marginal human welfare (here in monetary terms, but other metrics, such as lives saved, could be used). The downward-sloping demand curve, $D_{ES(M)}$, refers to marketed ecosystem service benefits, such as timber and fish, where the dollar value represents the market's willingness to pay for one more unit, i.e., the marginal value. Thus, as ecosystems are converted and supply decreases (moving left on the x-axis), the value we ascribe to the next unit increases (moving up the y-axis). $D_{ES(MNM)}$ is the demand curve for all ecosystem service benefits, including those that are not traded, such as flood protection. Because most ecosystem services are nonmarket services (public goods), we expect the $D_{ES(MNM)}$ demand curve to be considerably above the $D_{ES(M)}$ curve.

As for the supply curve, MC_{ES} represents the marginal cost of acquiring and managing additional units of ecosystems, such as hectares of land, as well as the marginal value of any opportunity costs (from forgoing alternative uses). The positive slopes reflect the expectation that providing each additional increment of an ecosystem service will be increasingly costly. In this figure, we also suggest that the rate of this increase could itself increase (the second derivative is positive). The safe minimum standard (SMS), or the minimum quantity of ecosystem structure and process (including diversity, populations, interactions, etc.), that is required to maintain a well-functioning ecosystem capable of supplying services. There is high uncertainty about just where this level is, and it surely will be different for different ecosystem services (Dobson et al. 2006). The two points ES_{MIN} and ES_{OPT} come from something called the equimarginal principle in economics, where the cost of providing an extra unit (of ecosystem services) is equal to the benefits gained from that unit (demand). For example, ES_{MIN} is the point where only marketed services of a landscape are provided (demanded). The marginal cost of providing that last unit of demand (i.e., cost of management, land purchases, and so on) is equal to the gains you receive from providing it. If you were to provide any more, the cost would outweigh the benefit. So, if trees only have value as marketed timber, the market will only pay for plantations and will not likely produce the optimal level of forest diversity and cover (ES_{OPT}) to supply other services such as biodiversity existence or perhaps even water regulation. A few general implications of considering ecosystem services within this economic framework emerge: (1) There is a fundamental uncertainty regarding the minimum level of ecosystem structure needed to provide a continual flow of services (SMS, infrastructure value). (2) A serious under-provision of ecosystem services will occur if only market benefits are considered, i.e., $ES_{MIN} < ES_{OPT}$.

Similar to the uncertainty surrounding a safe minimum standard level, $D_{ES(MNM)}$ will be difficult to make operational, since we will likely never be able to capture the true value of ecosystem service provision. Therefore, any demand curve for (or valuation attached to) ecosystem services would represent a lower bound. Further, monetary valuation is not always necessary or desirable. The *y*-axis could represent an index like vulnerability, lives saved, or happiness, depending on what the policy question is that drives the research. Understanding tradeoffs or cost effectiveness does not require monetizing the benefits, which can be difficult and imprecise (see Kahneman et al. 1993, Bateman et al. 1997*a*, *b*).

Here we focus on three key insights from this framework that should help to operationalize ecosystem services research as a decision support system. They are, as noted on Fig. 2, (1) the importance of marginal ecosystem service assessments, (2) understanding and investigation of a safe minimum standard level of ecosystem structure and function, and (3) the importance of capturing the benefits provided by nonmarketed ecosystem services, through some type of institutional arrangement.

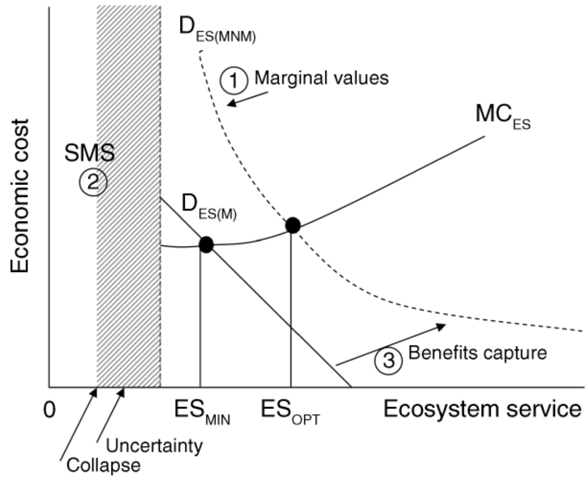

FIG. 2. An economic framework for ecosystem service provision offers three main insights for policy relevant research. (1) Ecosystem services should be studied as marginal changes in landscapes or seascapes. Researchers should ask questions such as "Does the conversion of one more hectare of forest to agriculture represent a beneficial trade-off?" This should lead to further questions of "Who benefits/loses?" and "Where is the benefit realized?" (2) At some level of degradation most systems will collapse. Knowing where this point is (safe minimum standard [SMS], i.e., some minimum level of structure or process) is crucial for point 1 (appropriate evaluation) and point 3 (policy integration). (3) Because most ecosystem services are public goods, the market will not provide an optimal level but only $D_{ES(M)}$, the demand curve (for marketed ecosystem service benefits). For optimal ecosystem service provision we need mechanisms to provide for nonmarket services, moving to $D_{ES(MNM)}$, the demand curve for all ecosystem service benefits, both marketed and nonmarketed. The supply curve, MC_{ES}, represents the marginal cost of acquiring and managing additional units of ecosystems; ES_{MIN} is the point where only marketed services of a landscape are provided (demanded); ES_{OPT} is the optimal level of forest diversity and cover to supply other services. For an explanation of terms, see *Economic framework ecosystem services: What can we do?*

WHAT SHOULD WE DO?

Understanding "margins"

The first concept that Fig. 2 helps us to acknowledge is the importance of marginality. Demand curves are marginal curves, and this aspect is essential if we are going to formalize and quantify the role that ecosystem services play in generating human welfare. Marginality denotes that the value of a particular service, regardless of the metric, is a function of (small) changes in the flow of that service. In the traditional marketplace, the value of a good, say, a pair of trousers, is estimated to be the price we would pay for an additional unit of that good, i.e., one more pair. Alternatively, it is the price we would pay to prevent losing one unit. So it is with ecosystem services. When forest services such as fuel wood provision are abundant relative to demand, our willingness to pay for an additional unit would be quite small. However, when the service benefit becomes scarce, the value we ascribe to the service will increase. This concept is important in making ecosystem service research policy relevant because it is at the margin where policy and economic decisions operate (Turner et al. 1998). The total value of the global forests, for example, is not useful for informing practical forest policy (Pearce 1998, Bockstael et al. 2000).

We can, however, make meaningful and informative valuations by recognizing that economic value is a function of marginal changes. For example, does the conversion of one more hectare of forest to arable land represent a beneficial tradeoff? We can imagine cases, particularly in the developing world, where this type of trade (i.e., conversion) still provides a net welfare benefit (Turner et al. 2003). However, it is not always clear how to quantify a "marginal change," especially with respect to complex systems like ecosystems (Turner et al. 1998). Despite the fact that hundreds of valuation studies have been conducted globally, the concept of marginality appears in only a handful (Balmford et al. 2002, Turner et al. 2003).

Before considering marginal analysis two things need to be clear. First, the scale has to be meaningful. It is neither meaningful nor appropriate to consider, for example, the loss of all the world's forests as "the next unit." Applying a value to the entire world assumes that the next unit lost will be all the forests on earth. Marginal analysis will likely have to consider a landscape as the largest possible region of change. Acknowledging that a "landscape" scale is somewhat arbitrary, it would have to be defended by the research approach. A likely guide for thinking about the "next unit" is considering what is the geographic extent that a policy decision can encompass (e.g., extending a forest reserve within national borders). This is an especially important point when considering that ecosystem services operate on several scales from the global, as in climate regulation, to very local as in soil formation. So, when considering policy for climate regulation, "the next unit" for decision makers might just be on the scale of whole countries, which will not likely lend itself to marginal valuation. Secondly, marginal analysis assumes that the response of an ecosystem to a small increase or decrease in structure or function does not result in large step changes in the services provided, such as might happen if the system were to "flip" from one equilibrium to another. For example, in the Aleutians, a decline in sea otter populations has led to a large increase in sea urchin biomass collapsing healthy kelp systems (Estes et al. 1998). In the context of Fig. 2, this means that for marginal analysis, we have to be on the demand curve away from the SMS zone. This is easier said than done, and future research will have to address just where this SMS zone lies for natural capital stocks and the services that flow from them, acknowledging that in some cases we might already be past the SMS zone. If these two criteria are met then marginal analysis is likely to be meaningful and appropriate.

On the ground, we can think of a few straightforward ways for researchers to incorporate marginal analysis into ecosystem service investigations. To do this requires knowledge of the drivers and pressures on the systems under study, as well as an understanding of how the system is changing or might change from its current state. We can call this its transition path. It is important to know if the transition path of the system is "stepped," as in the loss of a full coral reef system, or is it "relatively smooth" such as in species invasions creeping into a region slowly over time. By understanding this, we can force the analysis to consider losses or gains in service provision or economic value between two distinct states of the system. For example, if research finds that current land cover supports a pollinator population of X, then the services provided by this population or the value added to local agriculture by this population can be compared against a projected pollinator population Y, which is determined by likely future land cover changes. Likewise, researchers can envision, say, a 10% gain in service provision or a 15% loss. With this researchers can model what this means in terms of changes in the economic value of a service or even how the flow of services is distributed across a landscape. Assessing the losses or gains (read winners or losers) spatially can provide a practical way in which economic valuation can meaningfully be integrated into ecological studies for aid in decision-making.

A key point also highlighted by the conceptual figure is that, when investigating ecosystem services, we must consider the additional costs incurred for improved or continued provision. The driving question would be, "Does the benefit of increased ecosystem functioning outweigh the cost of obtaining it?" This is the reason for the MC_{ES} curve on Fig. 2. Any point to the left of ES_{OPT} represents a place where the benefits outweigh the costs. Identifying "small-loss, big-gain" (Defries et al. 2007) tradeoffs should be an important first step for ecosystem service research. For example, multiple studies have recognized the importance for pollination services of small forest patches near coffee plantations (Ricketts et al. 2004, Priess et al. 2007): a small loss in arable land but a big gain in coffee yield. Additionally, this small-loss, big-gain tradeoff has been advanced for focusing development away from ecologically sensitive bird habitats in the Greater Yellowstone Ecosystem (Defries et al. 2007).

Insurance for ecosystem service provision

The second point we can draw from Fig. 2 is the importance of knowing where the safe minimum standard zone lies. Ecosystems do not always respond to change in a simple manner, but instead behave nonlinearly, or flip into alternative steady states. Most ecosystems likely behave in these manners at some level of conversion or degradation (Balmford and Bond 2005). For example, shallow lakes have been shown to move from oligotrophic to eutrophic quite rapidly after

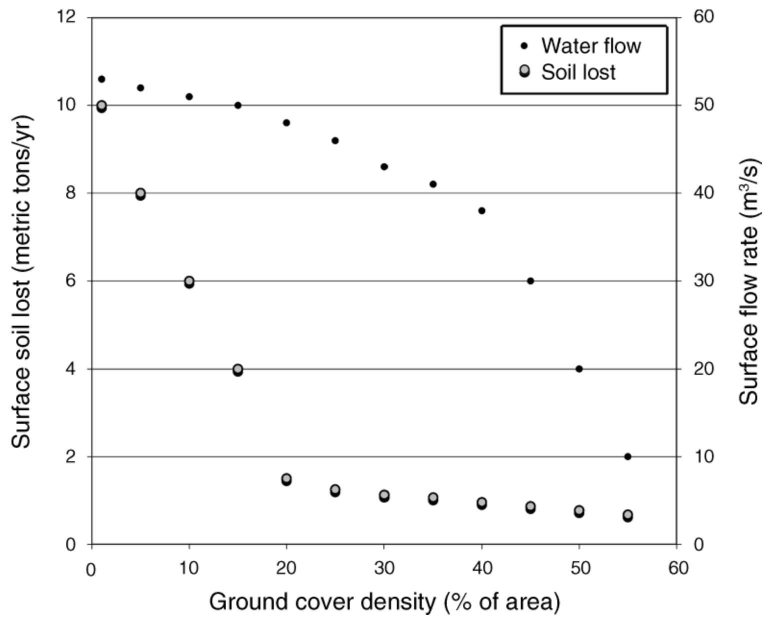

FIG. 3. Hypothetical relationships between ecosystem structure and service provision measured by proxy indicators. In this example, ground cover density represents some measure of ecosystem structure. The ecosystem services of soil retention and water regulation are measured by soil loss and flow measurements, respectively.

continued, but gradual, increases in nutrient loads (Scheffer et al. 1993, 2001, Carpenter et al. 1999). Likewise, Jamaican reef systems seemed to have flipped from herbivore-dominated systems to algae-dominated ones after over-fishing and disease caused a crash in herbivore populations (Hughes 1994). Now despite the presence of a certain food supply, herbivore populations have not been restored. In these cases, marginal analysis is not the appropriate assessment method, and we need to think about risk or uncertainty, and a zone of safe minimum standard. For example, if we are close to the minimum area of tropical forest needed to maintain a humid core that is resistant to forest fires, then a marginal analysis would be ill fitting, since the next unit of conversion might change the entire system from rain forest to dry forest.

The SMS zone in Fig. 2 is meant to represent this minimum level of a well-functioning ecosystem, one required to sustainably supply ecosystem services at a particular site. This level will be differently defined for each ecosystem service considered. For instance, the minimum area of forest required to supply pollination services will differ from the area of the same forest needed to supply water regulation services. Due to nonlinearity and time lags in how ecosystems respond to change, managing natural systems for ecosystem services will include some consideration of the precautionary principle. But the question remains of where to draw the line and how to manage systems under continuing uncertainty. This is partly an empirical question, and research addressing this on the scale of ecosystems is in the nascent stages (Gou et al. 2000, Carpenter et al. 2006).

However, at the level of species, much work has been undertaken to understand the role of increased biodiversity on ecosystem functions in the face of perturbation. This research is often regarded as investigating the "insurance value" of biodiversity. This area has seen a lot of experimental advances in the past decade, and provides a blueprint for future research regarding a similar and subsequent question: How much ecosystem function or structure is needed to insure a supply of ecosystem services? For example, the research by Tilman and Downing (1994) and Tilman (2006) in manipulating the number of grassland *species* to study ecosystem functioning, can be extended to manipulate structure or vegetation cover to study a particular service. In fact, it may be possible that many of the microcosm and plot scale studies of biodiversity–ecosystem functioning (BDEF) can be revisited to answer structure–service relationships. Larger scale manipulations and landscape-level statistical analysis should also be a part of this research agenda, so that the theory of safe minimum standard can be robustly and empirically defined.

Fig. 3 illustrates the sort of outputs that could come from such investigations. A study of vegetation densities on hill slopes could indicate the level of structure needed to supply a given level of soil retention and water regulation during storm events. We can imagine such products from plot studies, or larger sub-watershed manipulations. Statistical and simulation modeling exercises will also be needed at some scales and in some contexts. The work by Guo et al. (2000, 2001) offers an example of this, linking forest structure (canopy density, litter composition) with water regulation for downstream services, in their case, water provision for hydroelectric power. Additionally, it has also been reasoned that different trophic levels are mainly responsible for supplying different services, such as soil formation being mainly a function of the decomposers (Dobson et al. 2006). So trophic considerations in ecosystem service research are bound to help in determining a safe level of structure.

Benefits capture: demand for nonmarket ecosystem services

The third important insight gained from considering ecosystem service research within the economic framework of Fig. 2 is in terms of the difference in demand curves between marketed services and all services. We can see that without incorporating nonmarket services into policy decisions society would wind up at ES_{MIN}, well below the economically optimal point for human welfare (ES_{OPT}). Moving from $D_{ES(M)}$ to $D_{ES(MNM)}$ requires that we incentivize or invest in the provision of nonmarket ecosystem services. We call this benefits capture. There are several societal arrangements available for benefits capture, including formal institutional arrangements and informal communal arrangements. For our discussion here, we focus on formal institutional arrangements, but briefly consider the importance of informal arrangements at the close of this section.

There are several currently operational mechanisms for benefits capture functioning through formal institutional arrangements. A typical categorization of these mechanisms is to divide them "market-based" and "regulatory," although this dichotomy is somewhat artificial since market-based mechanisms are regulatory, but use markets rather than legislation to achieve their regulatory goals cost effectively (Russell and Powell 1999). These market-based mechanisms are becoming increasingly popular to achieve conservation goals and are already being used to compensate providers of ecosystem services.

Instruments here include taxes and user fees to curb undesirable behaviors, as well as payments and subsidies to encourage desirable behaviors. For example, requiring licenses for logging, fishing, and hunting is a way of using a price mechanism to limit certain behaviors. Another example is Sweden's decision to tax nitrogen use at U.S. $0.25 per kilogram, which has apparently reduced nitrogen use in agriculture by 10% and helped to minimize water quality problems (OECD 2001:109). As far as encouraging desirable behaviors, a popular example is tax relief for wetland protection, as used in many OECD countries. Another example is Brazil's ICMS ecologico, which redistributes sales tax to municipalities in proportion to conservation activities (Grieg-Gran et al. 2005).

One currently popular group of price-based instruments is payments for ecosystem services (PES). The PES approach has been recommended particularly for use in non-OECD countries, where regulatory and taxation systems are likely to be weak. In these schemes, landowners are compensated for providing services whose production previously went unrewarded. The Mexican government has such a system to pay landowners for conserving forest in hydrologically important catchments (Pagiola et al. 2005). The best-known payment scheme is Costa Rica's, which was established in 1995 and compensates landowners for carbon sequestration, water regulation services, biodiversity conservation, and scenic beauty provision. The measurement proxy is area of land forested and payments are around U.S. $45·per hectare per year. A recently conducted 10-year review showed that this program still has excess demand (Ferraro 2008) meaning that the program is popular with stakeholders and that the sellers might be "over-paying."

Other market-based instruments for capturing ecosystem service benefits include marketable permits, such as tradable fishing quotas as used in Canada and New Zealand (Dewees 1998), and market credits, such as carbon credits and wetland banks. Setting the quantity or scale of activity through the market by issuing permits or credits creates a scarcity and therefore encourages more efficient use and allocation of the resource in question.

Despite the popularity and increasing theoretical and empirical investigation of compensation mechanisms for ecosystem service provision, a number of non-trivial obstacles exist for successful implementation. Difficulties include establishing clearly defined property rights when necessary, being able to observe actual behavior to measure and verify outcomes, linking payments directly to desired outcomes and not some proxy outcome, getting prices correct, overcoming cultural disjoints and equity concerns, geographically concentrating undesirable behavior, and financing the mechanisms. Box 1 provides a further description of the obstacles facing market-based conservation mechanisms.

Some of these obstacles have been revealed in the Costa Rican program, where participation has been correlated with larger properties, higher education, and absenteeism (Zbinden and Lee 2005), indicating that market power and information asymmetries might exist and hinder the efficiency of the allocation mechanism. Also, some systems may intensify poverty, and hence, degradation by exacerbating land take and excluding traditional use and users (Rosa et al. 2004, Pagiola et al. 2005).

These formal institutionalized mechanisms are created as recognition of the importance of traditionally non-marketed ecosystem services. This is ostensibly moving the society beyond the $D_{ES(M)}$ curve. In addition to these formal institutionalized arrangements are small-scale informal arrangements. It is often assumed that

Box 1. Obstacles to market-based conservation mechanisms

Property rights

The establishment of property or assignment rights over the good or service being provided is often necessary, but seldom easy. Designating who "owns" resources such as wetlands, or who has the "rights" to utilize the waste absorption capacity of the atmosphere is fraught with governance and equity issues. Without property rights, incentives to invest (divest) in sustainable (unsustainable) actions often do not exist, and free riding is likely.

Measuring and monitoring

The ability to measure and monitor service provision is not straightforward with public or common pool resources. Individual behavior might not be observable, as it might take place in hinterlands, or the service might not lend itself to measuring, such as with pollination services.

Directly linking behavior and compensation

In connection with the difficulties of measuring and monitoring ecosystem service provision, direct links between service provision and compensation might not be possible and therefore allow defection. If we are interested in biodiversity provision, it might be impossible to measure and monitor the service directly and therefore we might use forested area as a proxy. In this case, we might achieve the proxy goal without the associated biodiversity goal (e.g., pressures from hunting could hinder the latter, but not affect the proxy).

Correct pricing

For price-based mechanisms, finding the correct price level to incentivize ecosystem service provision is not likely to be straightforward. For payment schemes, the payments would need to cover an agent's opportunity cost, but this is not always easy to elicit and the incentive exists for agents to overstate their opportunity cost, i.e., moral hazard. In Costa Rica, the land under payments contracts is more likely to be on steep slopes and inaccessible, suggesting that the buyer is overpaying. Information costs may be prohibitive (Pagiola et al. 2005).

Cultural hurdles

Price and quantity-based mechanism assume that to some degree a market institutional setup is common. In some places, assigning property rights to individuals or offering payments for expected behaviors may not be common practice or even acceptable. In several cases it has been shown that the price incentive is not always effective at changing behavior (Gowdy and Erickson 2005).

Externalizing

Conservation is often in danger of pushing deleterious activities to peripheries, i.e., toward places where schemes are not in place or enforced. Additionally, the translocation of services, such as wetlands banking, is unlikely to guarantee the same quality of services (Salzman and Ruhl 2006).

ecosystem service provision needs the help of markets and macro-regulation to overcome problems associated with externalities, public goods, and imperfect information. While this might be generally true at large scales, humankind has organized sustainable governance of public and common pool goods for thousands of years (Ostrom et al. 1999). Currently, there are more than 450 000 collective resource management groups in the world governing resources such as watersheds, irrigation systems, and forests (Pretty 2003). Through such arrangements, communities recognize that the provision of ecosystem services at the ES_{MIN} level is sub-optimal, and they move beyond market service provision toward ES_{OPT}. For successful provision of ecosystem services across scales, we will need to consider a variety of management options for effective and equitable outcomes, ranging from formal global arrangements to informal local ones.

Ecosystem Service Research And Decision-Making: What Has Been Done?

Further theoretical work and empirical studies on the connection between ecosystem services and human welfare will be critical for informing policies toward achieving sustainability (Carpenter et al. 2006). Societal preferences and changing public attitudes will also be a key aspect, but must ultimately be based on sound science. In order to understand where we stand now in this undertaking, we conducted a literature search for peer-reviewed articles that analyzed ecosystem services with either an explicit or potential policy interaction. The search extended up to the autumn of 2006 and utilized Web of Science, Elsevier/Science Direct, Blackwell Synergy, Google Scholar and Nature Valuation Financing Case Study Database (CaseBase). The search used the following filters for the inclusion of case studies:

TABLE 1. Summary of case studies used for policy review and survey.

Case study	Country	Ecosystem	Ecosystem services/benefits
1) *Ex post*			
Working for water/ invasive species removal	South Africa	fynbos, thicket, forest, grassland and semiarid savannah (rangelands) biomes	water regulation and supply (availability), provision of habitat (invasive species removal)
UK Reforestation Programme and Forest Services	United Kingdom	temperate forest	timber, recreational services, aesthetic values
Ecosystem services and green national accounting	Brazil	tropical forest	timber, NTFPs, recreational, global climate regulation, flood control, water regulation, erosion control, option benefits, existence benefits
Surat Thani (economic value of mangroves and role of community conservation)	Thailand	wetlands (mangroves)	storm protection, shore stabilization, control of soil erosion and flooding, nursery habitat; timber, charcoal, offshore fisheries; eco-tourism; non-use values
Florida Everglades	USA	wetlands	water quality and supply, flood protection, provision of habitat and biodiversity, disease control
Ecosystem service values and land use change	China	various: lakes/rivers, cropland, forest, urban, estuaries	biomes identified in Costanza et al.'s (1997) ecosystem services valuation model; most representative biome used as proxy for each land cover category
Ecosystem service values and land use change (San Antonio)	USA	rangelands (14 000 ha in Texas)	three watersheds looking at relevant biomes identified in Costanza et al. (1997)
Nordic forests under different management regimes	Scandinavia and Finland	temperate (boreal) forest	timber, agriculture, food, raw materials, carbon sequestration, recreation, existence values
Converted and conserved wetlands	Canada	temperate wetlands	agricultural/production, recreation, nitrogen fixation, water supply, habitat
2) *Ex ante*			
Ream National Park	Cambodia	coastal wetlands (mangroves)	fisheries, firewood, food, medicinal plants, construction materials, carbon sink, prevent saltwater intrusion and coastal erosion, storms and flood protection
Portland Bight Protected Area (PBPA)	Jamaica	various: wetlands, forests, marine (integrating terrestrial and marine PA)	fish/fisheries, wood (mangrove) forestry, tourism and recreation, coastal protection, carbon fixation, biodiversity/ habitat provision
Masoala National Park	Madagascar	tropical forest	timber, ecotourism, NTFPs, biodiversity products, watershed protection, carbon conservation
Ecosystem services vs. economic benefits of conversion (Brazilian Amazon)	Brazil	tropical forest	timber, NTFPs, water cycling, nutrient cycling, fire protection, watershed protection, tourism, carbon storage, biodiversity protection, recreational value, existence value
Kakadu Conservation Zone vs. mining profits	Australia	mixed forest	existence values

TABLE 1. Extended.

Policy relevance	References
Dual goals of natural capital restoration and poverty alleviation; linking ecological, economic, and social aims; ecosystem services valuation a key driver in understanding cost–benefit of invasive species. However, question of clash of socio-political and ecological priorities; regarded as success story although hindered by lack of consistent data monitoring and evaluation.	Van Wilgen et al. (1998, 2004), Milton et al. (2003), Binns et al. (2001), Le Maitre et al. (2002)
Designed to help the United Kingdom Forest Commission decision-making process. Prior to it, reforestation program was based solely on private benefits from timber production, so reforestation was mainly through coniferous trees. The study showed this was optimum decision from private point of view. When social benefits related to recreational services and aesthetic values (measured by the hedonic price method) were taken into account; however, broadleaf trees were the most beneficial option.	Garrod and Willis (1992)
Work combines green accounting methodology and TEV concept to estimate economic costs of past deforestation in Brazilian Amazon. Net present value of ecosystem services lost exceeds joint value income of the nine Amazonian states for every year of the study. Even Brazilian national GDP per capita growth of 0.7% p.a. falls to 0.3% p.a. when reductions in Amazon forest "stock" are included.	Torras (2000)
Market and nonmarket valuation illustrating policy failure in converting mangroves to shrimp aquaculture. Conversion for aquaculture delivered greatest private gains (neglecting external costs); global benefits (e.g., carbon sequestration) similar for intact and degraded systems; social benefits almost zero after conversion. TEV of intact mangroves exceeded aquaculture by ~70% (Balmford et al. 2002); Consideration of role of international compensatory transfer to support local conservation strategy.	Sathirathai (1998), Barbier and Sathirathai (2001)
One of largest natural capital restoration projects in the world (U.S. $8 billion over 30 years). Illustrates key ecosystem losses in terms of species, discharges, diseases, purification ability. Influence of ES paradigm?	Milon and Scrogin (2006), Schuyt and Brander (2004)
Multiple services considered and tracking site changes over time; using LANDSAT TM and/or ETM data sets to estimate changes in size of five land cover/land use categories; used previously published value coefficients (Costanza et al. 1997) to value changes in ecosystem services delivered by each land cover category.	Zhao et al. (2004)
Land use change 1976–1991 and effect on delivery of ES (using Costanza et al. (1997) coefficients) as well as services values per hectare. Loss in ES due to urbanization over 15 years: $6.49/ha/yr. TEV not estimated as economic benefits from residential and commercial capacity from land use change not valued.	Kreuter et al. (2001)
Comparison of private value of forest land (according to value of timber production) to agricultural land (market price); distortion by agricultural subsidies. Contingent valuation showed WTP of $4500–$6500/ha. Question of type of institutional arrangements needed to ensure broad set of forest benefits; preservation possible highest values use of marginal units of forest.	Hoffren (1997), Holgen et al. (2000), Hoen and Winther (1993)
Economic case in favor of conservation is clear once subsidies have been removed even without inclusion of the full set of environmental services.	Van Vuuren and Roy (1993), Turner et al. (2003)
Total value, net value, and average value/household estimated. Justify economic benefits associated with declaring the area a national park. Aimed to show reliance of community livelihoods on park resources and to quantify local opportunity costs of switching from activities that degrade wetland biodiversity.	Emerton (2005)
CBA (using NPV, incremental costs and benefits) for establishing PBPA. To illustrate benefits over costs of managing such an area. Results to feed into management plan.	Cesar et al. (2000)
Compares benefits of conservation with those of alternative uses (logging and agriculture) at local, national, and global scales. Conservation offered superior benefits on both local and global scales, but conversion is most beneficial option from national point of view. As country decisions about national parks are made at this last level, the study calls for creation of a market for protection of tropical forests (specifically for mitigating climate change).	Kremen et al. (2000)
Compares costs of deforestation (i.e. conservation benefits) with benefits. Deforestation costs are divided into private, local public, and global values. Special attention to double-counting leads to discarding nutrient cycling benefits. Deforestation benefits are measured by the impact each deforested hectare has on rural GDP. Concludes that, at present, deforestation benefits are equivalent to global costs. Suggests that as deforestation advances, its global costs will rise.	Andersen et al. (2002)
Was part of decision-making process by Australian government about whether to allow the opening of a new mine in Kakadu Conservation Zone or to integrate it to Kakadu National Park. Study was based on contingent valuation surveys and concludes that conservation benefits to Australian people were greater than potential gains from mining activity. Australian government opted for conservation of the area and publicly stated that decision due to concerns with aboriginal people.	Carson et al. (1994)

TABLE 1. Continued.

Case study	Country	Ecosystem	Ecosystem services/benefits
Kenyan parks	Kenya	savannah	timber, ecotourism
TEV and opportunity costs in the Brazilian Amazon	Brazil	tropical forest	timber, NTFPs, ecotourism, carbon storage, option value (biodiversity), existence value
Kala Oya River Basin	Sri Lanka	wetlands; traditional "tank systems" in dry-zone landscape	water (crops, livestock, domestic), food (plants, fish), plants (ornamental/ ceremonial use), habitat (breeding areas), regulating: (flood mitigation, water purification, nutrient retention)
Dutch Wadden Sea	Netherlands	coastal wetlands, marine	issue of impacts of gas extraction; multiple functions studies
Pantanal	Brazil	wetlands	multiple: regulating, provisioning, etc.
Leuser Ecosystem and National Park, Northern Sumatra	Indonesia	tropical forest	water supply, fisheries, flood and drought prevention, agriculture and plantations, hydro-electricity, tourism, biodiversity, carbon sequestration, fire prevention, non-timber forest products and timber
Value of Zambezi wetlands	Southern Africa (Zambia, Zimbabwe, Botswana)	wetlands	inventory of production and information functions made for each wetland; study has been limited to quantification of use values
Natural assets: boreal forests	Canada	temperate (boreal) forests	included timber from forests, oil and gas; and hydroelectricity and ecosystem services provided by wetlands and forests, such as purifying water, regulating climate and oxygen
Value of timber and NTFPs, Selangor	Malaysia	tropical forest	timber and NTFPs; water supply and regulation; recreation; maintenance of carbon stocks, endangered species
Mount Cameroon; comparing low-impact longing and stepped land use change	Cameroon	tropical forest	timber and NFTPs; social benefits such as sedimentation control, flood prevention, carbon storage
Mulanje Mountain valuation study: current and projected use	Malawi	tropical forest	water regulation, water provision, timber and non-timber products
Marshes on east shore of Lake St. Clair in southwestern Ontario	Canada	wetlands (freshwater marshes)	agricultural yields, hunting, angling, trapping, nursery and habitat
Philippine coral reef destruction	Philippines	coral reefs	tourism and recreation (diving), coastal fishing, habitat, coastal protection
TEV over alternative land uses	Cameroon, Sri Lanka, and Malaysia	tropical forests	goods and services location specific; affected by ecosystem attributes, cultural values, and extraction and intensity of use
Changes in TEV under different management scenarios	El Salvador	wetlands (mangroves)	timber, fuel wood, food/fisheries (artisinal and industrial shrimp and fish), erosion prevention, carbon sequestration benefit
3) Informing			
Bhitarkanika Mangroves	India	wetlands (mangroves)	storm and coastal protection, cyclone mitigation, flood control, erosion prevention
New Orleans and Hurricane Katrina	USA	wetlands	storm and coastal protection, hurricane mitigation, flood control, erosion prevention
Mangroves as tsunami defense	Sri Lanka	wetlands (mangroves)	storm and coastal protection, tsunami defense, flood control, erosion prevention

TABLE 1. Continued. Extended.

Policy relevance	References
Compares conservation benefits from ecotourism and sustainable forestry to potential returns of agriculture and livestock production. Concludes that on national level, benefits of alternative activities are higher than those of conservation. Suggests that including global values in the analysis would show that conservation is the global optimum choice. States that this situation is not sustainable in long run and that international community should bear part of the costs of conservation.	Norton-Griffiths and Southey (1995)
Compares unit TEV of standing Amazon forest with rental value of land. Argues that this is a better measure of opportunity costs in the region than the land's selling value because of medium/long-term property rights uncertainties. Concludes that national ecosystem services benefits are usually below opportunity costs and that it is necessary to internalize part of the forest's global external benefits.	Seroa da Motta (2005)
Economic values articulated supported their inclusion in regional land and water use decision-making. Valuation also played important role in development of different scenarios for various tank management options which fed into a cost–benefit analysis using both quantitative and qualitative indicators. Value of wetlands in livelihood and biodiversity terms to aid conservation.	Vidanage et al. (2005)
CBA which undermined assumptions of original industry study. Social cost–benefits and economic arguments fueled political debate at the time.	Wetten et al. (1999), Schuijt (2003)
Costanza et al. 1997 study re-estimated with more detailed and accurate data at local level; better understanding of the potential for the people of Pantanal to benefit from environmental stewardship.	Seidl and Moraes (2000), Schuyt and Brander (2004)
Rain forest decline causing loss of ecosystem services. Study assessed economic consequences of deforestation vs. conservation vs. selective use. TEV for ecosystem over 30 years calculated under different scenarios. Despite economic benefits of conservation, deforestation continues largely due to political power of the logging and plantation industries as well as wide dispersion of the main beneficiaries of conservation.	Van Beukering et al. (2003)
Wetland values derived from each function are estimated based on market prices. Results show that flood-recession agriculture is the main contributor to the TEV of wetlands in the Zambezi basin. Conservative estimate of the total value of the wetlands.	Seyam et al. (2001), Emerton and Bos (2004)
Study aimed to begin to identify, inventory, and measure full economic value of ecological goods and services provided by Canada's boreal region; aimed to give Canadian decision makers a boreal natural capital "balance sheet" for assessing sustainability, integrity, and full economic value of boreal region.	Anielski and Wilson (2005)
Quantified net marginal benefits of human uses under various management regimes, e.g., comparison of reduced impact and high-intensity logging whereby the latter drew greater private benefits but reduced social and global benefits. TEV of sustainable forestry 14% greater than otherwise.	Kumari (1994)
Forest conversion delivers higher private benefits; conversion to palm oil and rubber plantations yielded negative private benefits; sustainable forestry yielded highest social and global benefits. TEV of sustainable forestry 18% greater than small-scale farming; plantations had negative TEV.	Yaron (1999, 2001)
Effort at identifying and developing ways for conservation of natural resources to pay its own way through natural products industry development. In particular an exploration of whether water could become a "saleable" product in the Malawi context, through PES, and provided information for watershed management.	Hecht (2006)
Private and social returns from wetland preservation vs. conversion (draining for agricultural purposes). For three marsh types considered, conversion yielded net private benefits but TEV was 60% greater when wetlands remained intact.	Van Vuuren and Roy (1993)
Destructive reef exploitation (e.g., blast fishing) gave high initial benefits but followed by a far lower NPV of sustainable fishing; social benefits from sustainable use gave TEV 75% greater than destructive fishing.	White and Vogt (2000), White et al. (2000)
In all studies, forest conversion benefits short-term private gains but conservation makes economic sense when social and global benefits accounted for. Conservation strategy requires global community to provide incentives to local communities.	Yaron (2001), Batagoda et al. (2000), Kumari (1994)
Sustainable management option (i.e., felling only mature mangrove trees) delivers estimated NPV of $2344 ha/yr. Actual distribution of local benefits (through mangrove conversion) skewed away from poorest in society.	Gammage (1997)
Economic valuation of cyclone damage to houses, livestock, fisheries. Damage compared across three villages with different levels of protection and ES intact.	Badola and Hussain (2003, 2005)
Cost of reinstating natural infrastructure, ecosystem restoration (wetlands and coastal systems) to reduce vulnerability. Lessons learned in economical terms to feed into policy/management plans and rebuilding process.	Kunreuther and Pauly (2006), Costanza et al. (2006)
Effect of degraded/converted wetlands and mangrove forests in contributing to damage inflicted by tsunami. Implications for rebuilding/restoring natural defenses.	Dahdouh-Guebas et al. (2005)

TABLE 1. Continued.

Case study	Country	Ecosystem	Ecosystem services/benefits
Rewarding upland poor for environmental services (RUPES)	Indonesia, Nepal, Philippines	tropical and agroforests	watershed services (regulation, erosion control), carbon sequestration
Market-based instrument (MBI) approaches	Australia	various (mainly agricultural landscapes)	biodiversity, carbon, salinity mitigation; water quality, and NRM issues

Notes: Studies are divided into (1) *ex post* studies, where research occurs after an existing policy or decision context; (2) *ex ante* studies, where research has been part of the policy forming process; and (3) informing, where ecosystem service research has the potential to inform policy. Case studies in boldface indicate cases where researchers responded to our survey. Abbreviations are: p.a., per annum; NTFP, non-timber forest product; TEV, total economic value; ES, ecosystem services; GDP, gross domestic product; PES, payments for ecosystem services; and NRM, natural resource management.

(1) Cases where ecosystem service analysis helped to clarify existing policy decisions (*ex post*). (2) Cases where ecosystem service analysis has been an integral part of the policy process (*ex ante*). (3) Cases where an ecosystem service analysis has the potential to inform future policy (e.g., hurricane damage mitigation).

These three filters all select for important properties connecting ecosystem service research and policy. Filter 1 is important for selecting studies where good management has followed legislative requirements rather than economic valuations. This serves as an important learning exercise for future policy, showing cases where a detailed valuation exercise followed the management decision and where, subsequently unforeseen service benefits (e.g., tourism) have been realized. Filter 2 was very selective, since few studies in the literature make explicit policy linkages. This filter necessitates that some form of monitoring of implementation has been undertaken. This is rarely done in the studies themselves, and review and evaluation of such approaches seems to occur on an ad hoc basis at best. In order to fully understand this link, we needed to perform a policy appraisal or contact the researchers themselves for their perceptions of the policy impact of their work. Filter 3 allowed us to include studies that have potential to inform policy based on an ecosystem service analysis. We were quite exclusive with this last filter since the connections between ecological research and human welfare are deep and numerous, and therefore have great policy interaction potential.

The three filters revealed 34 cases. We summarize the studies in Table 1, indicating the study topic, country, ecosystem type, ecosystem services investigated, and policy relevance. In reviewing these studies, we found that almost all of them considered multiple ecosystem services rather than a single service. Almost all cases considered both marketed and nonmarketed goods and services, with the regulation services, such as water regulation, making up the bulk of the nonmarket considerations. This review provided us with several insights into the types of studies undertaken in regards to ecosystem services. The driving question being: To what extent have these studies incorporated the key points of marginality, safe minimum structure, and benefits capture (points 1, 2, and 3 in Fig. 1)?

This is what we found:

1) Marginality has not so far been considered requisite for ecosystem service studies. Most studies still just provide a snapshot in time that focuses on a current service delivery or total economic value (TEV). Explicit acknowledgment of the importance of changes in service delivery across disturbance states is rare (Balmford et al. 2002). This is where time-series data on land cover might be well integrated to service valuations. An example from our case study is Zhao et al. (2004), who used satellite imagery series to estimate changes in ecosystem services as a function of change in land cover over time. There were a few cases where service valuations were compared across alternative land uses, which can be considered a second-best option for research due to the fact that marginal changes are not always easy to define in dynamic and complex ecosystems (Turner et al. 1998). For example, Yaron (2001) compared the private and social benefits of three alternative uses of tropical forests in Cameroon: palm plantations, small-scale farming, and sustainable forestry. Likewise, Kremen et al. (2000) compared the local, national, and global benefits of conservation with those of logging or agriculture for forests in Madagascar. These types of studies offer a way forward for ecosystem service research and policy relevance by attempting to include some type of marginal change or transition. Moving toward understanding a more marginal change, such as 10% conversion, through modeling and scenario building, will be important for meaningful policy prescriptions (see Van Beukering et al. [2003] for an example of using future scenarios and value changes).

2) Most ecosystem service research does not explicitly investigate the SMS or minimum level of structure needed for continued service delivery. This is difficult to do at landscape-scale investigations of ecosystems services, but spatially explicit dynamic modeling may prove very useful here. None of our case studies explicitly investigated a SMS zone, but some do mention

TABLE 1. Continued. Extended.

Policy relevance	References
Dual goals of environmental/policy goals of biodiversity conservation and poverty alleviation; payment mechanisms in place to reward upland poor for land management practices; attempts at institutionalization with mixed results. Environmental services approach adopted; economic valuation only really featured (varying "success" in 3 of 6 sites).	Kallesoe and Iftikhar (2005), Van Noordwijk (2005)
Various case studies piloting MBIs in conservation and land management (NRM); trading mechanisms, auctions, and price signals to change behavior.	Proctor et al. (2002, 2007), Whitten et al. (2007), Whitten et al. (2002)

that thresholds will be crucial in future ecosystem service research. Field research may be guided here by the BDEF work, which has suggested some minimum level of species needed for sustainable supply of ecosystem services (see review by Hooper et al. 2005).

3) Not surprisingly, many studies did identify the problems of the market under-providing ecosystem services. This is not surprising because the field of ecosystem services was developed upon this realization. A third of the cases explicitly mentioned some benefits capture mechanism such as taxes, levies, and PES. The popularity of payments for ecosystem services schemes showed through as potential solutions in many projects, such as PES trials in the Working for Water program, and test sites in Southeast Asia under the RUPES program (see Table 1). At the same time, direct links from the initiation of the research to the policy process were found to be lacking many cases. This is more likely to be a function of how standard academic research is conducted, rather than a function of these particular studies. Nonetheless, the integration of ecosystem services analysis directly with agents and processes within decision-making arenas is largely absent.

In addition to connecting the case study with lessons about marginality, SMS, and benefits capture, the 34 studies revealed a few other insights. The main one being that the problem of double counting (for example, adding the value of a stand of trees for sustainable timber extraction to its carbon storage value) is rarely acknowledged in the literature. An exception is work like Anderson et al. (2002), who paid special attention to double counting by disregarding intermediate services (nutrient cycling) in their benefits aggregation. If ecosystem service research is going to provide decision makers with accurate information regarding the value of ecosystem services, then double counting needs to be acknowledged in every study. Utilizing the definition supported above (intermediate services $\rightarrow$ final service $\rightarrow$ benefits) should make the double counting issue transparent and unlikely.

Another finding comes out as an artifact of our literature review. In looking at Table 1, we see that most of the case studies are located in the developing world. We suggest that this is a function of our search for studies using the terms "ecosystem services" or "ecological services." These terms are quite new in the literature, but public goods research provided by nature has a long history in the developed world. Environmental economics has a strong history in this work with research on catchment management and broader cost–benefit analysis without using the term "ecosystem services." Future work should try to bring together the relevant literature from the developed world, which concerns ecosystems services, but does not employ the term.

ECOSYSTEM SERVICES AND POLICY INTEGRATION

To further study the level of integration between ecosystem service research and policy, we conducted a survey of the authors of the 34 papers in Table 1. The purpose of the survey was to find out the level of policy inclusion built into the design, funding, and findings of the research and also to see if some *ex post* assessment was carried out. Ideally, we would also survey and review policy documents in the locales where the studies took place, but simply surveying authors was deemed more efficient than such an in-depth policy appraisal. Of course there are biases and limitations to what information a qualitative survey of scientific authors can provide. However, we view the responses as indications of policy and management relevance of ecosystem service studies and the level at which scientists incorporate or consider policy-oriented steps in ecosystem service analyses. Recognizing this limitation, several respondents were very frank at the lack of policy traction of their work, and several offered the view that an ecosystem service argument or valuation was only a small input to the decision-making process.

We sent an e-mail questionnaire of eight questions (see Appendix) to the lead authors of the 34 studies in Table 1. In general, the questionnaire attempted to elicit the motivation for the research; if, and at what level, the research was used to influence a policy or management decision; whether a market-based incentive system was considered; and if a post-policy (post-research) review occurred. These questions were designed to understand the level of integration among the biophysical, social, economic, and political aspects of ecosystem services research.

We received 14 responses to our survey, a 41% response rate. The responses revealed that most of the research was initiated or commissioned by agents within the policy process. These were typically NGO–federal government collaborations or large international development organizations including the World Bank, Department for International Development (UK), and USAID (USA). Only three of the 14 studies were initiated solely out of academic interest.

The authors' perceptions of how their research interacted with the policy realm ranged from "no interaction" all the way to "influencing federal policy design." Guided by the responses, we can envision a spectrum of interaction possibilities for ecosystem service research and policy from low influence to great influence. These include (with actual responses in quotations):

1) distributing the research results to policy agents ("... the research was passed to relevant government and non-government bodies")
2) directly informing and engaging policy agents ("... presented results to municipalities and ministers")
3) providing influential support for current conservation initiatives ("... it was used to support policy initiatives")
4) informing a specific current policy debate ("... was used to inform the policy process, it enabled a better debate")
5) directly influencing government policy and investment ("... it was the major stimulus for the launch of the [conservation] program").

Additionally, seven of the 14 respondents indicated that market-based incentives were either being considered or were suggested. These included PES approaches, user fees, conservation credits, and tax breaks. We also included a question regarding any post-policy (or post-research) appraisal. The most common response to this question (Appendix: question 8) was that there was no post appraisal undertaken, or that the authors were unaware of any. A few authors suggested that it was too early to consider it, and others responded that they were continuing to monitor the situation.

Several of the general lessons highlighted by the authors surveyed provide additional support and caveats for the implementation of ecosystem service research. Several authors commented on how basic research into ecosystem service provision and subsequent dissemination can seriously enlighten the management processes and local stakeholders. One author explicitly stated that managing for well-functioning ecosystems provided services more cheaply and more reliably than typical built–capital responses: a fact that has been noted several times in the literature (VanWilgen et al. 1998, Dietz et al. 2003). Another said that the economic values of ecosystem services found through their research "supported the argument for their inclusion in regional land and water use decision-making."

Several researchers also noted the efficacy of monetary valuation for gaining traction in the decision-making process, as well as the importance of stakeholder participation for the same end. One response included "Community participation in the valuation process and scenario development ... facilitated a greater ownership of subsequent decision-making outcomes." Another stated that "there is little doubt that the monetary valuation of an ecosystem service, formalized in a cost–benefit analysis was the major stimulus for the launch" of a conservation program. One caveat offered regarding valuation studies, was that often valuation results are presented in a way that is meaningless for local stakeholders, and therefore unlikely to foster local buy-in. Additionally, government accountability and cultural sensitivity were also highlighted as critical for successful implementation. In the words of one respondent, "... with an unwilling (corrupt) government, no argument makes sense."

The responses to the questionnaire support the idea that ecosystem service research can be designed to have strong policy foresight, broad cooperation between policy agents and scientists, and possibly strong implementation effects. Keys to success are likely to include making an economic argument, delivering results in common language, elucidating tangible benefits to livelihoods in the short term, and multiple points of contact with those involved in the policy process. They also suggest that post-policy (or post-research) appraisal is often considered outside the scope of scientific research or requires extended cooperation between policy agents and scientists, something that is not currently de rigueur.

Conclusion

As highlighted by the Millennium Ecosystem Assessment, the ecological integrity of our world is rapidly changing. This will certainly affect human welfare. Our collective ability to manage these changes faces many obstacles including gaps in our ecological knowledge (Carpenter et al. 2006), shortcomings in our economic approaches (Barnes 2006), and flaws in our decision support systems and policy responses (Sachs and Reid 2006).

As currently conceptualized, ecosystem service research is relatively new. However, there is an emerging theoretical base, a growing understanding of how human and ecological systems are linked, and rising public awareness of the importance of well-functioning ecosystems. In this paper, we used a theoretical economic framework to highlight several concepts critical for formalizing ecosystem service research within a decision support system. Since political decisions often happen at the margin (i.e., what to do with the next unit?), and cost–benefit analysis drives many resource decisions, marginal analysis, and safe minimum standards are crucial. This is because they identify where valuation is appropriate and where some sort of wider risk/uncertainly analysis is necessary. From the literature review, we learned that currently, case studies rarely discuss marginal changes, SMS is rarely operationalized, but compensation schemes linking ecosystem service research and policy are more commonly considered.

We therefore call for future research in this field to not only understand, but also incorporate the concept of marginality and/or ecosystem transition states so that the results can more immediately be policy relevant. We also call for empirical studies of the amount of structure and function needed to produce a sustainable flow of services across a landscape, with special consideration to nonlinearities as we approach some minimum level. With this type of study we can begin to see where on the ecosystem service provision continuum (x-axis in Fig. 2) we currently stand, so that we can inform policy on which tradeoffs society can and cannot make. Finally, we call for researchers to think about the distribution of ecosystem service provision and use across a landscape and its associated human populations so that a variety of benefits capture mechanisms can be considered with due regard to local institutional and cultural contexts. This is essential for a societal move toward an optimal level of ecosystem services.

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Literature Cited

Andersen, L. A., O. Granger, E. J. Reis, D. Weinhold, and S. Wunder. 2002. The dynamics of deforestation and economic growth in the Brazilian Amazon. Cambridge University Press, Cambridge, UK.

Anielski, M., and S. Wilson. 2005. Counting Canada's natural capital: Assessing the real value of Canada's boreal ecosystems. Canadian Boreal Initiative and Pembina Institute, Ottawa, Canada.

Badola, R., and S. A. Hussain. 2003. Valuation of the Bhitarkanika mangrove ecosystem for ecological security and sustainable resource use. Study report. Wildlife Institute of India, Dehra Dun, India.

Badola, R., and S. A. Hussain. 2005. Valuing ecosystem functions: an empirical study on the storm protection function of Bhitarkanika mangrove ecosystem, India. Environmental Conservation 32:85–92.

Balmford, A., and W. Bond. 2005. Trends in the state of nature and their implications for human well-being. Ecology Letters 8:1218–1234.

Balmford, A., et al. 2002. Ecology. Economic reasons for conserving wild nature. Science 297:950–953.

Barbier, E. B., and S. Sathirathai. 2001. Valuing mangrove conservation in southern Thailand. Contemporary Economic Policy 19:109–122.

Barnes, P. 2006. Capitalism 3.0: a guide to reclaiming the commons. Berrett-Koehler, San Francisco, California, USA.

Batagoda, B., R. K. Turner, R. Tinch, and K. Brown. 2000. Towards policy relevant ecosystems services and natural capital values: rainforest non-timber products. Working paper GEC 2000-06. Centre for Social and Economic Research on the Global Environment (CSERGE), Norwich, UK.

Bateman, I., A. Munro, B. Rhodes, C. Starmer, and R. Sugden. 1997*a*. A test of the theory of reference-dependent preferences. Quarterly Journal of Economics 112:479–505.

Bateman, I., A. Munro, B. Rhodes, C. Starmer, and R. Sugden. 1997*b*. Does part–whole bias exist? An experimental investigation. Economic Journal 107:322–332.

Binns, J. A., P. M. Illgner, and E. L. Nel. 2001. Water shortage, deforestation and development: South Africa's working for water programme. Land Degradation and Development 12: 341–355.

Bockstael, N. E., A. M. Freeman, R. J. Kopp, P. R. Portney, and V. K. Smith. 2000. On measuring economic values for nature. Environmental Science and Technology 34: 1384–1389.

Boyd, J. 2007. Nonmarket benefits of nature: What should be counted in green GDP? Ecological Economics 61:716–723.

Boyd, J., and S. Banzhaf. 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecological Economics 63:616–626.

Carpenter, S. R., R. DeFries, T. Dietz, H. A. Mooney, S. Polasky, W. V. Reid, and R. J. Scholes. 2006. Millennium ecosystem assessment: research needs. Science 314:257–258.

Carpenter, S. R., D. Ludwig, and W. A. Brock. 1999. Management of eutrophication for lakes subject to potentially irreversible change. Ecological Applications 9:751–771.

Carson, R. T., L. Wilks, and D. Imber. 1994. Valuing the preservation of Australia's Kakadu Conservation Zone. Oxford Economic Papers, New Series, Special Issue on Environmental Economics 46:727–749.

Cesar, H. S. J., M. C. Öhman, P. Espeut, and M. Honkanen. 2000. An economic valuation of Portland Bight, Jamaica: an integrated terrestrial and marine protected area. Working paper. Institute for Environmental Studies, Free University, Amsterdam, The Netherlands.

Costanza, R. 2006. Nature: ecosystems without commodifying them. Nature 443:749–749.

Costanza, R., et al. 1997. The value of the world's ecosystem services and natural capital. Nature 387:253–260.

Costanza, R., W. J. Mitsch, and J. W. Day, Jr. 2006. Creating a sustainable and desirable New Orleans. Ecological Engineering 26:317–320.

Dahdouh-Guebas, F., L. P. Jayatissa, D. Di Nitto, J. O. Bosire, D. Lo Seen, and N. Koedam. 2005. How effective were mangroves as a defense against the recent tsunami? Current Biology 15:443–447.

Daily, G. C. 1997. Nature's services: societal dependence on natural ecosystems. Island Press, Washington, D.C., USA.

Defries, R., A. Hansen, B. L. Turner, R. Reid, and J. G. Liu. 2007. Land use change around protected areas: management to balance human needs and ecological function. Ecological Applications 17:1031–1038.

De Groot, R. S., M. A. Wilson, and R. M. J. Boumans. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics 41:393–408.

Dewees, C. M. 1998. Effects of individual quota systems on New Zealand and British Columbia fisheries. Ecological Applications 8:S133–S138.

Dietz, T., E. Ostrom, and P. C. Stern. 2003. The struggle to govern the commons. Science 302:1907–1912.

Dobson, A., et al. 2006. Habitat loss, trophic collapse, and the decline of ecosystem services. Ecology 87:1915–1924.

Emerton, L. 2005. Values and rewards: counting and capturing ecosystem water services for sustainable development. IUCN Water, Nature and Economics Technical paper number 1. World Conservation Union (IUCN), Ecosystems and Livelihoods Group Asia. IUCN, Gland, Switzerland.

Emerton, L., and E. Bos. 2004. Value. Counting ecosystems as an economic part of water infrastructure. World Conservation Union (IUCN), Gland, Switzerland, and Cambridge, UK.

Estes, J. A., M. T. Tinker, T. M. Williams, and D. F. Doak. 1998. Killer whale predation on sea otters linking oceanic and nearshore ecosystems. Science 282:473–476.

Ferraro, P. J. 2008. Asymmetric information and contract design for payments for environmental services. Ecological Economics 65:810–821.

Fisher, B., and R. K. Turner. 2008. Ecosystem services: classification for valuation. Biological Conservation 141: 1167–1169.

Gammage, S. 1997. Estimating the returns to mangrove conversion: sustainable management or short term gain? Environmental Economics Programme Discussion Paper. DP 97-02. June 1997. International Institute for Environment and Development, London, UK.

Garrod, G. D., and K. G. Willis. 1992. Valuing goods' characteristics: an application of the hedonic price method to environmental attributes. Journal of Environmental Management 34:59–61.

Gowdy, J., and J. Erickson. 2005. Ecological economics at a crossroads. Ecological Economics 53:17–20.

Grieg-Gran, M., I. Porras, and S. Wunder. 2005. How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. World Development 33:1511–1527.

Guo, Z. W., X. M. Xiao, Y. L. Gan, and Y. J. Zheng. 2001. Ecosystem functions, services and their values: a case study in Xingshan County of China. Ecological Economics 38: 141–154.

Guo, Z. W., X. M. Xiao, and D. M. Li. 2000. An assessment of ecosystem services: water flow regulation and hydroelectric power production. Ecological Applications 10:925–936.

Hecht, J. 2006. Valuing the resources of Mulanje Mountain current and projected use under alternate management scenarios. Occasional paper number 14. Community Partnerships for Sustainable Resource Management (COMPASS II). USAID, Washington, D.C., USA.

Heilbroner, R. 1968. The economic problem. Prentice Hall International, Englewood Cliffs, New Jersey, USA.

Hoen, H. F., and G. Winther. 1993. Multiple-use forestry and preservation of coniferous forests in Norway: a study of attitudes and willingness-to-pay. Scandinavian Journal of Forest Research 8:266–280.

Hoffren, J. 1997. Finnish forest resource accounting and ecological sustainability. Research Report 224. Statistics Finland, Helsinki, Finland.

Holgen, P., I. Mattsson, and C.-Z. Li. 2000. Recreation values of boreal forest stand types and landscapes resulting from different silvicultural systems: an economic analysis. Journal of Environmental Management 60:173.

Hooper, D. U., et al. 2005. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. Ecological Monographs 75:3–35.

Hughes, T. P. 1994. Catastrophes, phase-shifts, and large-scale degradation of a Caribbean coral-reef. Science 265:1547–1551.

Kahneman, D., I. Ritov, K. E. Jacowitz, and P. Grant. 1993. Stated willingness-to-pay for public goods: a psychological perspective. Psychological Science 4:310–315.

Kallesoe, M., and U. Iftikhar. 2005. Conceptual framework for economic valuation of environmental services in Southeast Asia for the RUPES Program. RUPES online report. 〈http://www.worldagroforestry.org/sea/networks/rupes/download/Annual_Reports/Appendices2005/Appendix6_Study_Mikkel_Usman.pdf〉

Kremen, C., O. Niles, M. G. Dalton, G. C. Daily, P. R. Ehrlich, J. P. Fay, D. Grewal, and R. P. Guillery. 2000. Economic incentives for rain forest conservation across scales. Science 288:1828–1832.

Kreuter, U. P., H. G. Harris, M. D. Matlock, and R. E. Lacey. 2001. Change in ecosystem service values in the San Antonio area, Texas. Ecological Economics 39:333–346.

Kumari, K. 1994. Sustainable forest management in peninsular Malaysia: towards a total economic valuation approach. Thesis. University of East Anglia, Norwich, UK.

Kunreuther, H., and M. Pauly. 2006. Rules rather than discretion: lessons from Hurricane Katrina. NBER working paper number 12503. National Bureau of Economic Research, Cambridge, Massachusetts, USA.

Le Maitre, D. C., B. W. Van Wilgen, C. M. Gelderblom, C. Bailey, R. A. Chapman, and J. A. Nel. 2002. Invasive alien trees and water resources in South Africa: case studies of the costs and benefits of management. Forest Ecology and Management 160:143–159.

Millennium Ecosystem Assessment. 2005. Millennium ecosystem assessment. Island Press, Washington, D.C., USA.

Milon, J. W., and D. Scrogin. 2006. Latent preferences and valuation of wetland ecosystem restoration. Ecological Economics 56:162–175.

Milton, S. J., W. R. J. Dean, and D. M. Richardson. 2003. Economic incentives for restoring natural capital in southern African rangelands. Frontiers in Ecology and the Environment 1:247–254.

Norton-Griffiths, M., and C. Southey. 1995. The opportunity costs of biodiversity conservation in Kenya. Ecological Economics 12:125–139.

OECD. 2001. Environmentally related taxes in OECD countries: issues and strategies. OECD, Paris, France.

Ostrom, E., J. Burger, C. B. Field, R. B. Norgaard, and D. Policansky. 1999. Sustainability—revisiting the commons: local lessons, global challenges. Science 284:278–282.

Pagiola, S., A. Arcenas, and G. Platais. 2005. Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. World Development 33:237–253.

Pearce, D. 1998. Auditing the Earth. Environment 40:23–28.

Pearce, D. 2007. Do we really care about biodiversity? Environmental and Resource Economics 37:313–333.

Pretty, J. 2003. Social capital and the collective management of resources. Science 302:1912–1914.

Priess, J. A., M. Mimler, A. M. Klein, S. Schwarze, T. Tscharntke, and I. Steffan-Dewenter. 2007. Linking deforestation scenarios to pollination services and economic returns in coffee agroforestry systems. Ecological Applications 17: 407–417.

Proctor, W., E. Comerford, S. Hattfield Dodds, M. Stauffacher, and S. Wilkinson. 2007. Motivating change in the catchment: a guide to revegetation design and incentives for catchment management bodies. CSIRO, Collingwood, Australia.

Proctor, W., S. Cork, J. Langridge, A. Langston, N. Abel, M. Howden, M. Anderies, R. Parry, and D. Shelton. 2002. Assessing ecosystem services in Australia. Sustainable Ecosystems, Commonwealth Scientific and Industrial Research Organisation (CSIRO). Seventh Biennial Conference of the International Society for Ecological Economics. Sousse, Tunisia, 6–9 March 2002. 〈http://www.ecosystemservicesproject.org/html/publications/docs/ISEE_02.pdf〉

Ricketts, T. H., G. C. Daily, P. R. Ehrlich, and C. D. Michener. 2004. Economic value of tropical forest to coffee production. Proceedings of the National Academy of Sciences USA 101: 12579–12582.

Rosa, H., S. Kandel, and L. Dimas. 2004. Compensation for environmental services and rural communities: lessons from the Americas. International Forestry Review 6:187–194.

Russell, C., and P. Powell. 1999. Practical considerations and comparison of instruments of environmental policy. Pages

307–328 *in* J. van den Bergh, editor. Handbook of Environmental and Resources Economics. Edward Elgar Publishing, Cheltenham, UK.

Sachs, J. D., and W. V. Reid. 2006. Environment: investments toward sustainable development. Science 312:1002–1002.

Salzman, J., and J. Ruhl. 2006. No net loss: instrument choice in wetlands protection. Pages 323–351 *in* J. Freeman and C. Kolstad, editors. Moving to markets in environmental regulation: twenty years of experience. Oxford University Press, Oxford, UK.

Sathirathai, S. 1998. Economic valuation of mangroves and the roles of local communities in the conservation of natural resources: case study of Surat Thani, South of Thailand. Economy and Environment Program for Southeast Asia, Singapore. 〈http://www.idrc.ca/uploads/user-S/10536137110ACF9E.pdf〉

Scheffer, M., S. Carpenter, J. A. Foley, C. Folke, and B. Walker. 2001. Catastrophic shifts in ecosystems. Nature 413: 591–596.

Scheffer, M., S. H. Hosper, M. L. Meijer, B. Moss, and E. Jeppesen. 1993. Alternative equilibria in shallow lakes. Trends in Ecology and Evolution 8:275–279.

Schuijt, K. 2003. Valuation of water: the process of economic valuation of ecosystems in water management. Dissertation. Erasmus University, Rotterdam, The Netherlands.

Schuyt, K., and L. M. Brander. 2004. Economic values of global wetlands. WWF International, Gland, Switzerland.

Seidl, A., and A. S. Moraes. 2000. Global valuation of ecosystem services: application to the Pantanal da Nhecolandia, Brazil. Ecological Economics 33:1–6.

Seroa da Motta, R. 2005. Custos e beneficios do desmatamento na Amazônia. Ciencia e Ambiente 32:73–84.

Seyam, I. M., A. Y. Hoekstra, G. S. Ngabirano, and H. H. G. Savenije. 2001. The value of freshwater wetlands in the Zambezi basin. Value of Water Research Report Series Number 7. International Institute for Infrastructural, Hydraulic and Environmental Engineering, Delft, The Netherlands.

Tilman, D., and J. A. Downing. 1994. Biodiversity and stability in grasslands. Nature 367:363–365.

Tilman, D., P. B. Reich, and J. M. H. Knops. 2006. Biodiversity and ecosystem stability in a decade-long grassland experiment. Nature 441:629–632.

Torras, M. 2000. The total economic value of Amazonia deforestation, 1978–1993. Ecological Economics 33:283–297.

Turner, R. K., W. N. Adger, and R. Brouwer. 1998. Ecosystem services value, research needs, and policy relevance: a commentary. Ecological Economics 25:61–65.

Turner, R. K., J. Paavola, P. Cooper, S. Farber, V. Jessamy, and S. Georgiou. 2003. Valuing nature: lessons learned and future research directions. Ecological Economics 46:493–510.

Turner, R. K., J. C. J. M. van den Bergh, T. Soderqvist, A. Barendregt, J. van der Straaten, E. Maltby, and E. C. van Ierland. 2000. Ecological-economic analysis of wetlands: scientific integration for management and policy. Ecological Economics 35:7–23.

Van Beukering, P. J. H., S. J. Herman Cesar, and M. A. Janssen. 2003. Economic valuation of the Leuser National Park on Sumatra, Indonesia. Ecological Economics 44: 43–62.

Van Noordwijk, M. 2005. RUPES typology of environmental service worthy of reward. World Agroforestry Centre (ICRAF), Bogor, Indonesia.

Van Vuuren, W., and P. Roy. 1993. Private and social returns from wetland preservation versus those from wetland conversion to agriculture. Ecological Economics 8:289.

Van Wilgen, B. W., M. P. de Wit, H. J. Anderson, D. C. Le Maitre, I. M. Kotze, S. Ndala, B. Brown, and M. B. Rapholo. 2004. Costs and benefits of biological control of invasive alien plants: case studies from South Africa. South African Journal of Science 100:113–122.

Van Wilgen, B. W., D. C. Le Maitre, and R. M. Cowling. 1998. Ecosystem services, efficiency, sustainability and equity: South Africa's Working for Water Programme. Trends in Ecology and Evolution 13:378–378.

Vidanage, S., S. Perera, and M. Kallesoe. 2005. The value of traditional water schemes: small tanks in the Kala Oya Basin, Sri Lanka. IUCN Water, Nature and Economics Technical Paper Number 6. IUCN, The World Conservation Union, Ecosystems and Livelihoods Group Asia. IUCN, Gland, Switzerland.

Wallace, K. J. 2007. Classification of ecosystem services: problems and solutions. Biological Conservation 139:235–246.

Wetten, J., J. Joordens, M. van Dorp, and L. Bijvoet. 1999. De Schaduwkant van Waddengas. AIDEnvironment, Amsterdam, The Netherlands.

White, A., and H. P. Vogt. 2000. Philippine coral reefs under threat: lessons learned after 25 years of community-based reef conservation. Marine Pollution Bulletin 40:537–550.

White, A. T., H. P. Vogt, and T. Arin. 2000. Philippine coral reefs under threat: the economic losses caused by reef destruction. Marine Pollution Bulletin 40:598–605.

Whitten, S., J. Bennett, W. Moss, M. Handley, and W. Phillips. 2002. Incentive measures for conserving freshwater ecosystems. Review and recommendations for Australian policy makers. Environment Australia, Canberra, Australia.

Whitten, S. M., A. Coggan, A. Reeson, and R. Gorddard. 2007. Putting theory into practice: market failure and market based instruments (MBIs). CSIRO Socio-Economics and the Environment Working Paper Number 2007-02. CSIRO, Canberra, Australia. 〈http://www.csiro.au/files/files/pf3h.pdf〉

World Bank. 2004. Ensuring the future. World Bank, Washington, D.C., USA.

Yaron, G. 1999. Forest, plantation crops or small-scale agriculture? An economic analysis of alternative land use options in the Mount Cameroon area. CSERGE Working Paper GEC 99-16. GY Associates, Herts, UK.

Yaron, G. 2001. Forest plantation crops or small-scale agriculture? An economic analysis of alternative land use options in the Mount Cameroon area. Journal of Environmental Planning and Management 44:85.

Zbinden, S., and D. R. Lee. 2005. Paying for environmental services: an analysis of participation in Costa Rica's PSA program. World Development 33:255–272.

Zhao, B., U. Kreuter, B. Lia, Z. Ma, J. Chena, and N. Nakagoshi. 2004. An ecosystem service value assessment of land-use change on Chongming Island, China. Land Use Policy 21:139–148.

APPENDIX

Questionnaire for researchers involved in ecosystem services research (*Ecological Archives* A018-074-A1).