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An ecological perspective on the valuation of ecosystem services

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Abstract

Ecosystem services are the conditions and processes through which natural ecosystems and the species that make them up, sustain and fulfil human life. Ecosystem service valuation is being developed as a vehicle to integrate ecological understanding and economic considerations to redress the traditional neglect of ecosystem services in policy decisions. This paper presents a critical review on the neoclassical economic framework, tools used for economic valuation of ecosystem services and the economic welfare approach to collective decision-making, from an ecological perspective. The applicability of the framework and techniques for valuing ecosystem services are evaluated in light of the challenges posed by the complex, non-linear nature of many ecosystem services. Decisions concerning ecosystem management are often complex, socially contentious and fraught with uncertainty. Although judicious application of economic valuation techniques to ecosystem services can provide valuable information for conceptualizing decision choices and evaluating management options, there are serious limitations in the economic welfare approach to decision-making. These shortcomings and their implications for ecosystem management are elucidated and alternative approaches that emphasize participation, explicit treatment of uncertainty and transparent decision-making processes are discussed.

Keywords: Ecosystem services; Valuation; Ecological economics; Contingent valuation; Cost-benefit analysis; Decision-making under uncertainty

1. Introduction

Ecosystem services are the conditions and processes through which natural ecosystems and the species that make them up, sustain and fulfil human life (Daily, 1997). The concept of ecosystem services encompasses the delivery, provision, production, protection or maintenance of a set of goods and services that people perceive to be important (Table 1). This includes goods such as seafood, forage, timber, biomass fuels, natural fibre, pharmaceuticals and industrial products, services such as the maintenance of biodiversity and life-support functions including waste assimilation, cleansing, recycling and renewal (Daily, 1997; Norberg, 1999), and intangible aesthetic and cultural benefits. Ecosystem services can be defined in myriad ways dependant on scale and perspective (Daily, 1997). However, to facilitate comparative ecological economic analyses, de Groot et al. (2002) recently constructed a typology for describing, classifying and valuing ecosystem functions, goods and services.

Economic theory recognizes four kinds of capital – human, financial, manufactured and natural. Ecosystem services are the equivalent of 'natural capital'. Developed economies have focused primarily on using the first three (which were considered limiting factors to development) to transform natural capital (which was considered 'free' and abundant) into consumer products and services (Hawken et al., 1999).

Ecosystem services tend to fall into the categories of open access and pure public services. This means that they tend to have no producer property rights, ambiguous entitlement structures and prohibitive transaction costs (Sternberg, 1996). As no one "owns" or has "rights" to these services and others cannot be excluded from using or benefiting from them, little incentive exists for beneficiaries to manage ecosystem services sustainably (Dasgupta et al., 2000). Additionally, it is difficult to extract compensation payment from beneficiaries for redistribution among intra- and intergenerational parties that might be affected by negative outcomes such as loss

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Table 1
A classification and examples of ecosystem services (adapted from Daily, 1999)

Production of goods
Food: terrestrial animal and plant products, forage, seafood, spice
Pharmaceuticals: medicines, precursors to synthetic drugs
Durable materials: natural fibre, timber
Energy: biomass fuels, low-sediment water for hydropower
Industrial products: waxes, oils, fragrances, dyes, rubber, precursors to synthetic products
Genetic resources: the basis for the production of other goods
Regeneration services
Cycling and filtration services: detoxification and decomposition of wastes, renewal of soil fertility, purification of air and water
Translocation services: dispersal of seeds necessary for revegetation, pollination of crops and native vegetation
Stabilizing services
Partial stabilization of climate
Moderation of weather extremes (e.g., temperature and wind)
Regulation of the hydrological cycle
Maintenance of coastal and river channel stability
Compensation and substitution of one species for another when environments vary
Control of the majority of potential pest species
Life-fulfilling services
Provision of aesthetic beauty, cultural, intellectual and spiritual inspiration
Existence value
Scientific discovery
Serenity
Preservation of options
Maintenance of ecological components and systems needed for the future
Supply of goods/services awaiting discovery

of biodiversity, pollution or irreversible degradation and depletion of ecosystem services (Sternberg, 1996). In effect, ecosystem services fall outside the sphere of markets and tend to be 'invisible' in economic analyses. Costanza et al.'s (1997) seminal paper on the value of the world's ecosystem services and natural capital, asserted that "because ecosystem services are not fully 'captured' in commercial markets or adequately quantified in terms comparable with economic services and manufactured capital, they are often given too little weight in policy decisions". Conversely, it has been argued that decisions about ecosystem conservation and restoration incur costs (or forgone benefits) and can lead to misuse of resources if not guided by some concept of value or trade-off (Pearce, 1998a; Howarth and Farber, 2002).

Proponents of ecosystem service valuation believe that valuations can: (i) improve understanding of problems and trade-offs; (ii) be used directly to make decisions; (iii) illustrate the distribution of benefits and thus facilitate cost-sharing for management initiatives and (iv) spur the creation of innovative institutional and market instruments that promote sustainable ecosystem management (Alyward and Barbier, 1992; Sinden, 1994; Daily, 1997; Dasgupta et al., 2000; Armsworth and Roughgarden, 2001; Salzman et al., 2001). For instance, van Wilgen et al. (1996) compared the cost of alien plant management in South African fynbos ecosystems with that of developing additional water supply facilities and demonstrated that alien plant clearing and management was a more cost-effective approach to ensuring water production and delivery than alternative supply options such as dam construction, effluent treatment and desalinization. Valuation of additional fynbos ecosystem services such as wildflower harvest, genetic storage and tourism opportunities further strengthened the argument for continued investment in alien plant clearing programs (Higgins et al., 1997). Similarly, New York City administrators decided that investment in restoring the ecological integrity of the Catskills Mountains watershed would be less costly in the long-run than constructing a new water filtration plant (PCAST, 1998). Watershed restoration would also provide additional ecosystem services such as flood and erosion control, carbon storage and visual amenity benefits. To finance initiatives for restoration Environmental Bonds were issued to raise funds which were then used to purchase land, halt development in the watershed, compensate landowners for restrictions on private development and subsidize improvement of septic systems (PCAST, 1998).

These case studies are compelling examples of how valuation of previously overlooked ecosystem services have been useful for re-framing decisions and prompting improved management of natural capital. Ecosystem service valuation is thus being developed as a means of putting natural capital into the equation of economic 'development' and on the agenda of policy-making (Munda, 2000). The aims of this paper are to: (i) critically review the neoclassical economic framework and principal methods used to value ecosystem services from a ecological perspective; (ii) assess the economic welfare approach to decision-making and (iii) present alternative approaches for collective decision-making concerning ecosystem service management.

2. Framework of neoclassical economics

Economics concerns itself with the efficient allocation of scarce resources as a means to satisfy human wants or desires (Tisdell, 1991; Tietenberg, 1992; Freeman, 1993). Consumer sovereignty is a fundamental normative principle of economics and the dominant theory of welfare economics is based upon the view that a configuration of economic activity and resource allocation should be chosen so as to satisfy to the maximum extent possible, the wants of individuals (Tisdell, 1991; Common, 1997). The ideological perspective of neoclassical economic theory encompasses: (a) market essentialism; (b) substitution, resource fungibility and technological optimism; (c) a utilitarian, anthropocentric and instrumentalist ethical framework and (d) consumer choice theory and the notion of a 'rational actor'. The following section outlines these concepts and discusses key objections that have been raised with respect to their applicability to valuing ecosystem services and aiding environmental decision-making.

2.1. Market essentialism

A market is any context in which the sale and purchase of goods and services takes place. Markets are so ubiquitous that it has become commonplace to regard the market as the ideal institution for efficient allocation of scarce resources (Simpson, 1998; Vatn, 2000). It is this belief that underlies arguments that ecosystem services are currently un- or under-priced and need to be incorporated into the market economy so that people cease to think of them as being 'free' and also to enable their relative scarcity to be reflected in price signals (Herendeen, 1998).

Although many ecosystem services by their nature and character defy commodification, it is possible to privatize to some degree access either to these ecosystem services or to the benefits provided by these ecosystem services. Examples of existing markets include individual transferable fishing quota systems in Australia, New Zealand, Canada, Chile, Iceland and the Netherlands, tradable permits for the emission of airborne pollutants in the Regional Clean Air Incentives Market (RECLAIM) in southern California (Froom and Hansjurgens, 1996) and tradable permits for saline water discharges in the Hunter River Salinity Trading Scheme in Australia (James, 1997).

Markets for ecosystem services, however, require considerable supporting legislation to operate efficiently. The property rights need to be definable, verifiable, enforceable, transferable and at low sovereign risk (i.e., at low risk of being devalued as a result of future government decisions) (Murtough et al., 2002). Sound understanding of the scientific basis underlying the relationship between the ecosystem service and use of the property right is also crucial. Markets may never emerge for ecosystem services which cannot satisfy these requirements.

2.2. Substitution, resource fungibility and technological optimism

Substitutability is an important concept in economics and it is considered that "there are very few things that are truly unique, in the sense that they have no substitutes" (Simpson, 1998). Economists recognise that the degree to which things are substitutable depends on the spatial and temporal scale of analysis, and the level of aggregation. However, for the purposes of economic analyses, most natural resources and processes are generally regarded as 'fungible' – meaning that, every natural resource/process has an adequate substitute and is freely interchangeable with another of like nature or kind (Goodland, 1995; Norton, 1995; Dasgupta et al., 2000). This viewpoint in turn engenders technological optimism, the belief that human ingenuity and technological progress will solve the problems that dwindling natural resources pose for economic growth through the substitution of other forms of capital for natural capital (Tisdell, 1991; Rees, 1998). Such positions interpret environmental arguments about the degradation of natural capital as alarmism (Myers and Simon, 1994; Lomborg, 2001).

In reality the situation is rather more complicated. A man-made waterbody stocked with game fish may substitute for a natural wetland from a recreation and visual amenity viewpoint, but it would not be equivalent in terms of functions such as habitat provision for many native organisms. The limits of substitution are illustrated when we consider services that operate on large spatial and temporal scales, such as the role of ecosystems in soil formation and in the regulation of the biogeochemical cycle, or the ozone layer.

2.3. Ethical framework

The ethical framework of conventional neoclassical economics is utilitarian in that things count to the extent that people want them; anthropocentric in that humans assign the values and instrumentalist in that the various components of the natural world are regarded as instruments for human satisfaction (Randall, 1988). In a neoclassical framework, an entity has economic value *only* if people consider it desirable *and* are willing to pay for it.

However, with respect to virtually every environmental issue such as the conservation of endangered species or forests, management of exotic species and management of greenhouse gas emissions, people voice moral, ethical and cultural principles and judgements that differ from a utilitarian, anthropocentric and instrumentalist ethical stance. Such positions reflect a deontological ethic which is defined as a concern with rights and duties rather than with utility (Ehrenfeld, 1988; Sagoff, 1995; Spash, 1997). Humans typically regard themselves as having intrinsic value recognized by agreements such as the United Nations Charter on Human Rights. In any society, there are things which its members consider wrong to buy and sell because their commodification may reduce their value, distort their functions or create perverse incentives (Vatn, 2000). Examples include friendship, votes, human organs and other human beings.

In an analogous manner many aspects of ecosystems are imbued with intrinsic value. A deontological philosophy applied to nature might recognize similar rights for plants, animals and ecosystems (Spash, 1997) and is articulated by documents such as the Earth Charter (www.earthcharter.org). The other aspect of a deontological ethical system is duty, a sense of social responsibility believed to be implicit in the "character, commitments, responsibilities or identity of the community as a whole" (Sagoff, 1998). The implications of this for valuation are discussed later.

2.4. Consumer choice theory and the economic agent as a 'rational actor'

In consumer choice theory, the human behaviour of an agent operating in a market for private commodities is formalized using the model of the 'rational actor' and embodies the following critical assumptions (after Langlois, 1998; O'Neill and Spash, 2000):

1. The individual is self-interested and purposeful rather than narrowly selfish.
2. The individual's values are expressed by his/her preferences.
3. The individual has a single, stable, invariant set of preferences which are ordered, internally consistent and structured – transitive, reflexive, complete and continuous.
4. The strength of the individual's preferences is measurable by his/her willingness to pay (WTP) for satisfaction or willingness to accept compensation (WTA) for benefits forgone.
5. The individual is omniscient – has complete information and perfect structural knowledge about a choice/decision.
6. The individual has reliable subjective probabilities about the likelihood of different outcomes.
7. The individual is rational and acts so as to maximize utility (satisfaction of preferences), given budget constraints and assignments of probabilities to different possible states of the world.

While the restrictions and limitations of these axioms are widely recognized, there has been little scrutiny of the nature and status of these assumptions by mainstream economists (Gowdy and Mayumi, 2001) and they have been tacitly accepted as necessary for modelling the basic features of decision-making in analytical representations of reality (Krugman, 1998). Recently however, they have been seriously challenged by researchers who believe that they are incongruent with the current state of knowledge about human behaviour (Lopes, 1994; Parson and Clark, 1995; Langlois, 1998; Rabin, 1998; van den Bergh et al., 2000; Gowdy and Mayumi, 2001).

Treating preferences as fixed under all circumstances allows economists to apply the principle of consumer sovereignty and the decision model of constrained utility maximization to produce a simple definition of what is optimal (Farber et al., 2002). But preferences are mutable, particularly over longer timeframes which are important because of the temporal scale of ecosystem dynamics. They change under the influence of education, advertising, variations in abundance and scarcity, changing cultural assumptions and specific social and environmental contexts (Norton et al., 1998; Farber et al., 2002), and are often determined by changes in outcomes relative to a person's reference level (Tversky and Kahneman, 1991). This creates difficulties for determining what is optimal and requires alternative criteria for judging.

An alternative criterion that has been suggested is sustainability which embodies notions of appropriate scale, fair distribution and efficient allocation (Farber et al., 2002). This however, implies enlarging the decision-making arena to obtain social consensus on issues of scale and distribution before addressing allocation (Farber et al., 2002). This may seem like a violation of methodological individualism but it acknowledges that individuals are a part of larger social entities such as families and communities and that preference formation particularly for public goods and services is influenced by socio-cultural contexts, learning, knowledge-sharing and social discourse. Individuals have 'plural' identities and often act differently in their capacities as consumers and citizens (Pearce et al., 1989). For instance, in their capacity as citizens, individuals may make a conscious decision to pay a premium for power generated from clean technologies even if power produced using fossil fuels is cheaper. In effect, individuals may possess two or more preference orderings and use different ones in different circumstances (Söderholm and Sundqvist, 2003). This has implications for the aggregation of individual values for collective decision-making.

Contrary to assumptions, people often are uncertain or ignorant about specific pieces of information within a known structure and about the nature of the decision situation they face (Langlois, 1998). Three decades of research in cognitive psychology have demonstrated that people have very poor subjective perceptions of the likelihood of different outcomes. For instance, with respect to judgements on risks, expectations depend upon the visibility of potential hazards, catastrophic potential, the level of understanding of the issues at hand, personal experience and equitability of distribution of the risk (Pidgeon et al., 1992). Under uncertainty, people rely on a limited number of heuristic principles which reduce the tasks of assessing probabilities to simple judgemental operations and biases in judgement can lead to severe systematic errors (Tversky and Kahneman, 1974).

Constrained maximization of utility is a convenient modelling assumption and suffices when the goal of analysis is to seek an efficient solution where knowledge is perfect and no uncertainty about possible outcomes exists. However, equating rationality with utility maximization is too restrictive for real life decisions concerning ecosystem management where people often seek to optimize multiple objectives under situations of imperfect knowledge, uncertainty and limited cognitive resources. In such situations, alternative conceptions of rationality such as bounded rationality and procedural rationality (Simon, 1957, 1972) and alternative models of decision-making such as habitual behaviour, rule-following, social comparison, satisficing, regret minimization and choosing a frame for strategic interactions (Mellers et al., 1998; Gintis, 2000; Jager et al., 2000; van den Bergh et al., 2000) may be more valid.

3. What do we mean by ‘value’?

Farber et al. (2002) provides detailed exposition on the various economic concepts of ‘value’. Traditionally economics has been concerned with direct use values focussed on quantifying and analyzing goods and services that produce tangible benefits. Economists however, have broadened their scope in recognition of the growing appreciation for the indirect use, non-use, existence, bequest and option values of ecosystems (see Table 2) and have developed techniques to extend monetary valuations to these ecosystem services (Tietenberg, 1992).

4. Methods of economic valuation

The principal techniques for the monetary valuation of environmental goods and services are shown in Table 3. The following section presents a brief overview of each and highlights the associated limitations.

4.1. Production function analysis

The production function (PF) approach is based on estimating the contribution an ecosystem service makes

Table 2
A compilation of meanings of the word ‘value’ (adapted from Gilpin, 2000)

Meanings of the word ‘value’
Market value – the exchange value or price of a commodity or service in the open market
Intrinsic value – the value of entities that may have little or no market value, but have use value
Intrinsic, non-use – the value attached to the environment and life forms for their own sake
Existence value – the value attached to the knowledge that species, natural environments and other ecosystem services exist, even if the individual does not contemplate ever making active use of them
Bequest/vicarious values – a willingness to pay to preserve the environment for the benefit of other people, intra- and intergenerationally
Present value – the value today of a future asset, discounted to the present
Option value – a willingness to pay a certain sum today for the future use of an asset
Quasi-option value – the value of preserving options for future use assuming an expectation of increasing knowledge about the functioning of the natural environment

Table 3
Principal techniques for monetary valuation

Market	Basis of approach	Main techniques
Market-based	Production approach	Production function analysis (PF); replacement or restoration cost (RC).
Surrogate market	Revealed preference	Travel cost method (TCM); hedonic pricing (HP).
Simulated market	Stated preference	Contingent valuation (CV)

to the production of some marketed/marketable service such as drinking water or a fish harvest (Ellis and Fisher, 1987; Mäler et al., 1994). The analysis generally uses scientific knowledge on cause-effect relationships between the ecosystem service(s) being valued and the output level of the marketed commodity. It relies primarily on production or cost data, which are generally easier to obtain than the kinds of data needed to establish demand for ecosystem services (Ellis and Fisher, 1987). PF analysis has been applied to valuing ecosystem services such as the environmental functions of tropical wetlands (Barbier, 1994), the habitat function of mangroves on shrimp fisheries (Barbier and Strand, 1998; Barbier, 2000), the value of watershed habitat on coho salmon fishery (Knowler et al., 2003), the groundwater recharge function of wetlands on agricultural production (Acharya, 2000; Acharya and Barbier, 2000) and the water flow regulation function of forest ecosystems on hydro-electric power production (Guo et al., 2000).

The main limitation of this method is the lack of adequate data and understanding of cause-effect linkages between the ecosystem service being valued and the marketed commodity (Daily et al., 2000; Spash, 2000). Ecosystems are complex, dynamic systems whose component variables often interact in nonlinear ways across a range of temporal and spatial scales. This coupled with the impact of stochastic effects from outside the system results in considerable uncertainty in predicting the desired level of supply of ecosystem services (Daily et al., 2000). The interconnectivity and interdependencies of ecosystem services may also confound the valuation process because of the likelihood of double-counting (Barbier, 1994; Costanza and Folke, 1997). For instance, Barbier (1994) noted that the nutrient retention function of tropical wetlands might be integral to the maintenance of biodiversity. Therefore, if both ecosystem services were to be valued separately and then aggregated, this would 'double count' the nutrient retention function which is already 'captured' in the biodiversity value. The formidable challenges of understanding and modelling the spectrum of interdependent ecological functions, uses and values across varying states of ecological disturbance, may account for the preponderance of single function valuation studies in the literature on environmental valuation of ecosystem services (Turner et al., 2003).

The fact that this method is partially dependent on the demand for a marketed service also means that market forces exert considerable influence on the monetary value of ecosystem services (Sagoff, 1998). It is questionable how useful it is to measure the value of an ecosystem service based on prices for services observed in real markets that do not value those services sufficiently anyway (Norgaard, 2000).

4.2. Replacement/restoration cost technique

The restoration cost (RC) approach assesses the value of an ecosystem service by how much it costs to replace/restore it after it has been damaged (Garrod and Willis, 1999). The objective of replacing/restoring an ecosystem service to its pre-damaged state is to reinstate lost consumer surplus and non-use value (Garrod and Willis, 1999). Expenditure actually incurred on replacement/restoration is a measure of the minimum WTP to recover or continue to receive a particular benefit. It gives only a minimum estimate because more may have been spent had it been seen to be necessary to do so (Binning et al., 1995). A prominent example of this technique is Gosselink et al.'s (1974) study in which an estimate of the costs of a tertiary sewage treatment system was used as an estimate of the economic value of the nutrient removal function of a wetland.

Precise definitions of the attributes to be restored/replaced are critical (Bingham et al., 1995). For example, restoring the ecosystem services provided by a healthy waterway might involve a whole suite of remedial actions from revegetating the riparian zone to removing alien plant species to restoring the natural flow regime and so on. Subtle variations in descriptions of the characteristics of the ecosystem service to be reinstated can lead to vastly different cost estimates.

From an economic point of view, the optimal level of restoration or replacement must be determined by the value of amenity benefits to society (Garrod and Willis, 1999; Bockstael et al., 2000). Economists therefore insist that monetary values derived using RC are valid only if individuals in aggregate would be willing to incur these costs if the natural services were no longer available (Bingham et al., 1995; Bockstael et al., 2000). Indeed, prominent economists such Ellis and Fisher (1987), Pearce (1998b) and Bockstael et al. (2000) have criticised studies such as Gosselink et al.'s (1974) and Costanza et al.'s (1997) for using RC to derive economic values for ecosystem services without complying with this requirement. However, rigid adherence to this requirement might mean that poorer communities may not be able adequately to express the strength of their preferences for ecosystem services.

4.3. Travel cost method

Travel cost method (TCM) evaluates individual preferences for non-market goods where consumption is commensurate with the costs of travel to acquire it (Sinden, 1994; Garrod and Willis, 1999). TCM is predominantly applied to outdoor recreation modelling and is applicable to valuation of certain ecosystem services. For example, to evaluate recreational fishing, a TCM survey would typically gather information on travel costs, license fees, on-site expenses and capital expenditure on fishing equipment. Varying such costs and predicting fishing activity can then be used to derive surrogate demand functions for fishing at a specific location.

A range of technical issues that arise in applying the method (see Garrod and Willis, 1999; Spash, 2000) mean that the analyst's judgements (e.g., with regard to the treatment of costs) can greatly influence the monetary estimates that are inferred from these choices (Bingham et al., 1995). It is difficult to discern the extent to which aggregated costs reflect the values of concern. For instance, the closer an ecosystem is to large human settlements, the more likely there are to be frequent visitors and hence a larger aggregate monetary value may be associated with the site. On the other hand, a wilderness area with restricted access may be regarded as having little or possibly no value under the TCM. Furthermore, if visitors fail to recognise the importance or existence of a site's characteristic then this characteristic will be absent from the valuation. Ecosystem services that are not visible, commonly appreciated or well understood such as nutrient cycling capability, flow regulation, sediment control and pollination are unlikely to form part of site values.

4.4. Hedonic pricing

Hedonic pricing (HP) relies on the proposition that the value an individual places on a service is based on the attributes it possesses (Garrod and Willis, 1999). The economic value of a characteristic of the service is derived from the market price of the service (Sinden, 1994). Price per unit is regressed on the characteristics of the service and the implicit marginal value of a unit of the characteristic of interest is derived from the parameters of the regression (Sinden, 1994). HP has been applied mainly in real estate markets to estimate the contribution of environmental amenities on land and housing values (King and Sinden, 1988). Aspects of the environment contribute to the value of a particular land parcel or house. Thus, characteristics such as land condition, soil fertility, water rights, proximity to clean water, air, urban forests, recreational opportunities, peace and quiet can all be expected to increase prices of land in certain markets (King and Sinden, 1988; Tyrväinen, 1997; Geoghegan, 2002). The method implies that there exists a set of measurable attributes that will predict the price of a commodity when it is traded.

However, finding suitable variables to measure environmental attributes can be problematic. Constructing a model and estimating parameters depends on sets of prior transactions that are typically absent for ecosystem service valuation. The assumptions underlying HP mean it will give inaccurate estimates of environmental externalities if buyers lack reliable information about relevant environmental variables, are unable to maximise utility or have to operate in a market in disequilibrium (Spash, 2000). These are frequent occurrences in real-world situations.

4.5. Contingent valuation

Contingent valuation (CV) is a 'stated preference' technique attributed to Ciracy-Wantrup (1947). The procedure is based on a hypothetical market in which people are asked to manifest through questionnaires and/or interviews, their demand function for a certain environmental good/service (Garrod and Willis, 1999). CV is regarded with some reservation because it is not based on actual market behaviour (Portney, 1994). It has nevertheless been widely used for valuing ecosystem services as it is capable of eliciting monetary value for goods which have no exchange value (Freeman, 1993). It is applicable to ecosystem services in practically any context (Pearce et al., 1989).

There are several stages to conducting a CV study: setting up the CV market; obtaining WTP (if the individual does not own the service) or WTA amounts (if the individual owns the service) (Garrod and Willis, 1999); evaluating bias and calibrating responses; estimating mean and median WTP and/or WTA amounts; aggregating the WTP or WTA amounts and assessing the validity of the CV study and the many potential sources of bias (Bishop et al., 1986; Wilks, 1990; Arrow et al., 1993; Hanley and Spash, 1993; Spash, 2000). CV has been applied in well over 1600 studies relating to environmental policy issues (Gregory, 1999). Applications to ecosystem services valuation include wildlife (Samples and Hollyer, 1990; Stevens et al., 1991), dilution of wastewater, natural purification of water and erosion control (Loomis et al., 2000).

Numerous critics have pointed out that the method is fraught with technical and conceptual problems. A detailed typology of biases arising from technical issues related to survey design and execution in CV studies is available in Mitchell and Carson (1989). The description and framing of what is to be valued is critical to the reliability of the method (Bingham et al., 1995). The information a survey provides as well as the order in which questions are asked substantially influences WTP (Samples et al., 1986; Samples and Hollyer, 1990). Prior knowledge, preconceived opinions and level of understanding in respondents affects the results of CV (Lord et al., 1979; Wilks, 1990; Arrow et al., 1993). Compliance bias occurs when respondents provide bids they think the interviewer would approve of (Bishop et al., 1986; Wilks, 1990).

The composition and characteristics of the reference group, particularly their level of income and education also has a strong influence on the magnitude of bids. The *Exxon Valdez* oil spill in 1989 provides a good illustration (Gatto and De Leo, 2000). The population of the United States was used as a reference group to calculate the damage to the existence value of the affected species and ecosystems using CV methods. Exxon was ultimately ordered to pay US$5 billion in compensation to the people of Alaska for their losses. This huge figure was a consequence of the high income of the US population. If the same accident had occurred in Siberia where salaries are lower, the payout would have been much lower (Gatto and De Leo, 2000).

Zero bids may come from respondents who genuinely believe that what is being valued is not worth anything, but can also arise from a host of other reasons. Individuals may be opposed to paying because they lack adequate information on what is being valued. Or it may be a form of protest against the proposed payment vehicle or against the commodification of ecosystem services. Or they might believe that paying for the maintenance of ecosystem services is the responsibility of the government, or of other social groups (such as polluters, loggers and hunters) (Jorgensen et al., 2001). Respondents may be unwilling to accept compensation because they believe that what is being valued should be protected at all costs, or they may have an ethical objection to the trade-off being requested (Spash, 2000).

Kahneman and Knetsch (1992) demonstrated an 'embedding' effect (also known as part-whole bias) which is the tendency for respondents to state much the same WTP for a part of a resource as for the whole. They speculated that respondents may in fact acquire a sense of moral satisfaction (a 'warm glow of giving') by voluntary contribution to a public good and hence, may actually be purchasing a sense of satisfaction (see also Chilton and Hutchinson, 2000). Contrary to being motivated primarily by their wants as consumers, researchers have also found that respondents frequently respond in their capacity as citizens, basing their WTP/WTA on their desire to do their 'fair share' or on ethical concerns for what they believed was better or worse, right or wrong from a social point of view (Stevens et al., 1993; Vatn and Bromley, 1994; Sagoff, 1998).

CV can also be undermined by strategic behaviour such as free-riding, over and under-bidding (MacMillan et al., 1998). For example, if respondents believe that their WTP bids will actually be collected, they may understate their 'true' WTP for ecosystem services that have an open access or pure public nature (Garrod and Willis, 1999). WTP may also be overstated to encourage reservation of an area; or may be understated to minimize the possibility of a significant user-charge or levy. Strategic bias is problematic because it is extremely difficult to detect in CV surveys (Garrod and Willis, 1999).

Numerous studies have shown that for identical services, WTA amounts systematically and substantially exceed WTP (Gregory, 1986; Vatn and Bromley, 1994). This discrepancy may be caused by faulty questionnaire design or interviewing technique; strategic behaviour by respondents and psychological effects such as 'loss aversion' and the 'endowment effect' (Garrod and Willis, 1999).

CV markets lack incentives to induce individuals to put as much time and effort into thinking about the value of a good/service and the price they are prepared to pay for it, compared to actual markets (Garrod and Willis, 1999). There is no penalty for 'getting it wrong', unlike in actual markets where people learn from their spending mistakes (Bishop et al., 1983). People do not tend to have well-articulated preferences for non-marketed services (Kahneman and Knetsch, 1992; Diamond and Hausman, 1994), particularly ones that are novel and complex as many ecosystem services are. Consequently, responses tend to be constructed on the spot when CV questions are posed and this risks undue reliance on available cues and the method of elicitation (Payne et al., 1992; Gregory, 1999). In the absence of explicit validation, the numerous problems outlined above seem to be debilitating for the purposes of valuing ecosystem services.

5. Ecological considerations with respect to valuation

Ecosystems are complex, highly interconnected and feature nonlinear interactions between variables at a range of spatial and temporal scales. These characteristics coupled with stochastic influences mean that it is often impossible to predict their dynamics in any detail (Harwood and Stokes, 2003). The interdependencies between various types of ecosystem services mean that it may be impossible to classify certain services into independent conditions and processes for valuation (Costanza and Folke, 1997; Daily, 1997). Furthermore, to understand such dependencies, it may be necessary to perturb the system and measure correlated responses. Such manipulations are possible usually only within an adaptive management framework and relevant data will be available typically for only very few ecosystem components. In practice, unless they are inferred from first principles, such associations are ignored. Due to the inherent non-linearities of ecosystems, small modifications in dynamics can become magnified through interactions and lead to large uncertainties in not only the rate, but the direction of change of system dynamics (Arrow et al., 2000).

Resilience is the capacity of a system to maintain its characteristic patterns, structures, functions and rates of processes (such as primary productivity, allocation of photosynthate, energy exchange, nutrient cycling and food-web structure) despite perturbations (Walker, 1992). Resilience derives from partially redundant control processes that act at different scales to mitigate effects of perturbations (Carpenter and Cottingham, 1997). Although there is some degree of in-built resilience in the functioning of ecosystems, usually it is difficult to determine the exact basis, boundaries or limits of this resilience. Furthermore, the relative contribution that a particular ecosystem service makes towards maintaining ecosystem resilience is often highly context specific and might be variable over a range of time and space scales (Carpenter and Cottingham, 1997).

Walker (1992) explained the concept of ecological redundancy with the following analogy: some species are determinants, or 'drivers' of the systems of which they form a part while others might be 'passengers'. Eliminating the 'drivers' might cause a cascade effect but the loss of passengers might not result in any noticeable impact on the rest of the system. However, apparent passengers at a particular time scale might turn out to be occasional determinants (Walker, 1992). Indeed, perturbation studies such as Frost et al.'s (1995) whole-lake acidification experiments found that the patterns of compensation effects in natural populations of zooplankton *prior* to the application of environmental stress were not good predictors of functional complementarity in the *actual* response to the environmental stress. Ecosystem response to a given disturbance might depend on only a fraction of the species pool, but the critical species seem to be situation specific and can rarely be anticipated (Carpenter and Cottingham, 1997).

Alternative stable states may exist for some ecosystems and the response of ecosystems to perturbations such as variations in climate, nutrient regimes and disturbance regimes like fire, pest outbreaks and impacts of anthropogenic activities can vary from smooth to discontinuous (Scheffer et al., 2001; Scheffer and Carpenter, 2003). Ecosystem responses and services may vary linearly over a range of disturbance intensities until conditions approach a critical threshold whereupon the response can be dramatic and very refractory to reverse (Arrow et al., 2000). For instance, over-fishing can reduce fish populations below sustainable levels, engendering collapse that may be difficult or impossible to reverse (Roughgarden and Smith, 1996). Extensive land clearance can lead to rise in groundwater levels and consequent land and river salinity problems which may be impossible to remediate (Allison et al., 1990; Hobbs et al., 2003). When normal resilience mechanisms collapse, they may be superseded by new resilience mechanisms and qualitative changes in the ecosystem – changes which may be stable, but undesirable for humans (Carpenter and Cottingham, 1997; Limburg et al., 2002). Maintaining the resilience of ecosystems is vital in the management of ecosystem services, but it is not at all clear how this emergent property might be economically valued. Focussing on valuation of single elements or functions may obscure synergistic properties (Vatn, 2000).

6. Economic welfare approach to collective decision making and cost-benefit analysis

The economic welfare approach to decision making respects the principle of 'consumer sovereignty'. It is determinedly non-judgmental about people's preferences and what the individual wants is presumed to be good for the individual (Randall, 1988). Individual valuations are established by one of the techniques described above and then aggregated by simple summation to represent the valuations of a socially relevant unit such as a community, society, state and nation (Randall, 1988; Farber et al., 2002). Costs are subtracted from the overall social benefit to obtain an estimate of net social benefit which then provides an economically defensible basis for collective decisions. Advocates of this approach consider that it is the best way to make decisions in the public domain because it emphasizes the preferences of individuals rather than those of political representatives (Jenkins-Smith, 1990). cost-benefit analysis (CBA) is used extensively in investment, project and policy appraisal and is essentially a method for organizing the relevant information and data on costs and benefits to analyze options with a view to attaining efficiency in resource allocation (Department of Finance, 1991; Pearce, 1998a).

Examples of studies which have employed CBA in the valuation of ecosystem services include: van Wilgen et al.'s (1996) study on the costs and benefits of a program to eradicate alien plants from fynbos vegetation in water catchments in the Western Cape Province of South Africa and Scott et al.'s (1998) study on the costs and benefits of maintaining the ecosystem services of the shrub-steppe habitat at the Fetzner/Eberhardt Arid Lands Ecology (FEALE) Reserve in Washington State, USA.

Discounting is a standard practice in CBA used to obtain net present value (NPV). It is calculated by the formula $B_t/(1+r)^t$, where B_t represents the amount the beneficiary will receive in future year, t, r represents the discount rate and t, the number of years from the present when the beneficiary receives the money. There are two main reasons for discounting. According to the social time preference hypothesis, people prefer the benefits in the present because of impatience, risk of death, uncertainty about the future and diminishing marginal utility of consumption (Pearce et al., 1989). Alternatively, people may prefer what they can obtain in the present because they can convert it into productive capital to generate further gains. This represents the social opportunity cost rate of discounting (Pearce et al., 1989) and the notions of substitutability and technological optimism are implicit. Typical discount rates for the assessment of public policy programs range from about 4% to 10% (Henderson and Sutherland, 1996). Discount rates influence levels of investment, rates of natural resource exploitation, decisions involving risk, uncertainty and irreversibility and have an important bearing upon the issue of inter-generational equity. A comprehensive discussion of effects of discount rates on these issues is available in Pearce et al. (1989) and Heal (1998) discusses the alternatives to discounting.

Limitations in valuation methodologies and difficulties in estimating future costs and benefits result in uncertainty about the single point estimates of NPV which need to be conveyed to decision-makers. Uncertainty about key variables in CBA is handled using sensitivity analysis while decisions under uncertainty are addressed using expected value analysis and options analysis (NCEDR, 2001).

Sensitivity analysis investigates uncertainty by changing input variables and observing the change in predictions. It can be done on a variable by variable basis or by altering groups of variables at once, in which case it is known as scenario analysis (NCEDR, 2001). A range of values representing 'optimistic', 'most likely' and 'pessimistic' outlooks may be defined for each variable. This process is useful because it provides a feel for where uncertainties are critical for the analysis. However, this is seldom undertaken in a routine and rigorous manner (Pearce, 1998a).

Expected value analysis deals with risk by assigning probability estimates to alternative outcomes. These probabilities are multiplied by their associated utilities and summed to produce the expected NPV which in effect represents the probability weighted average of the possible outcomes (Morgan and Henrion, 1990). The assumption underlying the expected value is that it is the average value that would be obtained if the project were repeated numerous times (Department of Finance, 1991). Where probability estimates are unavailable, subjective probability estimates have to be developed. Unfortunately, the quality of information underlying the probability estimates is not usually systematically evaluated (NCEDR, 2001). In addition, expected value analysis assumes risk neutrality meaning that the decision-maker places the same weight on gains as on losses. Adjustments may therefore be required in accordance with actual risk attitudes.

Options analysis relies on two main concepts: sequential decision analysis and consideration of irreversibility. Sequential decision analysis involves reformulating the task and subdividing it such that information gained during the early parts of the activity can be used to reduce the uncertainty in the later parts of the activity (NCEDR, 2001). Irreversibility can be technical or economic – the former occurs when technology which can mitigate the negative impacts of the development does not exist, for instance, when nuclear waste is created or a valley is flooded for a hydroelectric dam or a species becomes extinct. Economic irreversibility arises when the costs of remediating the damage are prohibitive even though technologies exist for doing so (Zhao and Zilberman, 1999; Hobbs et al., 2003).

The importance of irreversibility with respect to natural resource projects was first highlighted by Fisher and Krutilla (1974). They argued that irreversible developments result in welfare loss from both forgone benefits and a reduction of options. One partial solution was proposed by Krutilla and Fisher (1975) who suggested that the forgone benefit resulting from environmental loss in future years should be treated as a cost. Furthermore, this cost can be expected to increase over time because the demand for ecosystem services will increase and their supply is likely to be limited.

A consideration of equity and the 'intangibles' (unpriceable costs and benefits) completes the CBA. CBA aggregates costs and benefits across individuals without explicit regard for equity or distribution of costs and benefits between different income groups, ethnic groups, regions and so on. One approach to this issue has been to employ distributional weights. Distributional incidence matrices are constructed to depict potential gainers and losers resulting from the policy. Analysts then attach differential weights to costs and benefits which accrue to particular groups (Department of Finance, 1991; Pearce, 1998a). The treatment of intangibles in CBA is often cursory and recommendations on how to incorporate unpriceable costs and benefits for proper consideration are vague. To illustrate, the Department of Finance's (1991) handbook on CBA devotes about 120 out of its 140 pages to discussions and detailed explanations on calculation of costs and benefits, choice of discount rates and NPV and techniques for risk analysis. In contrast, the instructions given with respect to intangibles simply says to "list and describe as fully as possible".

7. Weighing up the welfare economic approach to decision-making

CBA assumes that societal preferences can be adequately represented by simply aggregating valuations obtained from individuals in isolation. This is a reasonable assumption when the services being valued are purely individually enjoyed and one person's use creates no externality impacts. However, it has been pointed out that it is inappropriate in instances where group values may depend on communal interaction, where preference formation is partially a social process, where shared knowledge is important and where services valued have substantial interpersonal or social implications (Sagoff, 1998; Farber et al., 2002). For example, the value of a forest to a community whose social system and cultural identity is intimately linked to it would be greater than the sum of independent values (Farber et al., 2002). This is akin to the concept of 'emergent properties' in complex ecosystems where the whole may be greater than the sum of its parts. Methodological individualism effectively ignores these emergent features of group preferences. It follows that utility and efficiency do not constitute sufficient bases for decision-making when people aspire towards greater goals such as sustainability and equity with respect to environmental management decisions (Costanza and Folke, 1997).

Expected value analysis predicts the average outcome in the long-run, but is misleading as it seldom applies in most real-world situations where there are few or no repeat opportunities for decision-making. Increasing awareness that ecosystems exhibit multiple stable states and that cumulative effects on the slow variables moderating ecosystem dynamics can result in novel risks, (as exemplified by the discovery of the damaging properties of chlorofluorocarbons), surprising responses and catastrophic regime shifts is leading to greater interest and emphasis on how to make decisions in the face of risk, uncertainty and irreversibility (Chichilnisky, 2000; Wätzold, 2000). More sophisticated tools are needed to meet the challenges of decision-making under these situations.

The analysis of irreversible developments entails considerable uncertainty. What decision rules should then apply? Fisher and Krutilla (1974) advocated a conservative policy with respect to irreversible modification of the environment with the rationale that if society was averse to risk, then there would be value in retaining the scope of future choices, especially when the future demand for environmental services is uncertain. They also suggested that a risk neutral society could retain option value by employing sequential decision analysis to improve later decisions using insight gained from earlier decisions. This notion has a great deal in common with adaptive ecosystem management proposed by Holling (1978). Finally, they argued that even assuming perfect certainty about the costs and benefits of alternative actions, an activity which yields positive returns in the short-run and negative thereafter, and which cannot be terminated should perhaps not be undertaken in the first place. This coincides with the precautionary principle which urges conservative policies and erring on the side of caution when faced with uncertainty and actions with irreversible consequences (Goodland, 1995).

In managing ecological systems, option value may lie in the avoidance of catastrophic changes (Limburg et al., 2002). For systems that are relatively intact, functioning well and resilient, this would mean investing in policies that nurture and maintain resilience (Gunderson, 2000; Walker et al., 2002). In systems that have been substantially altered and are less resilient, this would mean investing in policies that aim to keep these systems away from their critical thresholds.

8. What are the alternatives?

The top-down, technocratic welfare economic approach provides few opportunities for stakeholders to contribute to the decision process of complex and socially contentious problems beyond expressing individual preferences via monetary bids in response to the various valuation methods. Furthermore, there is insufficient emphasis on uncertainty and irreversibility in decisions that have potentially far-reaching and long-lasting consequences. Societal interests may be better served by encouraging citizen science and participatory approaches, which allow for broad-based debates in a more comprehensive manner involving social learning to facilitate problem ownership, value formation, discussion, deliberation, risk assessment, negotiation and reconciliation of interests (Lee, 1993; Irwin, 1995; Sagoff, 1998; Niemeyer and Spash, 2001).

Real-world problems often involve multiple criteria and constraints. In many genuine decision-making situations, neither the criteria nor the alternatives are "given" a priori (Stewart and Scott, 1995; Zeleny, 1998). Criteria and alternatives have to be devised and problems are formulated to represent an "optimal pattern" of interaction between alternatives and criteria. This cognitive equilibrium, termed optimal "pattern matching" may be guided by the decision maker(s) value system which may be based on defined principles, but is also rooted in context and circumstances (Zeleny, 1998).

This suggests a need for frameworks that will allow informed choices by providing opportunities for genuine, substantive participation in decision-making supported by best available scientific knowledge that also incorporates uncertainty in an honest, rigorous and consistent manner. Such approaches should include mechanisms for: (a) articulating visions about what sort of ecosystem services people want; (b) learning about the decision problem; (c) exploring system dynamics and potential outcomes associated with decision options; (d) risk assessment and analysis of uncertainty; (e) facilitating discussion, deliberation and negotiation about trade-offs; and (f) evaluating options in the search for compromise solutions. Techniques such as discourse-based methods (e.g., Citizens' Jury), simulation modelling, probabilistic risk assessment (PRA), multi-criteria decision analysis (MCDA) and scenario planning provide these mechanisms, but as each tends to focus only on one or two of these requirements, these techniques may need to be used in concert.

Participatory and discourse-based approaches in natural resource decision-making are aimed at achieving wider community understanding, social equity and greater legitimacy for policies (Wilson and Howarth, 2002; Proctor and Drechsler, 2003). The basic idea is that small groups of citizen-stakeholders (playing an implicit role as a social decision-maker) can be brought together to deliberate the decision problem in a setting that ensures procedural fairness and fosters joint discovery. The outcomes derived from this forum may then be used to guide policy. By implementing a fair and openly structured procedure for deliberation, it is assumed that small groups of citizens can structure, learn, articulate and debate preferences for alternative options to render informed judgements about public decisions that take into account broadly held social values (Sagoff, 1998; Wilson and Howarth, 2002).

It may be reasonably assumed that for most of us, the cognitive burden imposed by attempting to grapple with ecosystem dynamics influenced by stochasticity, interconnectivity, nonlinear interactions and spatial and temporal lags in ecosystem responses is overwhelming. Modelling is a particularly effective strategy for confronting this challenge. Computer models play diverse roles in ecosystem management and can vary in degree of detail and complexity. Simple models can be extremely useful for illustrating general patterns of system behaviour. When these models are usable and understandable by diverse participants and easily modified to incorporate novel situations, they provide a powerful tool for engaging stakeholders in learning about problem structure and system dynamics in response to decision options (Costanza and Ruth, 1998; Carpenter et al., 1999).

Carpenter et al. (1999) likened the role of such models to the role of metaphor in narrative, as tools to be "used as caricatures of reality that spark imagination, focus discussion, clarify communication and contribute to collective understanding of problems and solutions". Examples include integrated social-economic and ecological models of lake eutrophication by non-point pollution from land management practices (Carpenter et al., 1999; Janssen and Carpenter, 1999) and of lake fisheries management under anthropogenic impact (Carpenter and Gunderson, 2001).

In this context, the role of models and PRA is to cross-examine ideas and to ensure conceptual models are consistent with data and ecological theory. Quantitative models formalize understanding about the relationships under consideration in the system (e.g., cause–effect, dose–response, exposure–effect). Uncertain driver variables may be identified and their distribution parameters and correlations with other uncertain variables estimated. These distributions are input into the model and used to produce probabilistic estimates of the range of possible outcomes. These operations are too complex to be intuitive and the models ensure internal consistency. PRA has been applied in a number of disciplines such as engineering, economics, ecology, ecotoxicology and public health. In conservation biology, for instance, population viability analysis (PVA) models have been used in the risk assessment of various species (Burgman et al., 1993). However, they have rarely been applied in a subordinate role to support community decision-making.

MCDA techniques originated over three decades ago in the fields of mathematics and operations research and are well-developed and well-documented (e.g., see Hwang and Yoon, 1981). They provide a structured framework for decision analysis which involves definition of goals and objectives, identification of the set of decision options, selection of criteria for measuring performance relative to objectives, determination of weights for the various criteria and application of procedures and mathematical algorithms for ranking options. Criteria are scored on interval or ratio scales and then transformed to ensure commensurability before algorithms based on value or utility functions, goal programming, outranking or descriptive/multivariate statistical methods are applied to rank the options (Howard, 1991; Stewart, 1992). The method is well-suited to eliciting values and preferences and evaluating stakeholder interests.

Proper application of MCDA requires sound understanding of how the description of objectives, number of criterion used per objective, procedure for elicitation of weights, choice of scale transformation technique, and criterion weights interact to affect the resultant ranking of options (Howard, 1991). Sensitivity analysis can be used to explore the impact of some of these interactions (e.g., see Proctor and Drechsler, 2003). Important considerations in the choice of a ranking procedure are discussed in Stewart (1992) and Drechsler (2004).

Traditional MCDA assumes that there is a single decision-maker such that clear, unambiguous, nonconflicting objectives can be identified from a single perspective. Furthermore, it is assumed that the relevant criteria are well-defined, independent of each other and can be measured with certainty (Stewart, 1992; Fenton and Neil, 2001). In order to extend MCDA to group decision situations where there might be conflicting objectives and to incorporate uncertainty into the decision-making process, MCDA needs to be used in conjunction with discursive participatory methods and modelling tools such as those used in PRA. Tools such bayesian belief networks (BBN) may also be useful for taking uncertainty into account in MCDA (Fenton and Neil, 2001).

Fernandes et al. (1999) provide an example of MCDA in a participatory setting for coral reef management in Saba Marine Park, a Caribbean island in the Netherlands Antilles. The process provided a forum for tabling, discussing and documenting the community's concerns and allowed the unexpected degree of general agreement to become apparent. In this sense, it facilitated social discourse, value formation and learning about the interactions of the social, economic and ecological system. An approach combining a Citizens' Jury with MCDA to examine options for management of tourism and recreation activities in the upper Goulburn Catchment in Victoria, Australia was also found to be useful for overcoming the shortcomings of each used in isolation (Proctor and Drechsler, 2003). Drechsler (2004) provides an example of how goal conflicts and uncertainty may be incorporated into MCDA by using sensitivity analyses on a formal model to estimate the impacts of actions on decision criteria.

Scenario planning is a disciplined method for imagining and depicting possible futures where prediction is not possible (Schoemaker, 1995). Peterson et al. (2003) advocated its use in conservation planning and provides an overview of the technique. It is well suited for use in a participatory context and provides a comprehensive framework for exploring decision-making in situations where uncertainty is high and the system under consideration is difficult to control. Qualitative and quantitative knowledge are incorporated into scenarios along with a strong emphasis on uncertainties (particularly those which are uncontrollable) in order to provide insight into drivers of change, reveal implications of current trajectories, expose possibilities for surprise, challenge conventional thinking and illuminate options for actions (Carpenter, 2002; Peterson et al., 2003). Scenario planning considers scenarios in sets of three or four that collectively represent an instructive range of ambiguous and uncertain outcomes (Carpenter, 2002). Scenario stories present the range of possibilities in tangible, evocative descriptions about alternative futures to stimulate thinking. They can be used to examine how existing policies would fare in different scenarios and to identify policies that perform well in all scenarios, in other words, robust policies that produce acceptable consequences regardless of how events unfold (Harwood, 2000; Carpenter, 2002; Peterson et al., 2003). Perceptions of attainable futures and the process of reflection about the sort of futures people want (or conversely, seek to avoid) also impels people to assess their role in creating such futures (Costanza, 2000; Walker et al., 2002). All this may stimulate further questions or suggest novel policies or areas for research and issues to monitor (Peterson et al., 2003).

9. Conclusion

As the examples of the Catskills Mountains and fynbos ecosystems demonstrate, ecosystem service valuation has the potential to inform policy decisions by highlighting the benefits of sustainable ecosystem management. However, the techniques used for valuation suffer from serious limitations and in addition, many ecosystem services are simply not amenable to valuation by the techniques available. Ecosystem management problems are often complex, multi-faceted, socially contentious and fraught with uncertainty. As we have seen, the welfare economic approach to decision-making is too narrowly focussed and does not take adequate account of uncertainty and irreversibility. It is therefore imperative to consider more comprehensive approaches for facilitating genuine, substantive stakeholder participation with opportunities for social learning, value formation, problem exploration, risk assessment, analysis of uncertainty, broad-based debate and reconciliation of interests. This involves utilizing a broader range of tools such as Citizens' Jury, simulation modelling, PRA, MCDA and scenario planning. Mathematical representations and models of ecological processes capture our understanding of ecological systems in a rigorous and consistent manner and can be used to help stakeholders explore management options in a participatory context. In this sense, they play a critical but subordinate role in the collective decision-making process. Tools such as PRA and BBN allow uncertainty to be incorporated and analysed in a rigorous and explicit way and MCDA provides a transparent method for evaluating options in the search for compromise solutions. And finally, scenario planning may be effective in situations where system control is difficult and uncertainty is great.

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