2 Valuation Methods and Approaches for Assessing Natural Resource Management Impacts

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Introduction

Impact assessment in natural resource management (NRM) is nascent but developing rapidly in response to felt needs. Determining how to value changes in NRM is a major challenge. The value of some NRM investments can be estimated via the value of increases in yield of marketable products or the value of savings in production costs, but many NRM investments generate benefits that are hard to measure because they are not bought and sold in markets. Environmental and resource service flows that offer indirect use and non-use benefits to society certainly have value, but their measurement is a challenge. This chapter focuses on valuation methods and associated issues for measuring the social benefits that result from NRM investments.

Several thorny issues are associated with valuation of the productivity and environmental impacts of NRM investments. These include incomplete understanding of ecosystem functions and difficulties in predicting the effect of interventions on major ecosystem functions and services; lack of measurable performance indicators when effects are relatively well understood; and problems in relating changes in the flow of ecosystem services to human welfare. The non-tangibility of the benefits, time lags, and spatial (scale) effects further complicate the measurement of social, economic, and environmental impacts from NRM interventions. Farmer investments in NRM often provide non-excludable and non-consumptive public goods to the local community and beyond. For example, vegetative barriers and trees planted on the upper reaches of a watershed by a private land-user provide watershed protection, biodiversity conservation and carbon sequestration services to the local community, some of the benefits of which may even extend beyond the micro-watershed to regional and global levels. Although precise estimation of the full economic value of such investments is costly and difficult, the application of appropriate valuation methods may provide useful estimates for the direct and indirect, marketed and non-marketed ecosystem services generated by NRM investments.

Despite the extensive work on environmental valuation and benefit–cost analysis, there is a dearth of literature on methods for valuation of ecosystem services from NRM technology adoption and a serious lack of empirical examples in the developing countries that estimate the social impacts of NRM research and development efforts. This chapter provides an overview of the valuation methods and methodological approaches used to evaluate the economic and environmental impacts of NRM interventions. How NRM investments affect the flow of ecosystem services, the issues involved in translating changes in service flows to welfare gains, and some promising approaches for valuation of welfare changes are discussed. The suitability of the methods described depends on such specific circumstances as: resource types and interventions, anticipated economic and environmental effects, and interaction of biophysical changes with socio-economic conditions. The second part of the chapter summarises the multiple ecosystem services associated with NRM. In the third part, the core issues involved in the valuation of ecosystem services are discussed. This is followed by a presentation of the theoretical foundations and overview of valuation methods along with some examples on applications in the area of natural resources. The next part summarises how economic and environmental impacts can be integrated to provide an assessment of the social net benefits from NRM interventions, and the conclusion highlights the major issues and most promising valuation methods.

Agro-ecosystem Services and Functions

Agro-ecosystems are communities of plants and animals interacting with their physical and chemical environments that have been modified by people to produce food, fibre, fuel and other products for human consumption and processing (Altieri, 2002). Watersheds and agro-ecosystems offer a number of ecosystem services of value to society (for simplicity ecosystem goods and services are referred to as ecosystem services). Ecosystem services consist of flows of materials, energy, and information from the natural capital of ecosystems that provide direct and indirect human welfare benefits. In many cases, such services are public goods that cannot be privatised at low cost (high costs of exclusion) and whose consumption by one consumer does not reduce the amount available for others (non-rival). Hence, self-interested private individuals may lack the economic incentive to provide such services in socially optimal quantities.

Ecosystem services that embody public goods include: biodiversity conservation, flood and erosion control, carbon sequestration, nutrient recycling, and water retention and storage (Bingham *et al.*, 1995). In other cases, the ecosystem service may involve high costs to privatise and may also be congestible (consumption by one reduces consumption by others). Costs of exclusion are often high for ecosystem services that arise from such common property resources as groundwater, community pastures and woodlots.

Technological interventions for NRM may have diverse effects on an ecosystem at various levels. The first step towards evaluating the economic and environmental impacts of NRM interventions requires an understanding of how such investments change ecosystem functions. Each ecosystem function can be conceived as a subset of complex ecological processes that provide specific goods and services that directly or indirectly satisfy human needs. The diverse ecosystem services generated through ecosystem functions therefore provide various economic, environmental, and socio-cultural benefits and values to people. De Groot *et al.* (2002) developed a typology for the classification of ecosystem functions and services. Although their general typology is meant for natural ecosystems, it can be adapted for use in agro-ecosystems to understand the likely impacts of NRM interventions. Changes in the scale and intensity of managing natural resources in agriculture will change the flow of agro-ecosystem services, which will in turn influence the quantity and/or quality of goods and services produced. Depending on the type of NRM technology used, the typology developed by de Groot *et al.* (2002) suggests that valuable ecosystem services may be generated through any of the following ecosystem functions:

- Production
- Regulatory services
- Habitat
- Socio-cultural (information) services.

Table 2.1 summarises the major ecosystem functions and services together with selected indicators of change due to NRM interventions in the context of agro-ecosystems.

Production of food and raw materials is a major ecological function of agro-ecosystems that includes food, feed, fuel, raw materials and medicines. This function is transmitted through the conversion of solar energy into edible plants by autotrophs for human and animal consumption. Farm animals convert fodder and herbaceous material into economic goods and services for use by humans. Natural resource investments may also influence the ability of the agro-ecosystems to produce products for ornamental and medicinal use, and the conservation of biological diversity. As shown later in this chapter, when data are available, simulation models and statistical methods can be used to establish the relationships between NRM investments and changes in the flow of goods and services (see also Chapter 5, this volume). These effects are typically realised on-site and create economic incentives for resource users to adopt new technologies. When the productivity effects are limited, farmers' direct economic benefits and the incentives for adoption and adaptation of NRM technologies will be low.

The regulation function relates to the role of agro-ecosystems in the maintenance of essential ecological processes and life-support systems. Such ecosystem services may be transmitted through changes in land cover that

Table 2.1. Ecosystem functions and potential indicators of change in agro-ecosystem services associated with natural resource management (NRM) interventions.

Ecosystem services	Ecosystem functions (processes and components)	Indicators for changes in agro-ecosystem services
A. *Production functions* – Provision of natural resources as factor inputs in production activities		
Food	Conversion of solar energy into edible plants and animals for humans	Changes in land productivity (crop and livestock)
Raw materials	Conversion of solar energy into biomass for feed, construction and other uses	Changes in fodder, fuelwood, timber, etc., output
Genetic resources	Conservation of genetic materials	Changes in agro-biodiversity
Medicinal resources	Bio-chemical substances, medicinal uses	Changes in availability of medicinal plants or changes in use benefits from medicinal plants
Ornamental resources	Ornamental use	Changes in economic benefits from ornamental plants and animals
B. *Regulation functions* – Maintenance of essential ecological processes and life support systems		
Climate regulation	Influence of land cover and carbon sequestration on climate	Changes in land cover and carbon sequestration
Water regulation	Role of land cover in regulating runoff and river discharge	Changes in runoff and sediment loss
Water supply	Filtering, retention and storage of fresh water	Changes in water availability and quality
Soil retention	Role of vegetation root matrix and soil biota in soil retention	Changes in rates of soil erosion and sediment loss
Soil formation	Weathering of rock, accumulation of organic matter	Changes in soil depth
Nutrient regulation	Role of biota in storage and recycling of nutrients	Changes in nutrient balances, soil fertility and organic matter
Pollination	Role of biota in movement of floral gametes	Changes in pollinating insects
C. *Habitat functions* – Providing habitat for wild plant and animal species		
Refugium function	Suitable living space for certain desirable species	Changes in the stock of wildlife, soil flora and fauna
Nursery function	Suitable reproduction habitat for certain desirable species	Changes in rates of reproduction
D. *Sociocultural functions* – Providing opportunities for cognitive development		
Aesthetic information	Attractive landscape features	Changes in landscape and scenery
Recreation	Variety in landscapes with (potential) recreational uses	Changes in recreational benefits (agro-ecotourism, outdoor sports, etc.)

Table 2.1 Continued.

Ecosystem services	Ecosystem functions (processes and components)	Indicators for changes in agro-ecosystem services
Cultural and artistic	Features with cultural and artistic value	Changes in cultural and artistic use (e.g. motivation for books, films, advertising, etc.)
Spiritual and historic	Agro-ecosystem types with spiritual and historic value	Changes in use for religious and historical use (e.g. heritage, spiritual symbol)
Science and education	Agro-ecosystem types with scientific and educational value	Recognition for scientific or educational purposes

Source: Updated based on Costanza *et al.* (1998) and de Groot *et al.* (2002)

influence and regulate: climate change (e.g. through carbon sequestration), water flows (runoff and river discharges), and protect soils from erosion, water supply through filtering, retention and storage of fresh water (e.g. wetlands, check dams, etc.); soil formation through decomposition of organic matter and weathering of rocks; nutrient regulation through storage and recycling of nutrients; biological control of pests; pollination through the role of fauna in the movement of floral gametes. A number of useful ecological (biophysical) indicators can be developed to monitor the NRM technology impacts on these kinds of ecosystem services (Chapters 3–5, this volume).

The habitat function indicates the useful services provided by agro-ecosystems in the provision of habitat (suitable living space) and nursery (reproductive space) services for uncultivated and cultivated plant and animal species. People derive non-material well-being from the flow of these services. It is difficult to develop simple indicators to monitor NRM impacts on these ecosystem services. The number of species in a given habitat (species richness) and the species diversity can be measured using different biological indices (Chapter 5, this volume).

Natural resource investments also provide such socio-cultural services as aesthetic information (e.g. attractive landscape), recreational services (e.g. ecotourism), and scientific and spiritual services. These are mainly public goods that provide useful services to society or the community as a whole.

When markets exist, changes in some of these agro-ecosystem services resulting from NRM investments can be quantified and valued in monetary terms. For public goods (e.g. changes in biodiversity, water and air quality) markets are either missing or often imperfect. The quantification of benefits and valuation therefore presents special difficulties. Before valuation methods are considered, the major issues and challenges surrounding valuation of ecosystem services are briefly described.

Issues in Valuation of Agro-ecosystem Services

There are two fundamental steps in the valuation of impacts from NRM investments: firstly, understanding and predicting the changes in the flow of ecosystem services attributable to the technological or policy intervention, and secondly, devising acceptable methods for valuing these changes. The first helps identify and quantify *what is to be valued* while the second one helps develop *suitable methods* for valuing the changes. In this section the issues involved in uncovering what is to be valued and how it is to be valued are discussed. As described above, ecosystems are very dynamic and complex, and human knowledge about them is very incomplete. This limits the ability to understand and quantify the changes in the ecosystems service flows associated with human interventions. The effects of NRM interventions can be physical, chemical or biological, and may take different forms over temporal and spatial scales. However, understanding and predicting the impacts of interventions on ecosystem functions is the prerequisite to economic valuation. Good valuation depends on sound agroecological information on the effects of policy and management interventions. Functional inter-linkages and feedback effects make it difficult to determine the causal relationship between human interventions and changes in ecosystem functions and processes (Bingham *et al.,* 1995). Any sensible effort to assess the impacts of NRM interventions requires a reasonable understanding of how and to what extent the different ecosystem service flows will change as a result of human interventions. This implies an interdisciplinary effort involving agroecologists, agronomists, biophysical scientists and economists. Bingham *et al.* (1995) argue that if there is no agreement on the effect of changes on the flow of ecosystem services, there can be no agreement on valuation of the impacts.

If changes can be predicted or quantified reasonably, the next question will be – which of these changes can be valued in monetary terms? The choice of which changes to value is an important challenge for the economist. Before values can be placed on the impacts, it is necessary to know what is to be quantified and how it can be measured. Indicators of changes in the service flows (immediate impacts) are critical for valuation. Indicators can be developed through experimentation and appropriate monitoring of changes over a sufficient period of time, or through the application of exploratory and predictive simulation models. The latter approach is most useful when changes are slow to evolve or when complexity of anticipated interactions makes actual experimentation very difficult. NRM combines both features and involves multiple interventions that make it problematic to isolate partial effects. In the absence of good counterfactuals, experimental data might not provide useful insights about the anticipated impacts. Oriade and Dillon (1997) provide a good review of applied simulation models used in agricultural systems.

There are various efforts to develop measurable indicators for changes in the flow of agro-ecosystem services (Dumanski and Pieri, 2000; Arshad and Martin, 2002). The next three chapters in this volume provide a detailed account of the measurable indicators for soil, water, and other agro-ecosystem services. The challenge is to develop indicators that could be easily monitored on-farm as part of the project cycle. Good indicators are those that capture major elements in a complex interactive system while simultaneously showing how the value obtained relates to some ideal or desired level. Smyth and Dumanski (1993) reported the use of participatory rural appraisal techniques for developing land-quality indicators for sustainable land management for sloping lands in Indonesia, Thailand, and Vietnam. The framework for evaluating sustainable land management was used to develop threshold levels for the sustainability of land-management indicators. Campbell *et al.* (2001) proposed linking indicators to changes in five livelihood assets (natural, physical, financial, social, and human capital). They suggest a collective measure for each of the capital assets that could be used to develop an aggregate index. Although the aggregate index gives little guidance as to what needs to be included under each asset category, it might serve as an organising framework to develop a few indicators under each asset category for those projects expected to have wide-ranging impacts.

Once the relevant changes are identified and quantified through appropriate indicators, the next question becomes – how to value these changes? There are many vexing issues on how ecosystem service flows are valued. Even if effects can be predicted and monetary valuation is possible, many still argue if money values could adequately inform decision-making, especially when irreversible changes, trade-offs, and distributional effects are involved. The term 'value' may also have different concepts and meanings for different disciplines (Bingham *et al.*, 1995; Bockstael *et al.*, 2000; Farber *et al.*, 2002). In common usage it means 'importance' or 'desirability'. An economic value measures the change in well-being associated with the change in the quantity or quality of the service flow. Changes in resource and environmental service flows can affect human welfare in complex ways and through marketed or non-marketed activities. The most common approach to translating these changes into monetary units is to express the welfare change as the amount a person would pay or be paid (in compensation) to be as well-off with the change as without it. The amount that individuals are willing to pay, or might accept as compensation, is not an absolute value; it will vary across individuals depending on property rights, perceived welfare gains/losses, the context, and the availability of substitutes.

There are two key questions that need to be answered in the process of economic valuation of ecosystem services. The first is how to construct a measure of how much better or worse-off an individual is because of the change in the quantity or quality of the service flow. The second is how to add up the individual welfare changes (gains and losses) to assess the value of this change for society as a whole. Recent advances in economic theory provide answers to these two fundamental questions and offer useful methods for the valuation of many ecosystem service flows regardless of the functioning of markets.

Valuation Techniques

Unlike agricultural products harvested in fixed time periods, environmental and ecosystem services associated with NRM interventions flow in real time on a continuous basis. Understanding the changes in ecosystem service flows, measuring and monitoring outcomes across time and space is very important for quantifying environmental impacts. The basic principles that guide valuation exercises and the different valuation methods relevant for NRM, including their strengths and weaknesses are discussed.

Theoretical foundation

The economic approach to the valuation of resources is based on the contribution of the resource to human welfare. Whether the good or service is marketed or non-marketed, its unit economic value is determined by the welfare contributions that it makes to humans. Changes in welfare are measured in terms of each individual's personal assessment of changes in well-being (Bockstael *et al.*, 2000). For traded commodities, the demand curve depicts the marginal willingness to pay (WTP) (or marginal benefit) for the good or service. The height of the demand curve at each point of the quantity demanded shows the maximum WTP for the commodity. The household will consume all units of the commodity where the marginal WTP exceeds the market price. The consumer enjoys a consumer surplus for all points where the marginal WTP is higher than the market price. The welfare change associated with a change in the price of a marketed commodity is often measured using the change in consumer surplus, derived from the Marshallian demand curve with a constant level of income. For a non-marketed ecosystem service, the maximum WTP for an improvement in quantity or quality is the area between the initial and new levels of the resource under the demand (marginal benefit) curve. Value estimation then involves determining directly or indirectly the shapes of these marginal WTP curves for the ecosystem services (Freeman, 1993).

Environmental and resource service flows typically exhibit public-good characteristics of high costs of exclusion and non-rivalry. This makes it very difficult for markets to value these ecosystem goods and services accurately, and leads to a market failure and non-tradability. In order to illustrate how the values for such non-marketed resources could be estimated, let us assume that a given household maximises its welfare (U) from consumption of a vector of marketed goods (c), ecosystem goods and services (q) and has a fixed budget y, such that:

$$Max\ U = U(c, q) + \lambda(y - p'c) \tag{1}$$

The standard utility-maximising solution to this problem will give the Marshallian demand function for the tradable commodity:

$$c^* = c(p, q, y) \tag{2}$$

which is a function of a vector of market prices (p), the disposable income (y) and the ecosystem services (q) considered to be a public good. If this is substituted into the utility function, the indirect utility function could be derived:

$$v(p, q, y) = U(c(p, q, y), q) \tag{3}$$

The marginal effect of the change in the level of the public good q_i on household welfare can be derived as:

$$\frac{\partial v(p, q, y)}{\partial q} = \frac{\partial U\big(c(p, q, y), q\big)}{\partial q} \tag{4}$$

This is equal to the marginal valuation of the environmental good in question. It is a measure of the marginal welfare benefit (demand curve) for the public good q (Johansson, 1987). For a given change in q from q^0 to q^1, the welfare effect on household h can be estimated as:

$$\Delta U^h = v^h(p, q^1, y) - v^h(p, q^0, y) = \int_{q^0}^{q^1} \left[\frac{\partial v^h(p, q, y)}{\partial q} \right] \tag{5}$$

The total welfare effect (WTP) summed over all the affected households (h) can be calculated as:

$$\sum_h \Delta U^h = \sum_h \int_{q^0}^{q^1} \left[\frac{\partial v^h(p, q, y)}{\partial q} \right] \tag{6}$$

In general utility functions are unobserved and it would be useful to convert Equation 6 into a monetary measure of welfare change. This is done by assuming constant marginal utility of income (λ_h) for each household, and dividing the marginal valuations in Equation 6 by (λ_h). This is the same as vertical summation of the demand curves and will provide the aggregate uncompensated WTP for all the affected households given the change in q from q^0 to q^1. The compensating surplus (CS) and equivalent surplus (ES) measures (analogous to the CV and EV measures for price changes) can also be directly derived from the indirect utility function. For an improvement in q from q^0 to q^1 the CS and ES measures can be computed as:

$$v(p, q^1, y - CS) = v(p, q^0, y) \tag{7.1}$$

$$v(p, q^1, y) = v(p, q^0, y + ES) \tag{7.2}$$

In terms of empirical applications, the expenditure function that can be derived from the indirect utility function using the envelope theorem by solving for the expenditure level (y) that will provide a given level of utility, can be very useful in directly estimating the monetary measure of the welfare change associated with provision of the public good (q). The expenditure function for household h is given by $e^h(p, q, \bar{u})$. The aggregate welfare change measure for a change in q from q^0 to q^1 for CS can be given as:

$$CS = \Sigma_h \Big(e^h(p, q^0, u^0) - e^h(p, q^1, u^0)\Big) = \Sigma_h \int_{q^0}^{q^1} \left[\frac{\partial v^h(p, q, u^0)}{dq} \right] dq \tag{8}$$

The aggregate ES measure for a change in q from q^0 to q^1 can be given as:

$$ES = \Sigma_h \left(e^h (p, q^0, u^1) - e^h (p, q^1, u^1)\right) = \Sigma_h \int_{q^0}^{q^1} \left[\frac{\partial v^h(p, q, u^1)}{dq}\right] dq. \tag{9}$$

The CS is the maximum amount of money that the individual is willing to pay to secure an increased provision of the public good q. The ES measures the minimum sum of money that must be given to individuals before the change to make them as well-off as they would have been following an increase in q. This forms the basis for valuation of non-marketed ecosystem services. When the environmental and resource flows serve as inputs in production of market goods by producers, the equivalent welfare measure of the change in productivity is the change in producer and consumer surplus (Ellis and Fisher, 1987). Freeman (1993) demonstrates the other indirect benefit estimation approaches where q enters the production function as a factor input or as an input in the household production of utility-yielding commodities. Before the various methods and approaches used for eliciting values for ecosystem services are discussed, the components of the total economic value and the effect of markets and externalities on the choice of valuation methods are briefly demonstrated.

Valuation of impacts

The valuation of changes in ecosystem services needs to take into account intended and unintended outcomes. Individuals may attach values for such changes because of the use benefits derived, or any anticipated or conceived non-use welfare benefits. Agricultural activities often impose external costs on society mainly because individual resource-use decisions occur at points that equate marginal private benefits and costs. Soil erosion and sedimentation, and use of fertiliser, pesticides and other chemicals are some examples that impose costs on other agents and ecosystems. Unintended economic effects that spill over to other agents are often called externalities. More formally, externalities are unintended effects on the production or consumption activities of an economic agent resulting from the activities of another economic agent that are not mediated through markets. Adoption of 'best practice' NRM technologies like integrated pest management (IPM) or upland watershed management reduces external costs imposed on ecosystems and on other farmers in the lower reaches of the watershed. Hence, NRM investments may provide multiple ecosystem services to different economic agents across spatial scales as illustrated in Table 2.2 (Pagiola *et al.*, 2002). The use value (UV) of a given NRM investment includes the sum of direct and indirect use benefits (marketed and non-marketed) that accrue to all beneficiaries on-site and off-site. The challenge is how these dispersed benefits could possibly be valued. This requires good knowledge about the nature of the effect, how long the effect will last, its spatial dispersion, and the affected parties. The use value of the resource to different groups of economic agents cannot exceed the perceived benefits accruing to the group. Therefore, local forest managers,

Table 2.2. Perceived (on-site and off-site) benefits from integrated watershed management investments (soil, water and vegetation).

Ecosystem goods and services	Local communities	Downstream water and land users	Distant stakeholders and global community
Supplemental irrigation			
Improved agricultural productivity			
Fuelwood, pasture and construction materials			
Reduction in flooding and siltation			
Water purification			
Carbon sequestration			
Biodiversity preservation			

for example, will not consider the water quality captured by downstream water users, and biodiversity benefits to the global community. In this case the total use value of the resource will be given as:

$$UV_i = \sum_{i=1}^{n}\sum_{j=1}^{m}\gamma_{ij} Y_i \tag{10}$$

where γ_{ij} is the distribution parameter reflecting the ecosystem service i captured at location j, and Y_i is the total use value of the ecosystem service i. Thus, the *UV* is the sum of all ecosystem services captured by all the beneficiaries across locations. Obviously, this poses practical difficulties in mapping out the benefit dispersion and in elicitation of values from different agents.

The total economic value (*TEV*) of a given resource may however include non-use values. The non-use values (*NUV*) include what are called option value (*OV*), bequest value (*BV*) and existence value (*ExV*). Figure 2.1 illustrates the components of the total economic value. *OV* is a measure of how much individuals are willing to pay for the option of preserving the asset for future personal use. *BV* is the value that individuals are willing to pay to ensure that the resource will be preserved for future generations. *ExV* is the value that individuals attach to the mere existence of a given natural resource or environmental asset unrelated either to current or optional use. Thus:

$$\begin{aligned} TEV &= UV + NUV \\ &= (\textit{Direct use value} + \textit{Indirect use value}) + (OV + ExV + BV) \end{aligned} \tag{11}$$

The nature of the externality and the structure of markets will have substantial implications on the choice of effects to be valued and the valuation methods to be used. This can be seen by relating the anticipated benefits across spatial scales and the existence of markets to value these benefits (Dixon *et al.*, 1994). As can be seen from Table 2.3, the benefits from goods and services in Group I are both tradable within the local economy and are captured on-

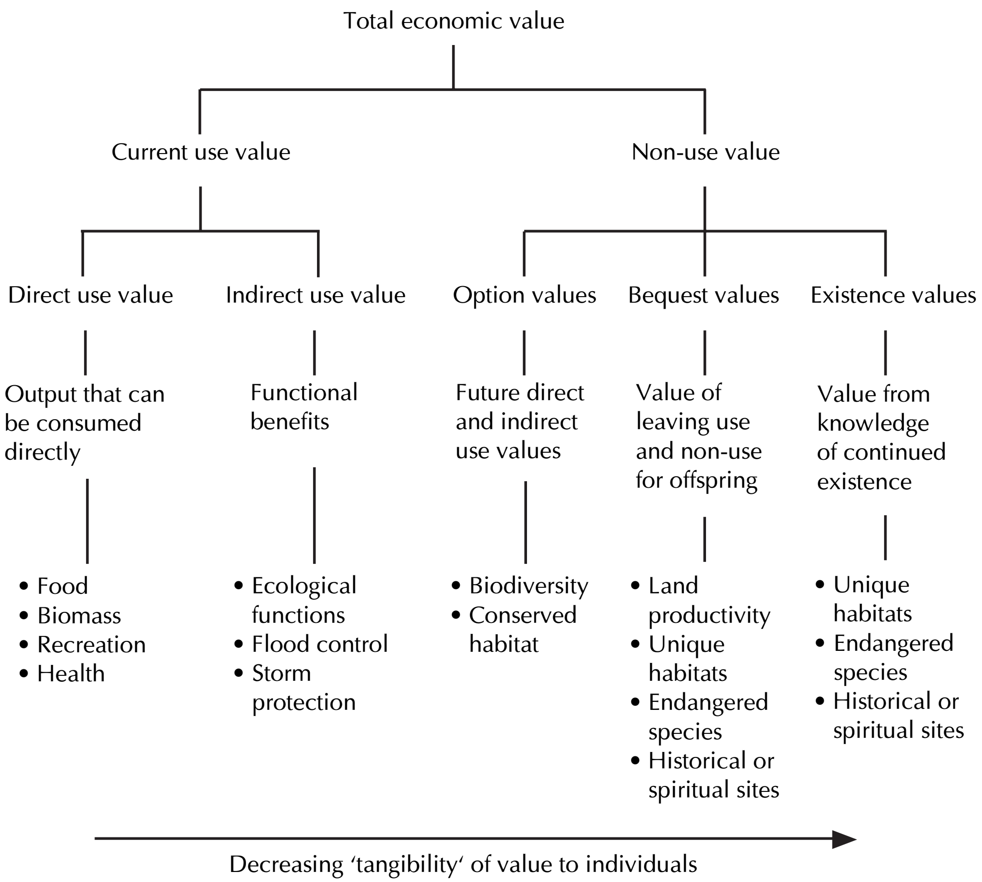

Fig. 2.1. Decomposition of total economic value of ecosystem goods and services (Munasinghe and Lutz, 1993).

site. These goods and services could be valued using market prices, with adjustments for any distortions that may exist (e.g. monopoly, externalities, and existing taxes and subsidies not reflecting external costs and benefits). For goods and services in Group II, market prices may exist, but local producers do not capture benefits, i.e. the lion's share of such benefits is 'externalised'. To the extent that these external benefits can be quantified, they could be incorporated into the social economic analyses of NRM impacts.

For those in Group III, benefits accrue within the local economy (of the household or village) but many of the goods and services are non-tradable. Missing markets mean that such benefits, however large, are seldom included in empirical impact assessments. Even the most difficult for valuation are goods and services generated from NRM investments, which are neither captured by the producers nor traded through markets (Group IV). Examples in this category include benefits of climate regulation (carbon sequestration) and biodiversity conservation resulting from upland

Table 2.3. Valuation of ecosystem goods and services from tree planting: The role of markets and externalities.

	Location of goods and services	
Tradability	On-site	Off-site
Marketed	**Group I** Benefits accrue on-site (e.g. fuelwood, fodder, timber, etc.) and are tradable – Usually included in impact assessment (IA)	**Group II** Off-site tradable benefits (e.g. higher crop yields or more hydropower resulting from reduced siltation in dams) – Sometimes included in IA
Non-marketed	**Group III** Benefits accrue on-site but are highly non-tradable (e.g. soil and water conservation, recreation, regulation of micro-climate, etc.) – Seldom included in IA	**Group IV** Off-site non-tradable benefits (e.g. Carbon sequestration, reduced flooding, biodiversity conservation) – Usually ignored in IA

Source: Modified based on Dixon *et al.* (1994) and others

tree planting. In the absence of markets, such non-tradable and external benefits need to be estimated by other methods using surrogate markets or constructed (hypothetical) markets in areas where the benefits are captured. Because of non-excludability, the level of investment by local producers to generate such services may be socially sub-optimal, necessitating many governments to intervene in their production and management.

Valuation methods

Advances in resource and environmental economics in the last few decades have provided many useful methods that can be employed to value use and non-use values of ecosystem goods and services, both marketed and non-marketed (Mitchell and Carson, 1989; Braden and Kolstad, 1991; Freeman, 1993; Bateman and Willis, 1999). The intention here is not to review the extensive literature on environmental valuation but to highlight briefly the methods that can be applied for valuation of NRM impacts in the context of developing countries. Table 2.4 presents an overview of some of these methods that can be potentially applied to value ecosystem services resulting from NRM investments. The methods can be distinguished by the type of market used, as well as the implied behaviour of the economic agent in the valuation of goods and services. Changes in productivity, replacement cost, avoided cost and opportunity cost methods use actual markets, but

Table 2.4. Valuation methods for ecosystem goods and services.

Implied behaviour	Conventional market	Surrogate market	Constructed market
Based on potential or stated behaviour	Productivity change approach (PCA) Replacement cost approach (RCA) Avoided costs Opportunity costs		Contingent valuation (CV) method Choice modelling (CM)
Actual or revealed behaviour	Provision costs Defensive expenditures Relocation costs	Hedonic pricing (HP) (land value) Wage differentials Travel cost	Experimental markets

behaviour is *assumed*, i.e. no actual outlays or market transactions have occurred. Defensive expenditures, provision costs and relocation costs also use actual markets but behaviour is *expressed* or *observed* through conventional markets. Methods like hedonic pricing and wage differentials use surrogate markets to value non-tradable goods and services indirectly through marketed goods and services that embody their values. In the extreme case, it may be possible to construct experimental markets (where behaviour is *revealed* with actual WTP or accept compensation for a change) or hypothetical markets (where behaviour is *stated* without actual transactions as in the case of the contingent valuation method). Methods that use information from conventional markets are presented below followed by those that use surrogate markets and those requiring constructed markets. The treatment gives more emphasis to methods that offer high potential for application in NRM impact valuation.

Productivity change approach (PCA)

Some agricultural resource-improving investments lead to changes in productivity and/or production costs. This means that physical changes in production or overall farm profits derived from adoption of such technologies can be established and valued using market prices.[1] This approach is quite attractive and suitable for evaluating NRM impacts, because physical changes in productivity can be observed and measured. Production functions, erosion damage functions, fertiliser response functions, crop–water responses to supplementary irrigation, and simulation models that relate changed resource conditions to productivity are good examples of PCA. A number of studies on the economic costs of land degradation, soil erosion, etc. have used this method to value the benefits from resource management investments. Magrath and Arens (1989) used detailed erosion–yield relationships to measure the on-site costs of soil erosion in Java, Indonesia. The capitalised cost of a 1% productivity decline is estimated to amount to 4% of the total value of dryland crops in Java. They also estimated off-site costs of sedimentation in reservoirs, irrigation systems and harbours, and found that these costs are about a quarter of the total erosion damage costs. Bojo (1991) used this approach to value the economic cost of soil degradation in Lesotho. However, Barbier (1998) noted these case studies probably overestimated the scarcity value of soil, because the economic value of conservation was not assessed as a net gain in profitability over the erosive (conventional) system. When there are no economically viable options for mitigation, soil degradation might not have on-site opportunity costs (i.e. on-site costs of soil degradation exist only when conservation is profitable on-farm). Shiferaw and Holden (2001) estimated various erosion–yield functions to evaluate conservation benefits and the net gain to farmers from adoption of conservation methods. Gebremedhin *et al.* (1999) also estimated returns to investments in terracing in the Ethiopian highlands, using experimental data on crop yields under different conservation methods. This is a good approach for valuing the economic cost of soil degradation or for evaluating conservation benefits.

Although production functions with resource conditions as factor inputs (along with other usual input factors) can be used to estimate the economic value of the resource, lack of such data often limits the application of this approach. One major difficulty is that it takes a long time for NRM investments to have an observable effect on the flow of ecosystem services. The first step in applying this method is to quantify the effect of changes in NRM on the quantity or quality of the resource base that affects resource productivity. For example, changes in NRM may affect rooting depth, water-holding capacity or organic matter levels in the soil. In this way, the condition of the resource stock (S) will be a function of the conservation effort and other exogenous characteristics such that:

$$S = g(K, Z) \tag{12}$$

where K is the level of NRM investment per ha, and Z is a set of exogenous factors (e.g. soil type, agroecological zone, rainfall, etc.). When the experimental data needed to estimate this statistically are lacking, simulation models may be used to estimate the effect of the change in K on the condition of the resource or the flow of ecosystem services (assuming that suitable parameters are available for the simulation models).

Moreover, the changes in productivity associated with changes in K may take a long time to be visible to farmers. Use of such other inputs as fertilisers and high-yielding varieties also often mask NRM investment benefits to farmers. When data that relate crop productivity with input use and biophysical conditions (e.g. soil depth, soil moisture, soil types) are available, econometric methods can be used to establish useful relationships such that:

$$Q = f(X, S, Z) \tag{13}$$

where Q is the productivity of land, X is a vector of inputs used, S is a vector of resource quality indicators, and Z is a vector of other exogenous factors that influence crop productivity. Controlling for variable inputs and fixed exogenous factors, the marginal effect of the anticipated change in the quality of the natural resource will be given as:

$$\partial Q(\cdot)/\partial S = f_S(X,S,Z)\Big|_{\{X=\overline{X},\, Z=Z_i,\, \forall i\}} \tag{14}$$

In some cases, the level of input use may not remain unchanged, e.g. improvement in soil fertility may prompt credit-constrained farmers to reduce the demand for fertilisers to produce the same level of output. In such cases, the productivity change associated with changes in NRM can be measured using the savings in input costs.

As shown in Equation 14, the marginal effect of NRM investments will depend on the quality of the resource stock and other exogenous factors. The total effect on productivity can be estimated by integrating over the level of change in *S* resulting from the change in *K*.

$$\Delta Q = \int_{S_0}^{S_1} \left(\frac{\partial Q(\cdot)}{\partial S} \right) dS \tag{15}$$

where S_0 is the old and S_1 the new levels of the resource condition associated with changes in *K* (NRM investments). The economic value of the change in NRM needs to be calculated as a producer surplus by including the opportunity cost of the variable inputs used in production (including the cost of *K*). If the change in output does not induce price changes, the producer surplus will be the value of the change in output minus the cost of production. The productivity changes may flow at different rates as the resource condition changes from S_0 to S_1 over a period of time. If the productivity changes are long-lasting, the present value of net productivity benefits (producer surplus) can be computed using the social rate of discount. While the approach is attractive and widely used, it has some disadvantages. These include high data requirements (when the econometric approach is used), lagged effects of NRM that hinder reliable assessment of productivity changes on-site, and difficulties in accounting for any off-site (externality) effects of the change in NRM. This last limitation is perhaps the greatest, because the PCA approach does not measure the value of non-marketed environmental goods and services.

Replacement costs approach (RCA)

Under this approach, potential expenses that may be needed to replace or restore the damaged natural resource asset are estimated using the prices of marketable products. The resulting estimate is not a measure of the benefits of avoiding the damage in the first place, since the damage cost may be higher or lower than the replacement cost. The implied expenditure to restore a given resource to a pre-damaged state or baseline condition may however be different from the costs of replacing its functions. Because of this, the RCA is mainly used in the latter context where the estimated resource values reflect the potential expenses needed to replace the services of the damaged resource through some substitutes (e.g. use of fertilisers or other fertility management practices to replace lost soil nutrients). Replacement costs can be a valid measure of economic value when the following conditions are met (Dixon *et al.*, 1994; Bockstael *et al.*, 2000):

- The magnitude of the damage is measurable and there are no secondary benefits associated with the replacement expenditure
- The substitute provides functions that are similar to the lost ecosystem service
- The substitute is the least-cost option for replacing the lost service
- Affected individuals in aggregate would, in fact, be willing to incur these costs if the natural functions were no longer available
- When the replacement costs are greater than the aggregate WTP or the social value of the productive resource destroyed, it will be economically inefficient to replace the damaged ecosystem service.

As Barbier (1998) noted, when these conditions are not fulfilled and least-cost replacement options are not known, simplistic application of the approach could lead to overestimated and misleading values. For the case of soil erosion, he noted that some of the eroded soil may be deposited on-farm and cannot be considered lost completely. Moreover, all the eroded soil might not have economic value if its marginal productivity effect is negligible. In these situations, the RCA can lead to overestimated resource values. By definition, the RCA includes only the costs of replacing damaged ecosystem services on-site, but the concept is equally applicable for valuation of any associated external effects. While the full restoration costs may include non-use values, the replacement costs reflect the use value of the resource or ecosystem service.

A number of studies have used this method. One example is the case study by Kim and Dixon (1986), which assessed the viability of alternative soil conservation techniques in upland agriculture in Korea. The difference in the estimated cost of physically replacing lost soil and nutrients (estimated based on differences in soil erosion) was taken as a measure of the potential benefits of preventing soil erosion. With the assumption that the value of retaining productive soil is higher than the replacement cost, the study found that preventive measures were more economical than physically replacing lost soil and nutrients.

Provision costs

Economic values for non-market ecosystem services that contribute to human welfare can sometimes be derived from people's decisions to use related resources or to substitute other resources where the quality of the service flow is impaired. The provision costs approach (PCA) refers to the actual expenditures that farmers or communities may incur to provide vital environmental goods and services. Unlike the mitigating expenditures, these expenses are directly targeted in the provision and production of the required good or service. While it can be considered as a variant of the RCA, the PCA does not refer to restoration of the ecosystem service, but to costs of providing the damaged service through alternative means. Some examples include farmers' expenses on drilling wells for irrigation and drinking water when water regulation services of watersheds are damaged, and the costs of alternative sources of household energy after deforestation. The strength of the method is in trying to value the resource in question using the actual cost outlays in producing the required good or service. However, the costs may also serve other purposes, and external benefits are excluded when private provision costs are considered. The method also relies on existence of markets for major inputs used in the production of the environmental good or service.

Defensive expenditures

Farmers, communities, and governments often incur actual expenditures to mitigate or prevent productivity loss or reduce degradation problems. When the extent and potential effect of resource degradation or improvement is difficult to assess, actual preventive or defensive expenditures may be used to assess a rough value of the change in the resource quality. Kim and Dixon (1986) use lowland farmers' defensive expenditures to prevent deposition of silt on rice fields to evaluate alternative soil management techniques designed to stabilise upland soils. There are several problems in the use of this method. Firstly, defensive expenditures, like all WTP, are limited by income and the value so obtained may not reflect the social scarcity value of the resource. It may at best be a lower-bound estimate. Secondly, the value tends to be quite arbitrary as actual expenditures may be targeted to meet several objectives (e.g. conservation of multiple resources).

For use in NRM impact assessment, it is important to determine the anticipated change in resource conditions attributable to the intervention, and how much farmers often spend to prevent an equivalent deterioration in the resource. If defensive expenditures on-site and off-site can be estimated, they may provide a rough indication of the value of the improvement in the resource. In some cases, relocation costs associated with environmental change can be considered part of defensive expenditures. Hence, the relocation costs approach is not discussed separately.

Hedonic pricing (HP)

The theory of hedonic prices is based on the premise that market prices reflect a bundle of observable characteristics and attributes of differentiated products (Rosen, 1974). Different attributes of the same product reflecting differences in its inherent worth will have an associated price, and consumers can easily identify what they are paying for in selecting various options. When goods and services contain non-priced environmental attributes embedded in them, consumers may also place implicit values on each of the attributes so that market prices are composed of environmental and non-environmental attributes. Therefore, when the good or service provided by NRM investments cannot be directly valued using conventional markets, behaviour revealed through surrogate markets can be used for valuation.[2] For example, the value of access to clean water and air can be estimated indirectly through the differences in market prices for houses in polluted and clean localities, after controlling for their structural and other attributes (Harrison and Rubinfeld, 1978). Wage differentials for occupations with different levels of health or environmental risk have also been used to estimate certain environmental values. The HP method is designed to control for certain non-environmental attributes so that the remaining property value differentials can be 'surrogate' values of the non-priced environmental goods and services.

To the extent that surrogate markets are competitive, the HP approach can therefore be very useful for valuing NRM impacts. For example, land values in competitive markets can be used to value differences in land quality. If prices for agricultural land reflect quality changes, the hedonic function for a given parcel with a vector of biophysical (environmental) characteristics $L = (l_1, l_2, \ldots, l_n)$ and socio-economic characteristics of the location and the buyer $Y = (y_1, y_2, \ldots, y_n)$ can be estimated econometrically as:

$$P = g\,(L, Y) \tag{16}$$

where P is the market price of a unit of land. The socio-economic characteristics include such variables as buyer characteristics, population density, distance to urban areas, distance to markets, and type of crops grown. The coefficients of this model can be used to determine the implicit price associated with land characteristic, holding all other factors constant. For example, for soil characteristic l_i (e.g. soil depth) the implicit price is the partial derivative with respect to soil depth such that:

$$\frac{\partial P}{\partial l_i} = g_l\left(L, Y\right) \tag{17}$$

If the impact of NRM investments on the biophysical conditions of the resource is known, market prices can be used to value indirectly the changes in resource attributes. One disadvantage of this method is that it requires extensive information on selling or rental prices and associated socio-economic and biophysical characteristics of the property. Even when such data are available, market prices may not be competitive or may not fully reflect such non-observable quality differentials as changes in nutrient balances or biophysical attributes of the soil. The method works quite well if markets reflect quality differentials. Even when they do, market prices may reflect only the capitalised value of future on-site productivity gains from using the land. Changes in non-productivity benefits (e.g. biodiversity, carbon sequestration) and reductions in off-site effects from NRM investments might not be reflected in market prices. In a recent study Shiferaw *et al.* (2003) found that farmers' perceived value of land parcels in semi-arid Indian villages were able to clearly reflect soil and farm characteristics that affect land productivity. Factors such as irrigation, soil depth, soil fertility levels, and soil type had significant effects on perceived land values. For example, irrigated plots, *ceteris paribus*, were perceived to have values 45% higher than non-irrigated plots, whereas a one-level rise in ordinal soil depth increased land values by 5% and in soil fertility by 18%. Such other factors as conservation investments and erosion risk were found to have no significant effects on land values. This shows that the land value method can be used as an alternative to PCA for valuing the effect of NRM investments on land quality aspects that influence productivity. Due to market failures and imperfections, including incomplete land tenure rights, changes in other attributes like public goods aspects and non-use values cannot be easily valued using the land value approach. The contingent valuation (CV) method is useful for valuing such changes.

Contingent valuation (CV) method

In cases where people's preferences are not revealed directly or indirectly through conventional markets, the CV method is used to assess their WTP for marginal changes in quantity or quality of goods and services by posing hypothetical questions. The CV method is a direct stated preference method that involves asking a sample of a relevant population questions about their WTP or willingness to accept (WTA). The monetary value of the change in NRM is acquired by asking respondents about their WTP for a benefit, or what they are WTA by way of compensation to tolerate a cost or forgo a benefit. The name *contingent valuation* originates from the fact that the valuation is contingent on the hypothetical scenario put forward to the respondent. CV is mainly used for valuation of non-marketed ecosystem services and the non-use values associated with non-excludable and non-divisible resource and environmental flows. Unlike the indirect methods that use observed or revealed behaviour, the CV method relies on stated or potential behaviour as expressed in hypothetical markets. An important advantage of the CV method is that responses to WTP and WTA questions provide theoretically correct measures of welfare change as defined in Equations 5–9.

As discussed earlier, the appropriate welfare measures for changes in environmental quantity or quality are compensating surplus (CS) and equivalent surplus (ES) measures. Theoretically, an individual can be asked about WTP or WTA for either an improvement or a deterioration (Table 2.5). Which question is appropriate depends on the implied property right for the specific situation. The CS measure relates to the initial welfare level and implies entitlements to the *status quo*. Thus, asking about WTP to secure an improvement, or WTA compensation to tolerate a loss, implies that the individual is entitled to the existing level. The ES measure relates to the welfare level after the change and suggests the implied property rights in the change. Asking about WTA compensation to forgo an improvement implies an entitlement to the higher level, while WTP to avoid deterioration implies an entitlement to the lower level. WTP is also constrained by income whereas WTA is not. As a result, estimates of WTA tend to be higher than WTP. Some authors suggest using WTP for situations where individuals are expected to gain from an improvement and WTA in situations where people are forced to give up or suffer some damage to their welfare (Carson, 1991). Mitchell and Carson (1989) discuss ways to frame the payment questions to elicit WTP. Arrow *et al.* (1993) provide a guide for best-practice CV studies.

Table 2.5. Welfare measures for environmental quality and quantity changes.

	Compensating surplus (CS)	Equivalent surplus (ES)
Improvement	WTP[a] for the change to occur (to secure a benefit)	WTA[b] compensation for the change not occurring (to forgo a benefit)
Deterioration	WTA compensation for the change occurring (tolerate a loss)	WTP for the change not to occur (to prevent a loss)

[a]WTP = willingness to pay.
[b]WTA = willingness to accept.

In a nutshell, application of the CV method requires the following steps:

- Create a survey instrument to elicit WTP/WTA and the means of payment or compensation
- Administer the survey instrument with a sample population
- Analyse the responses and estimate the average and marginal WTP/WTA
- Estimate the total WTP/WTA for the population of interest.

In developed countries, various survey methods including mail and telephone surveys have been used. In developing countries, in-person interviews remain the most feasible and reliable option. Such surveys often start with discussions with key informants and focus groups followed by pilot testing of the survey format. The actual data collection should introduce the changes in the resource or environmental conditions being valued and the expected benefits or trade-offs to society resulting from this change. Pictures and maps can be used to illustrate these points. The survey should also include standard data on the socio-economic condition of the respondent (e.g. age, education, assets, income, etc.). Various approaches to eliciting WTP or WTA are suggested. Open-ended questions like 'What is the most you are willing to pay for…' or 'What is the minimum that you are willing to accept as compensation for …' have been commonly used in the past. This approach has been criticised for inviting strategic bias, by which respondents may use their replies to influence a more favourable research outcome (e.g. to reduce a payment they might expect to have to pay). In actual markets, buyers are offered a price and may bargain from there to arrive at the selling price. Many respondents find the open-ended approach difficult and fail to provide any bids. The iterative bidding approach that starts with an initial amount to be revised up or down until a no-change point is reached, has been used as an alternative to open-ended questions. This approach is now being abandoned because of a starting-point bias, i.e. the WTP/WTA amount tends to be systematically related to the initial bid value. An alternative approach that is gaining popularity is the binary choice or referendum format, where respondents are asked whether they would vote in support of a proposed change in policy or environmental condition that would cost a US$$x$ increase in tax payments. The offered amount can be varied and randomly assigned to the sample. Follow-up questions to the binary choice payment questions have also been used to identify the upper and lower bounds for the bids. It seems that depending on the design, a discrete-choice format with follow-up questions can mimic a bargaining process, commonly used in transactions in developing countries (FAO, 2000).

Once the data from a representative sample are collected, statistical analyses will be needed to estimate the average WTP/WTA and the aggregate value of the ecosystem service. The type of analysis of CV responses depends on the elicitation format used. If the payment question is open-ended, the stated WTP/WTA bids can simply be averaged.[3] The sample average is an unbiased estimator of the population mean. In cases where outliers influence the average bid, the median is a best estimate of a representative central value. As defined earlier, the WTP can be given as:

$$WTP_h = e(p, q^0, u^0, x) - e(p, q^1, u^0, x) \quad (18)$$

where $e(.)$ is the expenditure required to attain a given level of utility, WTP_h is the *WTP* for household *h* and $\boldsymbol{x}$ represents the socio-economic characteristics of the respondent and other exogenous variables that affect the WTP. In order to check the internal validity of the CV method, a regression model can be fitted as:

$$WTP_h = X_h \beta + \eta_h \quad \eta_h \sim (0, \sigma^2) \quad (19)$$

where X_h is a vector of explanatory variables and η_h is the error term distributed normally with means *0* and standard deviation σ. This function is often called the valuation function. As shown below, the valuation function is especially relevant for use in benefit transfer studies. It allows the new user to plug in mean values of explanatory values to predict the benefit value for a new setting. If the binary choice payment format is used, alternative methods for estimating the mean bid are discussed in the literature (Hanemann, 1984). Additional analyses will be needed to estimate the average bid and aggregate values for the change. The binary response is an indicator for the WTP/WTA that is observed only when the respondent's WTP/WTA is less than the offered bid value. A maximum likelihood probit model can be estimated using these binary responses to identify the factors that determine the probability of a positive response to a given bid. The mean WTP/WTA can then be obtained by calculating the predicted value of the valuation function at the mean values of the covariates.

Once the average WTP or WTA values for a representative group of people have been determined, they are aggregated to a total value directly dependent on the number of individuals affected. For ecosystem services that provide international public goods, the number of people with a positive WTP is likely to be large, and a modest estimate of the population size needs to be made. In principle, scaling up the average WTP/WTA across the affected population is analogous to the vertical summation of individual compensated demand curves for public goods.

As the examples in Box 2.1 demonstrate, carefully designed and administered CV surveys can provide useful estimates of the value of the changes in non-marketed ecosystem services resulting from NRM investments. The reliability of estimates and validity of results depend on the design and implementation. Of course, they also share the weaknesses of all stated-preference methods.

Box 2.1. NRM-related CV studies in the developing countries.

Today, there are several examples and good reviews of CV applications in developing countries (Munasinghe and Lutz, 1993; FAO, 2000; Pearce *et al.*, 2002). Whittington (1998) examines issues and lessons learned in administering CV surveys in developing countries. Two studies relate to NRM impact assessment in agriculture. Holden and Shiferaw (2002) applied the CV approach to estimate farmers' WTP to mitigate soil degradation in Ethiopia. In the light of increasing land degradation in the highlands, the intention was to elicit the farmers' WTP for NRM technologies that might not provide immediate benefits to farmers. The survey questions were framed to reflect the attributes of available and proposed NRM technologies with three alternative scenarios. Farmers were asked about their WTP for new NRM technologies that: a. sustain land productivity at current levels, b. enhance productivity by a fixed amount from the second year onwards, and c. enhance productivity by a fixed amount from the sixth year onwards. Teff (*Eragrostis tef*), the locally grown cash and staple cereal, was used as numeraire. The WTP surveys were administered as part of a larger survey where broader socio-economic data were collected that allowed estimating regression equations to identify the WTP covariates and check for internal validity. Farmers' expressed WTP for land management options was significantly lower than those implied by experimental and econometric estimates of soil erosion and productivity decline. Shyamsundar and Kramer (1996) applied the CV method to value forest ecosystem services in Madagascar using a binary choice payment format to elicit the local people's WTA compensation for welfare losses associated with land-use restrictions and loss of access to forests. Due to the extreme poverty of farmer respondents, Shyamsundar and Kramer used WTA questions specified in terms of bags of rice, the local staple food. They estimated a probit function and a valuation function to infer the WTA for specific households and the mean for the sample. This was used to estimate the aggregate use value of the forest service flows to the local people.

Although the approach has been widely applied for benefit–cost analysis of projects with environmental impacts, its use in assessment of technology and policy impacts in agriculture and natural resources has been scanty. CV surveys can be very useful for generating information that will inform policy choices in developing countries where market failures are more pervasive. The method is a relatively simple and cost-effective means (especially when literacy is widespread) to estimate the value of public goods and non-market ecosystem services associated with NRM investments.

Choice modelling (CM)

Choice modelling (also called choice experimentation) is an indirect stated preference method that arose from conjoint analysis and has been employed in marketing, transportation and psychology. Bennett and Blamey (2001) provided a collection of papers on the theory and application of CM in environmental valuation. Alpizar *et al.* (2003) provided a good review of using CM for non-market valuation. It differs from typical conjoint methods in that individuals are asked to choose from alternative bundles of attributes (alternatives) instead of ranking or rating them. Under the CM approach, respondents are asked to choose their most-favoured choice out of a set of three or more alternatives, presenting variations in the attributes of the item being valued (Adamowicz *et al.*, 1998). The *status quo* is given as one of the alternatives in the choice set. Each respondent gets the same number of choice sets, but the composition of the choice sets varies across respondents. This allows the researcher to value changes in attributes and the trade-offs compared to the *status quo* and different alternatives. Furthermore, in the case of damage to a particular attribute, compensating amounts of other goods (rather than compensation based on money) can be calculated. While several statistical methods can be used, multinomial and conditional logit models are commonly used to analyse the choices that people make. This approach can provide substantially more information about a range of possible alternative policies and can reduce the sample size needed compared to the CV method. However, survey design issues with the CM approach are often much more complex due to the number of goods that must be described and the statistical methods that must be employed. This may limit its application for valuation of NRM impact in the context of developing countries.

Comparison of alternative valuation methods

This section has reviewed the promising methods that can be used in valuation of NRM impacts. The choice of valuation methods depends on the existence of markets, the spatial and temporal diffusion of the impact, and whether the values relate to use or non-use values. Direct-use values such as productivity changes can be measured indirectly using data from observed or stated market behaviour of producers and consumers in conventional or surrogate markets. For non-use values, like benefits captured by future generations (sustainability) and indirect-use values such as ecosystem regulation functions, there is no observable market behaviour that contains relevant information, hence hypothetical behaviour in constructed markets must be used. The PCA, RCA, HP and CV methods are the most commonly applied in relation to environmental resources, and they offer promising opportunities for valuation of NRM impacts. Each of these methods measures different aspects of the total economic value (see Fig. 2.1) and has its strengths and weaknesses. The PCA and RCA use observed market information to measure use values indirectly. HP is also an indirect method that uses surrogate markets to measure use values. The CV is the direct stated preference method mainly used in respect to non-use values, but it could also be applied for use values.

Perhaps because of their relative ease and cost-effectiveness, the PCA and RCA are most commonly used in NRM valuation exercises. These two approaches measure different aspects of resource degradation focusing on productivity change and the costs of replacing damaged ecosystem services; hence, they often provide divergent estimates. The relative size of the two estimates may also be useful for NRM technology choice and farmer-investment decisions. Farmers are unlikely to adopt resource management practices unless the productivity benefits are higher than the investment costs. Drechsel *et al.* (Chapter 9, this volume) discuss this and the related pros and cons of these methods in more detail. The PCA needs to be computed as a net gain over the less-conserving alternative. When data on changes in resource conditions and productivity are available, the PCA is a recommended method for measuring the values of marketed productivity impacts. A major weakness is its inability to value external effects and non-market benefits.

The RCA imposes strict assumptions – that the substitute be the least-cost alternative and that the cost be less than the aggregate WTP. The RCA cannot be used to value ecosystem services that do not have marketed substitutes, and it cannot measure non-use values. When production data are limiting, RCA can be a useful alternative to value changes in certain resources like soil quality.

If markets reflect changes in environmental quality and resource conditions, the HP method is another promising technique for estimating benefits. The disadvantages of HP applied to land markets include lack of transaction data and failure of land markets to fully reflect non-productivity related changes in ecosystem services. When sales transactions are limited, land rental markets may provide an alternative source of relevant information.

When existing markets cannot be used to acquire the necessary information, the CV and CM methods can be the most useful approaches for NRM valuation. The strength of these approaches is their flexibility to generate information from constructed markets to measure both use and non-use values relevant to a given situation. When properly applied, the WTP/WTA estimates provide theoretically correct measures of welfare change. Although the survey design is more complex, the CM requires less data and provides more policy-relevant information than the CV method. However, these methods are criticised for their reliance on hypothetical markets where true behaviour is unobservable and also for survey implementation problems that may bias results. Several approaches can be used to reduce bias. If non-market ecosystem services and non-use values are a significant part of NRM impacts (as is often the case), these are the only conceptually justified approaches, and should be carefully applied depending on the availability of technical and financial resources.

When new valuation studies cannot be made due to time or financial constraints, the *benefit transfer approach* can be used to apply valuation estimates from other studies of similar changes in environmental quality at a new site. Although termed '*benefit transfer*', damage estimates can also be transferred. Four benefit transfer approaches exist: unit value transfer (e.g. direct transfer of mean WTP per household), adjusted unit value transfer (e.g. corrected for differences in per capita income levels), value function transfer, and meta-analysis. Value function transfer uses regression equations estimated for one location to predict resource values in another location, while meta-analysis uses independent case studies to synthesise and provide a summary estimate of resource value for specific conditions. Value transfer generally increases the uncertainty in the estimated environmental value. The early examples of benefit transfer were conducted in an uncritical manner, often lacking sound theoretical, statistical and empirical basis, and did not question the validity and reliability of the transferred values.

Recently, there has been growing interest in the development of benefit transfer methods and statistical techniques (Navrud, 2004; Navrud and Ready, 2004). Results from validity tests have shown that the uncertainty in spatial and temporal benefit transfer can be quite large, especially when economic and ecological conditions are quite different. Thus, care should be taken in using benefit transfer in policy uses where the demand for accuracy is high.

At present, there is a dearth of both benefit transfer applications in developing countries and sufficient valuation studies for meta-analyses. There is also a lack of validity tests of benefit transfer between developing and developed countries. One such study underlines the considerable uncertainty in using benefit transfer estimates (Barton and Mourato, 2003). Correcting for differences in gross domestic product (GDP) per capita seems to improve benefit transfer, but the actual difference in income levels in the two samples does not typically correspond to the differences in GDP, so correcting for income levels in unit-value transfers often makes things worse (Barton and Mourato, 2003; Navrud and Ready, 2004). Since the explanatory power of WTP functions is often poor, value function transfers may not do a better job in transferring benefits than simple value transfers.

Impact Evaluation

Since NRM interventions are expected to provide multiple economic and environmental benefits to various stakeholders including smallholder farmers, NRM impact evaluation should include non-marketed ecosystem goods and services along with marketed economic benefits. The market and non-market values of changes in goods and environmental services estimated using the valuation methods discussed in this chapter are vital in estimating costs and benefits that are used to evaluate the overall impact of the intervention. This requires a more holistic approach that would expand conventional impact assessments (Baker, 2000) to include non-tradable goods and environmental services. The welfare gains from NRM investments associated with direct economic benefits (e.g. yield gains or cost savings) can be assessed using a conventional approach. Unfortunately, as shown earlier, NRM investments generate other sustainability benefits and ecosystem services that have use and non-use values to people. Indirect welfare gains from such environmental improvements are legitimate parts of the welfare changes associated with NRM interventions and need to be considered in impact evaluation. The total welfare gain to people can then be decomposed into direct economic benefits derived from productivity changes and indirect environmental economic components. When NRM technologies generate productivity (including cost-saving) benefits in addition to changes in resource quality and sustainability, both sources of welfare gain are likely to be significant. In cases where the impact is expressed mainly in terms of non-tangible ecosystem service flows, the environmental and sustainability benefits could become a major part of the total welfare gain. Although precise estimation of non-market ecosystem service flows is always difficult, the valuation methods discussed above can be used to estimate the multiple welfare benefits associated with NRM interventions.

The conventional economic surplus approach (Alston *et al.*, 1995; Swinton, Chapter 7, this volume) includes changes in consumer surplus and producer surplus associated with supply shifts and price changes from changes in agricultural technology. As shown above, welfare gains associated with changes in environmental conditions are measured using the ES and CS measures of welfare gains to consumers and the producer surplus benefits to producers. When these extended economic and environmental welfare benefits are known, the social impact of research and development (R&D) investments in NRM can be evaluated using the benefit–cost analysis approach. The economic welfare gains from NRM can be given as:

$$\pi_t^{P} = \pi_t^{PN} - \pi_t^{PT} \quad (20)$$

where π_t^P is the period t productivity-related economic gain from change in NRM that can be calculated as the difference in net benefits between the new (π_t^{PN}) and the traditional (π_t^{PT}) NRM practices. π_t^P is essentially the flow of consumer and producer surpluses associated with productivity changes generated by NRM interventions. The environmental welfare gains from NRM can similarly be given as:

$$\pi_t^{E} = \pi_t^{EN} - \pi_t^{ET} \quad (21)$$

where π_t^E is the period t environmental welfare gains that can be calculated as the difference between environmental benefits from the new (π_t^{EN}) and the traditional (π_t^{ET}) NRM practices. This is the total WTP/WTA measure of welfare change resulting from changes in the flow of non-productivity related ecosystem services valued by people. π_t^E is essentially the social WTP for better NRM to enhance agricultural sustainability and the flow of ecosystem services (environmental quality). These values can also include the changes in external or off-site effects of NRM interventions. In order to assess the social impact of NRM interventions, additional information on the research, development and extension costs will be needed. If it is assumed that the flow of these costs is given by RE_t, such costs incurred up front could be quite significant, especially when the benefit flow is delayed because of the long time required for technology development and adaptation and when a positive discount rate is used in the calculation of net present values from the investment. The net welfare gain from NRM interventions will then be estimated as:

$$NPV = \sum_{t=1}^{n}\left(\pi_t^{P} + \pi_t^{E} - RE_t\right)\left(1 + r\right)^{-t} \quad (22)$$

where NPV is the social net present value of the NRM intervention, r is the real social rate of discount. Some of the changes in ecosystem services (e.g. soil fertility) may be reflected in productivity changes. The additive framework given in Equation 22 is valid when the productivity benefits and non-productivity related environmental or sustainability outcomes are clearly separable. When such separation is not possible, the approach can lead to double counting and overestimation of NRM impacts. An important area for further research is on the mechanisms used to separate productivity and sustainability effects, and under which conditions the estimated productivity and environmental values can be additive. If the social benefits of interventions are higher than the costs of the interventions, i.e. the total benefit is higher than the costs and gainers from the intervention can overcompensate losers, then NRM is considered to be socially beneficial. This may not be the case when externalities are ignored in the analysis. When environmental net benefits (π_t^E) cannot be estimated, the impact could be assessed in terms of the required tradeoffs and implications for sustainability of productivity gains. This could also include situations where the impact of NRM interventions is reflected in terms of reductions in production risk, improved stability of production, and reduced vulnerabilities of rural households to droughts, floods and other environmental shocks.

Parameters estimated for linking NRM changes with ecosystem goods and services (e.g. to estimate the effect of soil and water conservation on productivity) can also be integrated into bioeconomic models. The integration of important biophysical information and ecological processes with economic decision behaviour through bioeconomic modelling allows simultaneous assessment of welfare effects and environmental and distributional outcomes. Holden (Chapter 8, this volume) and Shiferaw and Holden (Chapter 12, this volume) further discuss these issues. One innovative approach for future research to evaluate the social impacts of NRM interventions is to compare the stream of aggregated net benefits (estimated based on optimised values derived from the model) with R&D investment costs.

Conclusions

The changes in environmental and resource service flows associated with NRM investments accrue over different temporal and spatial scales. Many of these ecosystem services generate valuable direct and indirect welfare benefits to people. When NRM investments generate private and public goods benefits, valuation of such changes is a crucial first step in the evaluation of overall social impacts. A prerequisite to effective valuation of NRM impacts is the ability to predict the changes in ecosystem service flows that can be attributed to the intervention itself. This requires a strong partnership between agroecologists and economists. The scientific understanding of ecosystem functions and services and how they are affected by human interventions is still incomplete. More work is needed to understand and quantify the effect of NRM interventions on ecosystem functions and services. Appropriate indicators are needed to measure selected changes in ecosystem services. Without reliable data, valuation efforts will not provide any useful economic values. With advances in agroecology and biophysical simulation modelling, the ability to predict the likely effects of certain interventions has improved. The economic approach to valuation of ecosystem services is based on the trade-offs that people are prepared to make in exchange for these services. The changes in the flow of ecosystem services can affect human welfare in complex ways and through marketed or non-marketed activities.

This chapter has offered an overview of ecosystem services from NRM investments, the need for indicators of ecosystem condition, key challenges to valuation of environmental services, and recent advances in the methods available for valuation of economic and environmental benefits. There is a dearth of examples in valuation of NRM impacts, especially in the context of developing countries. However, the recent progress in developing valuation methods has created new opportunities. For NRM impact assessment, the estimated values of changes in ecosystem services need to be social scarcity prices that account for non-marketed outcomes and external effects. Impact assessment of agricultural technologies has often ignored external effects and environmental impacts. However, resource management interventions typically generate non-marketed sustainability and environmental benefits. The greatest challenge in valuation of NRM impacts is in quantification and measurement of these non-productivity related outcomes and non-market benefits. Such standard techniques as the productivity change approach or revealed preference methods like defensive expenditures, provision costs or hedonic pricing can be used to measure productivity-related outcomes. However, markets and observed behaviour cannot be used for valuation of impacts on non-use and indirect use values related to regulation and habitat provision functions. When the impacts can be quantified using measurable indicators, stated preference methods could be used for the valuation of such effects.

The CV method is the most appropriate option when the indirect and surrogate market options cannot be used to value the change in ecosystem services. It is most appropriate for valuing non-use values and non-tradable use values of ecosystem services. However, the CV method has only rarely been applied to NRM impact assessment in the developing countries. Since poverty limits the ability to pay, the WTA compensation is a preferred approach for valuation of ecosystem services in poor communities. Choice modelling is an alternative and promising sated preference method. It is important to test and enhance these methods for valuation of non-market ecosystem services associated with NRM. Case studies are required to gain experience and develop improved protocols in the application of CV and/or CM methods for NRM impact assessment. In some situations benefit transfer approaches can be used to inform urgent policy decisions. However, more research is needed to enhance the transferability of benefits between countries or eco-regions.

Once the values of changes in ecosystem services are estimated, impact evaluation needs to compute the overall social gains from NRM interventions. Many NRM interventions imply supply shifts for both market and non-market goods and environmental services. This implies the need to estimate the size of the supply shift and the resulting effect on estimated unit resource values. However, more work is needed to understand how such values can be effectively integrated into impact assessment studies. In some cases, high uncertainties about the nature and magnitude of changes and the temporal and spatial impacts of NRM interventions may limit the policy relevance of monetary values. More research is needed to improve the validity and reliability of these estimates for use in policy analysis and impact assessments.

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Endnotes

[1]In *ex ante* assessments, prevailing market prices may not be an appropriate way to value productivity changes unless a 'small project' assumption is imposed so that prices remain largely unaffected. If productivity changes are expected to affect market prices, appropriate adjustments can be made using the general equilibrium framework.

[2]The travel cost method is another surrogate market approach for valuation of recreational use values of ecosystem services. It has been applied widely for the valuation of wildlife in protected areas. Since typical agricultural NRM investments do not provide marketable recreational benefits, the method is not discussed here.

[3]Apart from average WTP, marginal WTP is also of interest. This can be determined by estimating an inverse demand curve with price as a function of quantity. An inverse demand curve is also essential for estimating economic surplus rather than assuming constant average WTP.

References

Adamowicz, V., Boxall, P., Williams, M. and Louviere, J. (1998) Stated preference approaches for measuring passive use values: Choice experiments and contingent valuation. *American Journal of Agricultural Economics* 80, 64–75.

Alpizar, F., Carlsson, F. and Martinsson, P. (2003) Using choice experiments for non-market valuation. *Economic Issues* 2003, 83–110.

Alston, M.J., Norton, W.G. and Pardey, P.G. (1995) Science under scarcity. *Principles and Practice for Agricultural Research Evaluation and Priority Setting.* Cornell University Press, Ithaca, New York, and London, UK, 585 pp.

Altieri, M. (2002) Agroecology: the science of natural resource management for poor farmers in marginal environments. *Agriculture, Ecosystems and Environment* 93, 1–24.

Arrow, K.J., Solow, R., Portney, P.R., Leamer, E.E., Radner, R. and Shuman, H. (1993) Report of the National Oceanographic and Atmospheric Administration panel on contingent valuation. *Federal Register* 58, 4601–4614.

Arshad, M.A. and Martin, S. (2002) Identifying critical limits for soil quality indicators in agro-ecosystems. *Agriculture, Ecosystems and Environment* 88, 153–160.

Baker, J. (2000) *Evaluating the Impact of Development Projects on Poverty: A Handbook for Practitioners.* The World Bank, Washington, DC, 217 pp.

Barbier, E.B. (1998) The economics of soil erosion: theory methodology and examples. In: Barbier, E.B. (ed.) *The Economics of Environment and Development. Selected Essays*. Edward Elgar, Cheltenham, UK, pp. 281–307.

Barton, D. and Mourato, S. (2003) Transferring the benefits of avoided health effects from water pollution between Portugal and Costa Rica. *Environment and Development Economics* 8(2), 351–372.

Bateman, I.J. and Willis, K.G. (eds) (1999) *Valuing Environmental Preferences. Theory and Practice of the Contingent Valuation Method in the US, EU, and Developing Countries.* Oxford University Press, Oxford, UK, 645 pp.

Bennett, J. and Blamey, R. (eds) (2001) *The Choice Modelling Approach to Environmental Valuation*. Edward Elgar, Cheltenham, UK, 69 pp.

Bingham, G., Bishop, R., Brody, M., Bromley, D., Clark, E.T., Cooper, W., Costanza, R., Hale, T., Hayden, G., Kellert, S., Norgaard, R., Norton, B., Payne, J., Russell, C. and Suter, G. (1995) Issues in ecosystem valuation: improving information for decision making. *Ecological Economics* 14, 73–93.

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

Bojo, J. (1991) *The Economics of Land Degradation: Theory and Applications to Lesotho.* Stockholm School of Economics, Stockholm, Sweden.

Braden, J.B. and Kolstad, C.D. (1991) *Measuring the Demand for Environmental Quality.* Elsevier Science Publisher B.V., Amsterdam, The Netherlands, 370 pp.

Campbell, B., Sayer, J.A., Frost, P., Vermeulen, S., Ruiz-Perez, M., Cunningham, A. and Ravi, P. (2001) Assessing the performance of natural resource systems. *Conservation Ecology* 5(2), 22. [online] http://www.consecol.org/vol5/iss2/art22/index.html

Carson, R.T. (1991) Constructed markets. In: Braden, J. and Kolstad, C. (eds) *Measuring the Demand for Environmental Commodities*. North-Holland, Amsterdam, The Netherlands, pp. 121–162.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Parueol, J., Raskin, R.G., Sutton, P. and van den Belt, M. (1998) The value of the world's ecosystem services and natural capital. *Ecological Economics* 25, 3–15.

Dixon, J.A., Scura, L.F., Carpenter, R.A. and Sherman, P.B. (1994) *Economic Analysis of Environmental Impacts*. Earthscan Publications, London, UK, 210 pp.

Dumanski, J. and Pieri, C. (2000) Land quality indicators: research plan. *Agriculture, Ecosystems and Environment* 81, 93–102.

Ellis, G.M. and Fisher, A.C. (1987) Valuing the environment as input. *Journal of Environmental Management* 25, 149–156.

FAO (Food and Agriculture Organization of the United Nations) (2000) Applications of the contingent valuation method in developing countries: a survey. *FAO Economic and Social Development Paper 146.* FAO, Rome, Italy, 70 pp.

Farber, S.C., Costanza, R. and Wilson, M.A. (2002) Economic and ecological concepts for valuing ecosystem services. *Ecological Economics* 41, 375–392.

Freeman, A.M. (1993) *The Measurement of Environmental and Resource Values: Theory and Methods.* Resources for the Future, Washington, DC, 516 pp.

Gebremedhin, B., Swinton, S.M. and Tilahun, Y. (1999) Effects of stone terraces on crop yields and farm profitability: results of on-farm research in Tigray, Northern Ethiopia. *Journal of Soil and Water Conservation* 54, 568–573.

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

Hanemann, W.M. (1984) Welfare evaluations in contingent valuation experiments with discrete responses. *American Journal of Agricultural Economics* 66, 322–341.

Harrison, D. and Rubinfeld, D.O. (1978) Hedonic housing price and demand for clean air. *Journal of Environmental Economics and Management* 5, 81–102.

Holden, S.T. and Shiferaw, B. (2002) Poverty and land degradation: peasants' willingness to pay to sustain land productivity. In: Barrett, C.B. and Place, F. (eds) *Natural Resources Management in African Agriculture: Understanding and Improving Current Practices.* CAB International, Wallingford, UK, pp. 91–102.

Johansson, P.-O. (1987) *The Economic Theory and Measurement of Environmental Benefits.* Cambridge University Press, Cambridge, UK, 236 pp.

Kim, S.H. and Dixon, J.A. (1986) Economic valuation of environmental quality aspects of upland agricultural projects in Korea. In: Dixon, J.A. and Hufschmidt, M.M. (eds) *Economic Valuation Techniques for the Environment: A Case Study Workbook*. Johns Hopkins University Press, Baltimore, Maryland.

Magrath, W. and Arens, P. (1989) The case of soil erosion on Java: A natural resource accounting approach. *Environment Department Working Paper 18.* The World Bank, Washington, DC.

Mitchell, R.C. and Carson, R.T. (1989) *Using Surveys to Value Public Goods: The Contingent Valuation Method.* Resources for the Future, Washington, DC, 67 pp.

Munasinghe, M. and Lutz, E. (1993) Environmental economics and valuation in development decision making. In: Munasinghe, M. (ed.) *Environmental Economics and Natural Resource Management in Developing Countries*. The World Bank, Washington, DC, 240 pp.

Navrud, S. (2004) Value transfer and environmental policy. In: Tietenberg, T. and Folmer, H. (eds) *The International Yearbook of Environmental and Resource Economics 2004/2005. A Survey of Current Issues.* Edward Elgar, Cheltenham, UK, and Northampton, Massachusetts, pp. 189–217.

Navrud, S. and Ready, R. (eds) (2004) *Environmental Value Transfer: Issues and Methods.* Kluwer Academic Publishers, Dordrect, The Netherlands (in press).

Oriade, C. and Dillon, C.A. (1997) Developments in biophysical and bioeconomic simulation of agricultural systems: A review. *Agricultural Economics* 17, 45–58.

Pagiola, S., Bishop, J. and Landell-Mills, N. (eds) (2002) *Selling Forest Environmental Services: Market Based Mechanisms for Conservation and Development*. Earthscan Publications, London, UK, 299 pp.

Pearce, D., Pearce, C. and Palmer, C. (eds) (2002) *Valuing the Environment in Developing Countries: Case Studies.* Edward Elgar, Cheltenham, UK.

Rosen, S. (1974) Hedonic prices and implicit markets: product differentiation in perfect competition. *Journal of Political Economy* 82(1), 34–55.

Shiferaw, B. and Holden, S.T. (2001) Farm level benefits to investments for mitigating land degradation: empirical evidence from Ethiopia. *Environment and Development Economics* 6(3), 335–358.

Shiferaw, B., Reddy, R.V., Wani, S.P. and Rao, G.D.N. (2003) Watershed management and farmer conservation investments in the semi-arid tropics of India: analysis of determinants of resource use decisions and land productivity benefits. *Socioeconomics and Policy Working Paper Series 16.* ICRISAT, Patancheru, Andhra Pradesh, India, 28 pp.

Shyamsundar, P. and Kramer, R.A. (1996) Tropical forest protection: an empirical analysis of the costs borne by local people. *Journal of Environmental Economics and Management* 31, 129–144.

Smyth, A.J. and Dumanski, J. (1993) FESLM: An international framework for evaluating sustainable land management. *World Soil Resources Report 73.* Food and Agriculture Organization of the United Nations, Rome, Italy, 77 pp.

Whittington, D. (1998) Administering contingent valuation surveys in developing countries. *World Development* 26(1), 21–30.