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Cost-benefit analysis in the context of ecosystem services for human well-being: A multidisciplinary critique

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ABSTRACT

This paper provides a critique of the cost-benefit analysis tool for ecosystem services policy evaluation. We argue that when applied to public ecosystem services, the theoretical assumptions that underlie economic valuation and cost-benefit analysis fail to fully acknowledge the multiple dimensions of human well-being, the plural forms of value articulation, the complex nature of ecosystems, the distributional biases of markets and the fairness implications of spatio-temporal framing. The current monistic utilitarian approach to ecosystem services policy evaluation should therefore be replaced by a pluralist framework composed of a heterogeneous set of value-articulating instruments that are appropriate to the specific context within which decision-making takes place. It is argued that within this pluralist framework cost-benefit analysis may remain an appropriate tool to examine the contingent trade-offs of local policies that have limited impacts on ecosystems and their services.

1. Introduction

As the scale of most economies continues to grow, demands on ecosystems increase too, so that all demands cannot always be fulfilled simultaneously, and trade-offs across ecosystem services, among beneficiaries and between time periods are expected. To choose among policies that involve such trade-offs can be a complex and daunting task, and cost-benefit analysis is often considered an effective tool to guide this choice (Pearce et al., 2006; Daily et al., 2009). Cost-benefit analysis is increasingly being used to evaluate projects and policies that affect ecosystem services, and to guide the selection of projects that deliver maximum net benefits from the flow of these services to society (Carpenter et al., 2009; Daily et al., 2009).

In its application to ecosystem services, cost-benefit analysis is informed by the ecosystem services framework, a metaphor of nature that conceives of the environment as natural capital, and of natural processes as leading to a stream of ecosystem services that flow like 'interest' from the natural capital to society (Costanza et al., 1997; Turner et al., 2003; Norgaard, 2010). In this sense, ecosystem services consist of life-supporting services (e.g. the recycle of nutrients, the assimilation of waste, and the regulation of climate, watershed and pests/diseases) and provisioning services (e.g. water flow, domestic crops and livestock, and wild plants and animals). The combination of these services with human-related assets (e.g. man-built infrastructures, knowledge, networks) leads to the realization of those tangible and intangible benefits that are directly experienced by humans, such as food, hydroelectric power, a stable climate, and psycho-physical equilibrium (MA, 2003; Fisher et al., 2008). The ecosystem services framework is used in a range of policy applications, including the construction of macroeconomic indicators of sustainability, the evaluation of environmental policies, and the formulation of market-based incentives for conservation. Cost-benefit analysis is used specifically to evaluate the economic efficiency of alternative policies that impact on ecosystem services.

Cost-benefit analysis compares alternative policy options by quantifying their impacts on the flow of ecosystem service through a monetary metric, identifying positive changes in such flow as benefits and negative changes as costs. The obtained monetary values are then aggregated to calculate the total net benefit of each policy option in terms of net present value (NPV). Policies that pass the Kaldor-Hicks compensation test, i.e. NPV > 0, are then ranked by their NPVs, and those with the highest NPV are selected as they are assumed to bring about the largest increase in social welfare (Pearce et al., 2006).[1] The use of money (or willingness to pay) as a measure of value indicates that cost-benefit analysis is firmly

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[1] A policy is said to pass the Kaldor-Hicks compensation (or potential Pareto improvement) test when the beneficiaries can hypothetically compensate the losers and still have some net gains left over, i.e. when the NPV of that policy is positive.

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rooted in neoclassical economic theory, which conceive of human well-being in terms of utility (or preference satisfaction), and focus on the price (or exchange) value of goods and services (Vatn, 2005; Farley, 2008; O'Neill et al., 2008; Gómez-Baggethun et al., 2010). As such, cost-benefit analysis is rooted in *ethical individualism* (or welfarism), according to which social welfare should be assessed on the basis of the individual utility of society's members (Sen, 1985; Accocella, 1998), and personal preferences are revealed through market-related choices, either directly through existing markets or indirectly through complementary and hypothetical markets (Vatn, 2005; Pearce et al., 2006).

The ecosystem services framework achieved its popularity climax with the release of the Millennium Ecosystem Assessment (MA, 2003), a comprehensive assessment of the world's ecosystem services and their implications for human well-being, and such popularity provided renewed scientific and political thrust for ambitious environmental cost-benefit analysis ventures. These include for example the Stern Review on the Economics of Climate Change (Stern, 2006), which attempted to highlight the economic costs of ecosystem services decline caused by climate change, and 'The Natural Capital Project' (Tallis and Polasky, 2009), which has developed a software for 'Integrated Valuation of Ecosystem Services and Tradeoffs' (InVEST) to assess the impacts of ecosystem change in several regions of the world. In its original conception, the global initiative 'The Economics of Ecosystems and Biodiversity' (TEEB, 2008) also planned to carry out cost-benefit analyses of the ecosystem services decline caused by biodiversity loss. However, the aims of TEEB have been recently reformulated so as to acknowledge the limits and complexities involved in this practice (TEEB, 2010).

In light of these far-reaching developments, the aim of this paper is to provide a comprehensive review of the theoretical validity and empirical applicability of cost-benefit analysis in the context of ecosystem services. We aim to bring clarity and synthesis to the rich and often disconnected set of criticisms made to cost-benefit analysis from a broad range of disciplines, and to review the arguments advanced in its defence, in order to facilitate a consensus about the applicability of cost-benefit analysis to ecosystem services policy evaluation. We argue that ecosystems and their relation to human well-being have certain characteristics that challenge some of its theoretical assumptions, and that consequently under certain circumstances cost-benefit analysis can generate inaccurate estimates leading to misguided policy decisions. Our view is that it should be used with judgement, and as only one among a set of potential tools for environmental policy evaluation.

The paper is structured as follows. The next two sections contain a discussion of the neoclassical economics paradigm that underlies cost-benefit analysis, the axiomatic assumptions of value articulation and value aggregation being scrutinised in Sections 2 and 3, respectively. The choice of spatio-temporal frames is discussed in Section 4. Section 5 examines the main arguments developed in defence of the use of cost-benefit analysis in policy-making. This overall discussion leads us to address the need for a pluralist approach to environmental policy-making in Section 6. We conclude by highlighting the key points raised in the paper.

2. Ecosystem services and value articulation

Since the first formulation of cost-benefit analysis in the 19th century, a range of criticisms have been made to its normative and theoretical foundations. Cost-benefit analysis has been criticised because its normative focus on subjective utility, rather than actual functionings, is an inadequate indicator of human well-being and social welfare (Sen, 1985); because the hypothetical nature of the Kaldor-Hicks criterion, and the principle of separation between efficiency and distribution goals that underlies it, overlooks the actual distribution of policy outcomes (Little, 1949); because the Kaldor-Hicks criterion is afflicted by a range of theoretical inconsistencies that undermine the validity of the analysis (Chipman and Moore, 1978; Gowdy, 2004, 2007)[2]; because the aggregation rule used in the social welfare function is, unavoidably, arbitrarily selected among an infinite number of such functions (Pearce et al., 2006); and because it is based on price information that is easily distorted by externalities, imperfect forms of competition, subsidy schemes, distributional biases and imperfect flow of information.

The use of cost-benefit analysis in the context of ecosystem services poses further challenges, since public ecosystem services have characteristics that challenge the neoclassical economics theories of value articulation and value aggregation that underlie cost-benefit analysis. More specifically, the theoretical assumptions of neoclassical economics are inadequate when dealing with public ecosystem services that (i) affect intangible dimensions of human well-being, (ii) are intrinsically and collectively valued, (iii) are characterized by thresholds, complexity and uncertainty, (iv) differently affect poor and wealthy groups in society, and (v) are endogenously valued. In these circumstances, willingness to pay may represent an invalid measure of the value of ecosystem services, as it may fail to represent the actual values that people hold towards many of these services. We discuss each of these circumstances in turn (refer to Table 1).

2.1. Plural dimensions of human well-being

Welfare economics and cost-benefit analysis are intrinsically utilitarian and welfarist, i.e. they conceive of human well-being in terms of the satisfaction of personal preferences for goods and services, and assess social welfare on the basis of individual utility (Sen, 1985). However, increasing evidence shows that personal well-being can greatly surpass the notion of utility, as it may consist in the realization of a broader range of capabilities instead (Kahneman et al., 1997; Sen, 1999; Frey and Stutzer, 2002). In fact, results from environmental psychology confirm that ecosystems have relevance to human well-being far beyond the satisfaction of preferences, including a strong bearing on psychological health, social integration and socio-cultural identity. For example, Chiesura and De Groot (2003) review experimental studies that show that hospital patients have a faster and less painful recovery when assigned to rooms looking out on a natural scene, and that urban nature generates feelings of freedom, peace and beauty that are believed to reduce stress and restore the psycho-physical equilibrium. Coley et al. (1997) show that nature encourages the use of outdoor spaces, increasing social interaction and integration among neighbours, and O'Neill et al. (2008) stress the value of landscapes as carriers of common historical heritage and cultural identity.[3] The obstacle for cost-benefit analysis is that these intangible values of nature belong to the cognitive and emotional realm of human beings and as such they are hardly quantifiable in terms of preference satisfaction and willingness to pay (Chiesura and De Groot, 2003; Kumar and Kumar, 2008). It follows that the net benefit calculations used in cost-benefit analysis may be incomplete, as they may not account for the impact

[2] Such inconsistencies include the so-called 'Scitovsky paradox' (Scitovsky, 1941), the 'Boadway paradox' (Boadway, 1974), the 'second best' effect (Lipsey and Lancaster, 1956-7), and the fact that it cannot be proved that more is preferred to less (Samuelson, 1950).

[3] Some authors suggest that these intangible benefits of nature may still be valued in monetary terms through revealed preference methods. For example, health benefits can be quantified in terms of reduced public expenditures for health care, stress reduction benefits can be measured through increased productivity at work, and the heritage value can be calculated through the donations made to environmental organizations (Chiesura and De Groot, 2003). However, these methods are inadequate if we accept the hypothesis of motivational pluralism (Paavola and Adger, 2005) (see section 2-viii).

Table 1
Neoclassical economics and alternative theories of value (or preference) articulation and aggregation.

	Neoclassical and welfare economics	Alternative theory	(Proxy) discipline	Authors cited
Theories of value articulation	Well-being is uni-dimensional–it consists in the optimisation of utility	Well-being is pluri-dimensional–it consists in the fulfilment of plural capabilities (including psychological, social and cultural)	Institutional economics (e.g. capability approach)	Kahneman et al. (1997) Sen (1999)
			Environmental psychology	Chiesura and De Groot (2003) Kumar and Kumar (2008)
			Environmental philosophy (and cultural anthropology)	O'Neill et al. (2008)
			Institutional economics	Vatn (2005)
	Value is utilitarian - total economic value (TEV)	Value can be intrinsic	Environmental philosophy (and cultural anthropology)	Sagoff (1988) Aldred (2006) O'Neill et al. (2008) Spash (2008)
	All values/preferences exist ex ante and can be elicited by individuals in isolation (methodological individualism)	Some values/preferences need to be socially constructed through collective communication	Political philosophy Institutional economics	Dryzek (2000) Vatn (2005) Howarth and Wilson (2006)
	Marginal and relative value	Ecological non-linearity	Ecological sciences	Baumgärtner et al. (2006) Farber et al. (2002) Pearce et al. (2006) Farley (2008)
	Partial equilibrium models	Ecosystem complexity	Ecological sciences	Daily et al. (2000) Neumayer (2003) Wallington et al. (2005) Brander et al. (2007) Farley (2008)
		Ecosystem resilience	Ecological sciences	Chee (2004) Arrow et al. (2008)
	Omniscience	Ignorance and uncertainty	Ecological sciences	Chee (2004) Carpenter et al. (2009)
			Environmental philosophy (and cognitive psychology)	Sagoff (1988) Spash and Hanley (1995)
	Prices are a neutral measure of values/preferences	Prices are influenced by the distribution of wealth	Institutional economics	Bromley (1991) Sen (2000) Martínez-Alier (2002) Vatn (2005) Spash (2008)
	Other-considering values/preferences are an expression of personal utility	Prices are influenced by resigned adaptation	Moral philosophy (and behavioural psychology and neuroscience)	Sen (1999) Nussbaum (2000) Teschl and Comim (2005)
		Prices are influenced by status-seeking	Behavioural psychology and neuroscience	Frank (1985) Frey and Stutzer (2002)
		Other-considering values/preferences can be motivated by deontological ethics and religious views	Institutional economics Institutional economics	Brekke et al. (2003) Etzioni (1988) Vatn (2005) Aldred (2006)
Theories of value aggregation	All values/preferences are exogenous	Values/preferences can be endogenous	Social psychology	Sen (1985) Dittmar (1992)
	All values/preferences are commensurable	Values/preferences can be lexicographic	Moral philosophy Institutional economics	Sagoff (1988) Harsanyi (1955) Sen (1977) Brekke and Howarth (2000) Nyborg (2000) Vatn (2005)
			Moral philosophy	Sagoff (1988)
	All preferences are commensurable with money	Lexicographic preferences are incommensurable with money	Institutional economics	Spash and Hanley (1995) Rekola (2003) Gowdy (2004)
			Moral philosophy	Sagoff (1988) Aldred (2006) O'Neill et al. (2008) Spash (2008)
	Values/preferences are static	Values/preferences are dynamic	Institutional economics	Vatn (2005) Gowdy (2007)
	All values/preferences are complete, i.e. can be aggregated on a single monetary scale of measure	Endogenous and lexicographic values/ preferences are incomplete, i.e. they resist being aggregated on a single monetary scale of measure	Moral philosophy Institutional economics	O'Neill et al. (2008) Sen (1977) Brekke and Howarth (2000) Nyborg (2000) Niemeyer and Spash (2001) Vatn (2005)
	All values/preferences can be measured on a cardinals scale and compared across individuals	The values/preferences of heterogeneous groups of people cannot be measured on a cardinals scale and compared across individuals	Theoretical welfare economics	Harsanyi (1955) Gowdy (2007)

of environmental policies on these important dimensions of well-being.

2.2. Altruistic and intrinsic values

Implicit in neoclassical economics is the assumption that the value that people attach to ecosystem services is 'consequentialist' and 'self-interested'. In reality, human behaviour can encompass also choices informed by consequentiality concerns for the well-being of other human and non-human beings (Sagoff, 1988), as well as choices informed by deontological motivations without regard to any of the consequences (Paavola and Adger, 2005). Economists have tried to broaden the economic notion of value through the concept of total economic value (TEV), which includes other-considering forms of value–altruistic and bequest values are held towards goods and services from which other humans may benefit in the present and future time respectively, and existence values are held towards elements of nature that no person may ever benefit from (Pearce and Turner, 1990; Heal et al., 2005). However, it can be argued that TEV remains a consequentialist and 'self-interested' framework, as it still treats other-considering values as an expression of personal utility in the form of a 'warm glow' or 'moral satisfaction' from knowing that others are made better off or that an element of nature exists (Andreoni, 1990). This consequentialist characterisation of other-considering values remains necessary if these values are to be measured in terms of willingness to pay and included in cost-benefit analysis.

The drawback of the TEV concept rests in the fact that not all other-considering values are motivated by self-interested desires of moral satisfaction, but on the contrary these values are often regarded as intrinsic to an entity independently from any personal satisfaction (Etzioni, 1988; Nyborg, 2000). For example, altruistic forms of value may be associated with fundamental ecosystem services such as water regulation and climate stabilisation on the basis of deontological principles, i.e. the belief in people's rights to a certain quality of life and freedom of choice. Similarly, elements of nature such as a totemic species or a natural landscape may be invested with intrinsic value because of a person's cultural/historical identity (e.g. ties with particular places) and cosmovision/religion (e.g. Buddhism and Hinduism). The hindrance for cost-benefit analysis is that intrinsic values exhibit monetary incommensurability, i.e. individuals are unable and often refuse to measure them along the scale of money (Sagoff, 1988; Spash, 2006; O'Neill et al., 2008). As a consequence, like the psycho-cultural dimension of well-being, any intrinsic form of value that may be attached to nature remains excluded from cost-benefit analysis.

2.3. Collective value formation

Welfare economics and cost-benefit analysis endorse *methodological individualism*, which assumes that individuals hold their values (or preferences)[4] in advance and can elicit them in isolation (Accocella, 1998). However, it has been highlighted that the benefits of ecosystem services and their anthropogenic change are complex and collective processes that cannot be comprehended and valued individually (Sagoff, 1988). Consequently, environmental values may not be held in advance, needing to be discovered or elicited, but rather need to be socially constructed through some form of deliberative communication (Dryzek, 2000).

The importance of collective deliberation to the formation of preferences for public goods is empirically supported by studies on the social psychology of small group decision-making, which show that when effective group procedures enable participants to access relevant information, share their perspectives and engage in collective thinking, participants make more informed and articulated group choices than when preferences are aggregated from isolated individuals (Howarth and Wilson, 2006). Hence, it is debatable whether the aggregation of individual preferences constitutes a reliable basis for decision-making when public ecosystem services are at stake.

2.4. Non-linearity and irreversibility

As generally conceived by neoclassical economics, the value that people attach to an object is 'marginal' and 'relative', that is, determined by the value of the last unit used and by its scarcity compared to other substitutable commodities or income (see e.g. Baumgärtner et al., 2006). Linked to the notion of relative scarcity are those of opportunity cost and willingness to trade-off (or give up something in exchange). That is, depending on its scarcity relative to other valuable commodities, cost-benefit analysis assumes that the marginal unit of a good or service is traded-off for something else that determines its value. The willingness to trade-off is usually defined relative to a monetary budget as willingness to pay (WTP) or willingness to accept (WTA), so that money generally becomes the measure of value.

This economic concept of value presents some limitations when applied to ecosystem services, since the latter are not typical commodities, but instead are characterised by limited degrees of substitutability, non-linearities and critical thresholds that imply that they might not always change in marginal ways. When an ecosystem is approaching a critical threshold only minor disturbance may alter the ecosystem in a dramatic and often irreversible manner, so that some of the services it provides may become scarce in absolute rather than relative terms, i.e. if these services are fundamental to the satisfaction of critical human needs and rights, then their further loss can no longer be compensated through trade-offs with other goods and services (Farber et al., 2002; Baumgärtner et al., 2006). Examples include the pollution of shallow lakes (Scheffer, 2001) and coral reefs (DeYoung et al., 2008), land and river salinity problems (Chee, 2004), and burning and shifting cultivation in tropical forests (Wegner et al., 2009). Therefore, when a critical threshold is approached and fundamental ecosystem services become scarce in absolute terms, the marginal WTP for these services may be expected to rise dramatically, perhaps to infinity. However, according to economic theory, the level of WTP cannot be infinite, as it is bounded by the level of available income (Pearce et al., 2006; Farley, 2008). Consequently, in the region of a critical threshold for ecosystem service provision economic valuation becomes meaningless and cost-benefit analysis is of limited applicability (Farley, 2008).[5,6]

Recently there have been some efforts to deal with ecological non-linearity in ecosystem service valuation, for instance by modelling non-convexities of coupled ecological-economic systems (Dasgupta and Mäler, 2004), or by measuring the 'insurance' service provided by a natural asset when close to a threshold (Farber et al., 2002; Walker et al., 2010). However, the high complexity that characterise ecosystem processes (Fisher et al., 2008; Norgaard, 2010) and the inapplicability of economic

[4] In the rest of the paper the terms 'value' and 'preference' are used interchangeably.

[5] In theory, compensating and equivalent surplus measures could be used to value non-marginal environmental change (Freeman, 1993). However, in the context of complex ecosystems generally characterised by non-linearities and irreversibility, economic valuation is better suited to handle marginal ecosystem change far from ecological thresholds.

[6] The technical incompatibility between ecological thresholds and economic valuation also explains why attempts to determine the total economic value of the world's ecosystem services (e.g. Costanza et al., 1997; Balmford et al., 2002) are meaningless – the absolute economic value of ecosystem services would simply be infinite, and this information is of little practical usefulness (Pearce et al., 2006).

valuation under threshold conditions imply that these theoretical developments may never be confidently included into cost-benefit analysis.

2.5. Complexity and resilience

Ecosystems are complex and highly interconnected, and are endowed with some degree of in-built resilience (Wallington et al., 2005). When valuing changes in ecosystem services, it would be important to account for these properties of ecosystems in order to enhance the accuracy of estimated values. In fact, the interdependence of several ecosystem functions implies that even minor changes in one component of an ecosystem can have large impacts on other components and change the provision and thus relative value of its services (Daily et al., 2000). The larger is the ecosystem change analysed, the stronger is the co-variation between ecosystem functions and ecosystem service values. Accounting for these interdependences would require the use of general equilibrium models, whereby the economic values used in the analysis readily co-vary. However, these models necessitate a large amount of data and rely on a series of highly questionable theoretical assumptions (Neumayer, 2003). Hence, in practice, partial equilibrium models remain the norm in cost-benefit analysis and ecological interdependency tends to be ignored (Norgaard, 2010).

The incorporation of the property of ecological resilience *sensu* Holling (2001) in the economic valuation of ecosystem services is also important. When resilience is eroded, disturbance may cause an ecosystem to shift into a new stable state characterized by a different set of structures and functions. This means that changes in this property under different policy regimes may affect the level of provision of an ecosystem service, and therefore its marginal value. However, accounting for changes in this crucial property when measuring and valuing ecosystem services is still arduous, because our knowledge of the relationship between ecosystem components and resilience is still insufficient (Chee, 2004; Arrow et al., 2008; Pascual et al., 2010).

2.6. Ignorance and uncertainty

Another key assumption of neoclassical economics is that humans are omniscient actors; that is to say, we have complete information and perfect understanding of our set of choices, and hence we can always form preferences over goods and services. However, research in cognitive psychology suggests that people can often be ignorant, or uncertain at best,[7] about the nature of the decisions they face (Chee, 2004; Kumar and Kumar, 2008). This is accentuated in decisions regarding ecosystem services, where ignorance and uncertainty can characterise both stages of biophysical quantification by scientists and economic valuation by citizens. In biophysical terms, ecosystem services stem as the result of complex interactions between biotic and abiotic processes over a wide range of spatio-temporal scales, and knowledge of the cause-effect linkages, thresholds, irreversibility, resilience and spatial idiosyncrasy of these interactions is often scarce or non-existent. As a consequence, estimation of the relationship between ecological functions and the flow of ecosystem services is very complicated (Daily et al., 2000; Carpenter et al., 2009). Even when biophysical processes can be confidently measured, their economic valuation by citizens can be affected by ignorance and uncertainty, because the latter may not have sufficient understanding of and experience with ecosystem services (Sagoff, 1988; Spash and Hanley, 1995). For example, the disaster-mitigation value of coastal mangroves became widely evident only after the crushing tsunami of 26 December 2004 (Danielsen et al., 2005).

Some scholars have attempted to deal with uncertainty about biophysical processes through the calculation of a real-option value, which is given by the change in net benefits that would be obtained from a decision if that decision was postponed until further knowledge about its impacts is obtained (Arrow and Fisher, 1974). However, it remains unclear how this value is to be calculated in practice, given that the benefits and costs of postponing a decision until further knowledge is obtained are not known in advance (Vatn, 2005). Another mechanism to deal with uncertainty about biophysical processes is to use probabilistic methods, which consist in assigning a probability distribution to the biophysical process and deriving the expected value of the service it generates (Pascual et al., 2010). However, this approach is not helpful when the studied ecosystem approaches an ecological threshold, because in this case the potential outcomes of an ecosystem change and their probability distribution are hardly known–i.e., in the region of an ecological threshold science is afflicted by ignorance. In this case, the safe minimum standard (SMS) rule is often presented as a useful alternative to cost-benefit analysis. The SMS rule focuses on the physical stock of natural capital rather than on the flow of ecosystem services. It prescribes aversion from the reduction/degradation of resources below minimum levels that are safely distant from uncertain thresholds, unless the social opportunity costs of their conservation are 'unacceptably large' (Ciriacy-Wantrup, 1968). Unfortunately, the calculation of SMS is hampered by significant problems of data collection and analysis, as well as by inter-temporal inconsistencies (Hanley, 2000) and lack of clarity with respect to what constitutes 'unacceptably high' costs (Neumayer, 2003). It therefore seems that a satisfactory mathematical model to guide decisions under radical uncertainty of ecological thresholds is still to be devised.

2.7. Market prices and inequality

When natural resources are exchanged in the market and prices exist, valuation methods based on prices are considered the most reliable, because prices are taken to directly reflect the preferences of individuals and the scarcity of resources (Pearce et al., 2006). But market prices are not as valid indicators of people's preferences as purported. Markets may be distorted by the presence of environmental externalities, imperfect forms of competition and imperfect flows of information, in which case they are not an accurate reflection of consumer preferences and resource scarcity. Even when markets are not distorted, prices may still not be considered a neutral and accurate reflection of consumer preferences, because instead they are determined also by the distribution of wealth in society (Martínez-Alier, 2002). Consequently, decisions based uniquely on market prices tend to be biased towards the preferences of the wealthiest groups and to perpetuate distributional disparities (Bromley, 1991; Sen, 2000; Spash, 2008). This is particularly relevant in the context of ecosystem services, because the budget constraints of the poor may result in an underestimation of those ecosystem services that are crucial to the long-term sustainability of their livelihoods. For example, less utilised agricultural crops are generally considered of minor importance because they fail to command sufficient market prices when compared to major agricultural crops, and therefore they tend to attract less investment in research and development. Nonetheless, these crops are often far from minor in the lives of the rural poor, for whom they constitute an important source of micronutrients and health remedies, as well as a crucial resource for

[7] The term 'uncertainty' indicates a situation where the possible consequences of a decision can be fully enumerated and their probability distribution may or may not be known. The term 'ignorance' (or 'radical uncertainty'), on the other hand, refers to a situation where the possible consequences of a decision cannot even be enumerated (Pascual et al., 2010).

adaptation to marginal ecosystems, low-input agriculture and climate change (Padulosi et al., 2002).

The unequal capacity of poor and rich people to express their preferences for ecosystem services through prices can be further exacerbated by two aspects of human behaviour: 'resigned adaptation'[8] and 'status seeking'. A form of resigned adaptation is the development of 'adaptive' or 'resigned' preferences by disadvantaged groups, i.e. preferences that they adjust to their deprived circumstances when there is no prospective of escaping them (Sen, 1999; Nussbaum, 2000). People who have persistently lacked access to certain basic ecosystem services such as clean air and water, or to intangible services such as education and recreation through nature, may show little or no willingness to pay for these services, either because they are unaware of their potential effects on their lives, or because they consider them out of their reach. As a consequence, decisions based on market prices tend to continue under-provide the poor with these important ecosystem services. For example, cost-benefit analysis may indicate the development of cheap housing in an unhealthy urban environment (e.g. an area polluted by hazardous disposal and industrial facilities) as an optimal choice on the basis of the willingness to pay of low-income households who have adapted their preferences to cheap but unhealthy housing, resulting in the under-provision of these households with clean air and water (Fisher et al., 2006).

For what concerns status-seeking behaviour, this has been studied in moral philosophy, behavioural psychology and institutional economics. There is ample evidence that people are concerned not only with their absolute levels of consumption, but also with their relative status in comparison with others in their peer group or society at large (Frank, 1985; Frey and Stutzer, 2002). Consequently, the willingness to pay for ecosystem services is a function not only of utility and budget constraints, but also of efforts to maintain and/or increase a favourable self-image with respect to peers. What is relevant to our discussion is that status-seeking behaviour may accentuate the divergence between the prices afforded by the rich and those afforded by the poor, because the market prices of 'positional' goods of wealthier groups are generally higher than those of poorer groups (Brekke et al., 2003). For example, status-seeking behaviour may accentuate the discrepancy between the prices paid by wealthy tourists for a fashionable safari experience in a nature reserve, and those afforded by poor local communities for other uses of the same reserve (Brooks, 2005). Cost-benefit analysis based on these prices is therefore likely to indicate tourism, and the exclusion of local people from other competitive uses, as the optimal land-use choice for that reserve.

Finally, it has been noted that the aggregation function generally used in cost-benefit analysis, which assigns the same weight to all individual costs and benefits, is unable to account for the hypothesis of decreasing marginal utility of income (Pigou, 1920), according to which the same change in income causes higher benefits and costs to the poor than to the rich. In fact, according to this hypothesis a policy alternative that cost-benefit analysis indicates as able to deliver positive net benefits, i.e. a policy that passes the Kaldor-Hicks compensation test, may nonetheless fail to deliver an improvement in total social welfare if costs tend to fall on the shoulders of the poorest members of society, because the reduction in utility of the poor may actually counterbalance the increase in utility of the rich (Accocella, 1998).

Distributionally weighted cost and benefit calculations (e.g. higher weights on costs and benefits occurring to socially disadvantaged groups) have been advocated as a potential way to address equity distortions in cost-benefit analysis (e.g. Anthoff et al., 2009; Pascual et al., 2010). However, so far it has proved difficult to develop a consensus for the formulation, quantification and application of such weights. This is because there are practical difficulties as well as theoretical controversies as one moves from the basic identification of how costs and benefits are distributed to the actual assignment of weights on the basis of judgements about society's distributional preferences (Kriström, 2006; Pearce et al., 2006; Turner, 2007).

2.8. Valuation through cost-based, revealed and stated preference methods

Since the economic value of a public ecosystem service is not signalled by market prices, indirect valuation methods have been developed to estimate it, either by looking at the preferences of consumers for other marketed goods that are linked to the ecosystem service to be valued (revealed preferences methods), or by estimating the costs that would be incurred if the service were degraded and needed to be either restored or recreated artificially (cost-based methods), or by hypothesizing an artificial market for it (stated preference methods). Unfortunately, these valuation approaches are not without difficulties.

Revealed preference methods such as hedonic pricing and travel cost approaches can be problematic if we accept the assumption of motivational pluralism that underlies human choices. Motivational pluralism implies that choices related to public ecosystem services (e.g. climate regulation and biodiversity maintenance) may not be informed by self-interested utility considerations alone, but may be governed also by the fulfilment of other-considering and deontological motivations. In these cases, self-interested utilitarian preferences for related commodities (e.g. housing location and transport to environmental amenity locations) cannot be considered an adequate indicator of the values attached to a public environmental good (Paavola and Adger, 2005). In addition, revealed preference methods continue to pose significant technical obstacles (see e.g. Daily et al., 2000; Pearce et al., 2006).

Cost-based valuation approaches include avoided, mitigation, restoration and replacement cost methods. These methods have been extensively used in well-known attempts to estimate the total economic value of the world's ecosystem services (e.g. Costanza et al., 1997; Balmford et al., 2002). One problem with these methods is that they provide only a partial, lower bound indication of marginal value (Daily et al., 2000; Heal et al., 2005). More fundamentally, the definition of value in terms of avoided and replacement costs is inconsistent with economic theory, because, by definition, the economic value of an object is rather determined by how much consumers are willing to pay and producers are willing to supply for it at the margin (Portney, cited in Begley, 2002). In other words, the scarcer an ecosystem service is, the higher is its marginal value, independently from its avoided and replacement costs.

For what concerns stated preference methods, while contingent valuation is improving technically, sources of incoherencies do not disappear, including largely unsolvable issues of 'preference reversal' and 'question indeterminacy' (Tversky and Kahneman, 1986; Aldred, 2006), respondents' uncertainty about their own preferences for public ecosystem services (Spash and Hanley, 1995), and people's limited capacity to reduce disparate types of goods to a single metric, especially

[8] In the field of hedonic psychology, adaptation is defined as "any psychological mechanism that reduces the effects (perceptual, physiological, motivational, hedonic, etc.) of a constant and repeated stimulus" (Frederick and Loewenstein, 1999: 302). Depending on the circumstances, people may perform either 'hedonic' or 'resigned' processes of adaptation (Teschl and Comim, 2005). For the purposes of our analysis we focus on resigned adaptation.

when intrinsic values are involved (Vatn, 2005). These incoherencies mainly stem from the fact that the expression of a preference for a public ecosystem service is inherently different from the expression of a preference for a private commodity (Sagoff, 1988; Sugden, 1999).[9] Another unsolved issue in contingent valuation is the predominant use of WTP rather than WTA as a valuation measure, which ignores the actual allocation of rights over environmental resources and the pronounced discrepancy between these two measures when dealing with public ecosystem services (Horowitz and McConnell, 2002). Choice modelling methods, on the other hand, are constrained by cognitive complexity and statistical problems (Pearce et al., 2006). Overall, then, the inconsistencies observed in stated preference methods cast doubt about their reliability.

3. Ecosystem services and value aggregation

Cost-benefit analysis is carried out by aggregating the values of ecosystem services on a single scale of measurement. In this respect, the assumptions that all preferences are exogenous (i.e., given and context-independent) and continuous (i.e., commensurable with each other through a single unit of measure) are essential, because they make the aggregation of preferences on a single scale of measure possible (Vatn, 2005; Aldred, 2006). However, we shall now see that public ecosystem services pose serious challenges to these assumptions.

3.1. Endogenous preferences

Endogenous preferences are preferences influenced by the institutional context within which valuation takes place, such as the market, the community or the political arena (Niemeyer and Spash, 2001; Vatn, 2005). Seminal work on the social psychology of material possession has indicated that endogeneity is a pervasive characteristic of preferences. Sen (1985) showed how the perceived importance of material goods is strongly mediated by prevailing social institutions and cultural norms, and Dittmar (1992) highlighted how individuals assign value to commodities based on the symbolic meaning that these commodities hold in their social setting. From this literature it follows that an object may be assigned multiple values by the same individual depending on the institutional context within which valuation takes place (Brekke and Howarth, 2000; Vatn, 2005). For example, a tree species may be valued differently depending on whether the agent is acting as a consumer purchasing timber on the market, a community member deciding how to manage a local woodland, or a member of government designing a forestry policy for the whole country.

On the basis of the endogeneity of preferences, the existence of multiple and possibly conflicting utility functions or preference orderings has been postulated, reflecting whether an individual acts in her role of self-interested consumer or of citizen interested in the common good (Harsanyi, 1955; Sen, 1977; Sagoff, 1988). The problem for cost-benefit analysis is that if preferences are ordered according to different rules, then the aggregation of all preferences on a single scale of measure may not be feasible (Nyborg, 2000; Vatn, 2005; O'Neill et al., 2008).[10]

3.2. Lexicographic preferences

Lexicographic preferences are preferences holding intrinsic value that makes them incommensurable with other preferences on a single scale of measurement (Niemeyer and Spash, 2001; O'Neill et al., 2008). The existence of such preferences is revealed by 'protest bids' in all contingent valuation studies (Rekola, 2003). For example, people may refuse to attach a monetary value to a landscape because they consider it to embody their relation to the past and future of their communities (O'Neill et al., 2008). Or they may refuse to price a stable climate because they believe that the latter is related to a human right to a safe life. Moreover, lexicographic preferences constitute a theoretical obstacle to cost-benefit analysis insofar as they preclude the trade-offs simulated in the Kaldor-Hicks compensation test– when people have lexicographic preferences for certain environmental goods, the loss of these goods cannot be easily compensated for (Spash and Hanley, 1995; Gowdy, 2004). A trend in contingent valuation studies has been to either exclude lexicographic preferences from the analysis, or to force an equivalence between environmental and market goods upon respondents. In both cases, some respondents in the sample are disenfranchised and therefore results are biased (Sagoff, 1988; Gowdy, 2004).

3.3. Cardinality

There is some debate on whether cost-benefit analysis constitutes either a cardinalist or an ordinalist approach to policy evaluation. Some authors consider it to be a cardinalist approach, because it makes use of an aggregation social welfare function that necessarily utilises commensurable WTP addends (Accocella, 1998; Pearce et al., 2006), thus assuming that the strength of preferences can be quantified and compared across individuals. Others argue that it is an ordinalist approach, because it is based on the Kaldor-Hicks compensation criterion, which was originally developed as part of the Pareto ordinalist framework of new welfare economics (Kaldor, 1939). However, it has been noted that in reality the Kaldor-Hicks compensation criterion is based on the cardinal measure and interpersonal comparison of utility–compensation entails a comparison between the utility losses experienced by some with the utility gains experienced by others, and this comparison is only possible if the magnitude of respective utility gains and losses is known (Chipman and Moore, 1978; Pearce et al., 2006).[11]

The cardinalist approach to welfare economics was first developed by Pigou (1920) as a way to overcome the limitations of the Pareto ordinalist approach, since the latter allows only incomplete rankings of social choices that significantly restrict the scope of analysis, unless, as demonstrated by Arrow's (1951) impossibility theorem, dictatorship is allowed. However, the cardinality assumption has been largely contested too. In fact, it is generally argued that utility is a subjective state of consciousness that cannot be measured (Robbins, 1938), and that consequently the cardinal measure and interpersonal comparison of utility is

[9] Sometimes, the existence of markets for public ecosystem services (e.g. PES for water protection and carbon sequestration) is used as evidence to infer that an economic value can be attached to such goods through stated preference methods (e.g. Turner, 2007). In reality, the price of public ecosystem services in these negotiations is rarely defined on the basis of individual consumer preferences, as in markets for private commodities, but instead it is largely based on the postulated opportunity cost of provision incurred by the land-users. Therefore, the existence of PES schemes provides no theoretical support to the use of stated preference methods in ecosystem service valuation.

[10] During the past two decades, a new literature has emerged that attempts to integrate endogenous factors such as status-seeking behaviour, social norms and personal moral reasoning into economic models (e.g. Ng and Wang, 1993; Lindbeck, 1997; Brekke et al., 2003a). While important, these studies do not address how to deal with the endogeneity of preferences in cost-benefit analysis.

[11] For example, Pearce et al. (2006, p. 33) states that "interpersonal comparisons become essential with the hypothetical compensation test. If compensation is actually paid no problem arises. But if it is not actually paid then it is necessary to know if the gainers really could compensate the losers, i.e., the relative size of the gains and losses must be known, which means comparing utilities across different people".

only possible if we accept the unlikely assumption of an identical capacity for preference satisfaction across individuals. But if this assumption is rejected, then the fact that utility is a subjective non-measurable state of consciousness implies that interpersonal comparisons of utility can result in erroneous estimates (Harsanyi, 1955). When applied to environmental policy, some authors argue that this problem of interpersonal incomparability of utility may be safely ignored when preferences for ecosystem services are aggregated within limited spatio-temporal frames and across homogenous socio-economic groups (Gowdy, 2007). On the contrary, cost-benefit analysis estimates are subject to increasingly large margins of error when preferences are aggregated across geographically, generationally and socially distant groups of people who, we may assume, are characterised by greater biological, psychological and cultural differences (Harsanyi, 1955).

Overall, it seems that the mathematical (calculative) approaches to policy evaluation used in welfare economics are afflicted by an impasse. When used to evaluate a policy that impacts upon highly heterogeneous socio-economic groups, the cardinal measure and interpersonal comparison of utility can result in erroneous and misleading estimates. At the same time, it has been proved that it is not possible to base social choices on an ordinalist Pareto framework, because attempts to do so are either afflicted by a range of practical limitations and theoretical inconsistencies, or actually fail to avoid cardinality and interpersonal comparisons of utility. More specifically: (i) social welfare functions based on the Pareto principle allow only incomplete rankings of choices, which limits practical applications to policy evaluation and can give rise to a 'tyranny of the status quo' (Accocella, 1998); (ii) Arrow's axiomatic extension of the Pareto principle indicates that it is impossible to obtain a complete ranking of social choices on the basis of ordinal utility measures unless dictatorship is allowed; and (iii) social welfare functions associated with the Kaldor-Hicks compensation criterion actually fail to avoid cardinality and interpersonal comparisons of utility (Pearce et al., 2006), while also generating a range of theoretical inconsistencies (Gowdy, 2004). We therefore suggest that when highly heterogeneous socio-economic groups are involved, both ordinalist and cardinalist mathematical frameworks for policy evaluation be replaced by deliberative methods. These methods acknowledge that social choices tend to involve both gains and losses, and that consequently when evaluating a policy we cannot avoid comparing utility trade-offs among individuals and making ethical judgements about what outcome we may favour. In other words, these methods reject the ordinalist assumption that social choices can be made without interpersonal comparisons of gains and losses. At the same time, these methods also avoid problematic cardinalist calculations, and instead attempt to assess trade-offs through transparent and reasoned discussions among stakeholders.

4. Spatio-temporal frames

Ecosystem services are generated and experienced at different scales, from the short-term site-level (e.g. amenity services) to the long-term global-level (e.g. climate regulation) (Turner and Daily, 2008). As a consequence, a slight change in the spatial and temporal frames adopted in cost-benefit analysis can lead to a change in the range of consequences and stakeholders that are included in the analysis. The risk of biased framing is therefore particularly relevant (Hansson, 2007).

4.1. Biased spatial frames

There has been considerable progress in the use of spatially explicit valuation models, trying to account for the non-constancy of marginal values of services over the size of a given ecosystem, for the presence of substitute and complementary services, and for the rate of distance decay (Pascual et al., 2010). However, confusion remains regarding the choice of adequate spatial boundaries. The MA (2003) suggests that the area of analysis should be identified by overlaying the set of ecological factors that generate an ecosystem service together with the set of social factors that determine which stakeholders are affected by these services, mapping the location of discontinuities for each factor. The boundary of the area of analysis should be laid where a good number of these relative discontinuities coincide. This approach is generally endorsed, but it becomes an arduous undertaking when dealing with large-scale ecosystem change, in which case our knowledge of how and where ecosystem services are generated, altered and experienced is often insufficiently detailed. Consequently, when used to analyse large scale ecosystem change cost-benefit analysis risks adopting spatial boundaries that do not encompass all the impacts of a planned change.

In addition, large-scale ecosystem change carries the risk of approaching critical thresholds, and we have seen that under these conditions ecosystem services may assume absolute scarcity characteristics that cannot be properly captured through economic valuation. For example, the large-scale conversion of mangrove forest to shrimp aquaculture can lead to the long-term loss of protection from storm surges and of habitat for coastal fisheries (Barbier, 2000), which are fundamental life-supporting services for which no man-made alternatives are easily available. It is thus recommended that cost-benefit analysis be avoided when dealing with large-scale ecosystem change, since in this case it can result in inaccurate estimates and misguided decisions.

4.2. Biased temporal frames

The choice of the temporal scale of analysis is also critical, because it determines to what extent the interests of future generations are included in cost-benefit analysis. However, the use of temporal scales that extend far into the future can be problematic for at least two reasons. First, the more we extend our analysis into the future the less accurate our predictions become, because the co-evolution of social-ecological systems is not predictable, and this limits the accurate estimation of future ecosystem services scarcity and value (Winkler, 2006). The second issue regards the difficult choice of an appropriate discount rate. The conventional practice is to discount the future stream of ecosystem services by using a positive discount rate based on the Ramsey equation (Gowdy et al., 2010). This practice disfavours future generation (Chichilnisky, 1996), and yet it is generally defended on the basis of three hypotheses that are implicit in the Ramsey equation. We argue that these hypotheses are contentious.

Starting from the 'pure time preference' hypothesis, this justifies the use of a positive discount rate on the basis that individuals tend to attach less value to outcomes that they will experience in the future than to outcomes that they experience in the present. However, it is debatable whether this private rate of time preference, determined in the market-place through individual consumption and saving choices, should be used to guide society's public choices about the allocation of public ecosystem services across generations (Marglin, 1963). Individuals may discount their own future out of impatience, but society as a whole may be less impatient and pursue an equal treatment for all generations.

The hypothesis of 'diminishing marginal utility' of consumption advocates the use of a positive discount rate on the basis that future generations are assumed to enjoy higher levels of consumption than present ones. This hypothesis can be objected for several reasons. First of all, the multidimensional character of human well-being implies that opportunities for self-determination and development are in part linked to healthy and diverse ecosystems and cannot be replaced by higher per capita incomes (Norton, 2005). Furthermore, everlasting increases in total factor productivity or per capita income levels cannot be projected with any certainty (Dasgupta et al., 1999; Winkler, 2006). On the contrary, it is increasingly recognised that when the depreciation of natural capital and population growth are properly accounted for, projected rates of per capita GDP growth are greatly reduced, and in the case of poor people who heavily rely on ecosystem services such rates may actually be negative (Arrow et al., 2004). As a corollary, if the degradation of natural capital was accounted for in the Ramsey equation, the optimal discount rate would result to be lower than in current estimations, and in some circumstances even negative (Dasgupta et al., 1999; Gowdy et al., 2010).

Discounting future costs and benefits has also been claimed to be necessary to compensate present generations for any opportunity cost that they may incur when preserving natural capital for the benefit of future generations. However, this argument only makes sense if we assume that all forms of capital are truly substitutable on both the production and consumption sides of the economy. This is a highly debatable assumption when dealing with life-supporting ecosystem services like climate regulation and biodiversity maintenance. These services are non-substitutable and absolutely scarce, and consequently their preservation cannot entail an opportunity cost that can be discounted from future net benefits (Neumayer, 2007).

Finally, the use of a positive discount rate has been invoked on the basis that, otherwise, cost-benefit analysis tends to result in the selection of policies that demand excessive consumption sacrifices on current generations (Pearce et al., 2006; Turner, 2007), and that the most disadvantaged sections of present societies may be particularly affected by these sacrifices (Beckerman and Pasek, 1997). However, this normative view can be debated on the basis that the first mechanism to pursue urgent poverty alleviation should be a redistribution of consumption opportunities within and among societies in the present, rather than the discharge of costs into the future.

All in all, no conclusive argument in favour of positively discounting future ecosystem services has been advanced yet. At the same time, if positive discounting leads to biases against future generations, zero discounting may lead to similar biases against present (Chichilnisky, 1996). This happens because over long time horizons the number of future generations included in the analysis is large, and therefore with a zero discount rate the cumulative benefits that would accrue to them from constrained consumption in the present tend to substantially outweigh the present costs of this sacrifice (Olson and Bailey, 1981). The use of hyperbolic (or declining) discount rates has been advocated as a way to neutralise the cumulative effect of including many future generations in the analysis without at the same time biasing results against them, thereby ensuring that the interests of present and future generations are treated equally (Beltratti et al., 1998; Li and Löfgren, 2000).[12] Unfortunately, mathematical limitations (Hanley, 2000) and problems of time-inconsistency (Li and Löfgren, 2000) limit the applicability of a hyperbolic discount rate.

In the face of the controversies attached to each type of discount rate, Gowdy et al. (2010) suggest that a variety of discount rates may be used when assessing the long-term impacts of environmental change, including hyperbolic, zero and negative rates, depending on the time period and the type of ecosystem services involved in the analysis. While this approach offers a more flexible way of dealing with future ecological impacts, we also stress that since the use of any discount rate entails potential problems of intergenerational justice, we should restrict the use of cost-benefit analysis to the assessment of those policies whose impacts do not extend far into the future.

5. Why does cost-benefit analysis persist despite its own predicaments?

Despite the list of shortcomings discussed in this paper, the application of cost-benefit analysis to ecosystem services continues to receive support in academic and policy-making circles on the basis of four main postulated merits: (i) expediency, (ii) democracy, (iii) value-neutrality, and (iv) inescapability of trade-offs.

Expediency is one of the most popular arguments advanced in support of environmental cost-benefit analysis, claiming that the market is nowadays a ubiquitous institution for the satisfaction of large part of human needs and wants, and that consequently the couching of ecosystem services in economic terms is the best guarantee to their contemplation in public decisions (Pearce, 2000; Turner et al., 2003; Heal et al., 2005). However, this claim is contradicted by evidence from the real world of politics–even when governments recommend the use of cost-benefit analysis to guide policy-making, public decisions are often based on goals other than economic efficiency, such as the need to align with the prevalent public opinion or to co-opt interest groups and lobbies (Kalt and Zupan, 1984). Hence, the contemplation of public ecosystem services in decision-making may be better guaranteed by deliberative procedures that increase participation and transparency, rather than by abstract efficiency calculations. Furthermore, when dealing with ecological change that involves extensive and unequal impacts on society, high levels of scientific uncertainty and deontological ethics, the expediency argument may be automatically rejected–there is nothing pragmatic about using a policy evaluation tool that is based on inadequate theoretical assumptions and that can result in inaccurate estimates and misguided decisions (Gowdy, 2004; Norgaard, 2010).

Another assumption is often made that cost-benefit analysis constitutes a democratic approach to decision-making, because it treats the preferences of all individuals equally (basically, one person one vote one preference), and because its goal is the maximisation of social welfare. However, this argument is based on a reductive interpretation of democracy as the aggregation of isolated preferences, and on a reductive definition of the social goal in terms of economic efficiency. These notions are rejected by more articulated political theories that define democracy as 'deliberative governance' with the 'consent of citizens' (Howarth and Wilson, 2006), and the social goal as the creation of opportunities for individuals to identify and pursue a broader set of capabilities (Sen, 1999).

The valuation of ecosystem services through prices is not as democratic as purported also for other reasons. As discussed above, the distribution of wealth in society influences how prices are determined, so that the preferences of wealthier groups tend to be disproportionately represented in economic valuations (Bromley, 1991; Sen, 2000; Spash, 2008). In addition, the use of cost-benefit analysis in environmental decision-making can lead to a politics of exclusion. In fact, cost-benefit analysis consists of a restrictive set of rules concerning the role of participants (i.e. as consumers), what counts as data (i.e. prices), and how data is produced (i.e. individually) and aggregated (i.e. by summing individual prices)

[12] The use of time-varying discount rates in cost-benefit analysis has been supported also for other theoretical reasons. Frederick et al. (2002) support it because it better describes 'pure time preferences'. For Gollier (2002), on the other hand, a hyperbolic discount rate is a better way of dealing with our uncertainty about future per capita consumption. However, as for the positive discount rate discussed above, these reasons are misconceived – people's private time preferences are not an adequate reference for public decision-making, and human well-being is not determined uniquely by absolute levels of consumption.

(Vatn, 2005). Stakeholders who define environmental problems differently (e.g. as citizens and through collective deliberation) or are unfamiliar with market institutions may therefore be left out of the decision process (Martínez-Alier, 2002; Norton and Noonan, 2007).

The value-neutrality of cost-benefit analysis might also be seen as a naive postulation, because on the contrary this tool requires numerous normative judgments, from the choice of the geographical boundaries of analysis, to the use of equity-weights, to the setting of a discount rate (Spash, 2008). This is evident from economic studies of global environmental change such as the Stern Review (2006) on climate change, where normative judgements had to be made on important issues such as the treatment of the poor, the standing of future generations, and the characterisation of uncertainty and risk.

Lastly, an often heard argument in favour of cost-benefit analysis is the inescapability of trade-offs in decision-making. The preservation of elements of nature that are intrinsically valued often entails opportunity costs that also embrace other intrinsic values, and in these circumstances people may eventually feel forced to allow their trade-off (Beckerman and Pasek, 1997; Pearce, 2000). For example, an agent may believe that a globally threatened forest habitat has intrinsic values, and that therefore it should be preserved independently from any economic consideration. However, the conservation of this forest habitat may entail the loss of livelihoods and cultural identity for a local community, things that the agent may also value intrinsically as human rights. Consequently, the agent may eventually feel forced to trade-off some of the forest intrinsic values for the protection of some human rights. On the basis of this inescapability of trade-offs, it is argued, people may as well accept the monetary commensurability of the intrinsic values that they hold towards nature. However, while we do not deny that at high opportunity costs trade-offs may be unavoidable, it is arguable whether such conflicting ethical choices can be properly dealt with through a technocratic approach such as cost-benefit analysis (Martínez-Alier et al., 1998; Norton and Noonan, 2007; O'Neill et al., 2008). Instead, their resolution requires open deliberation over the underlying interests, motivations and values. This may apply not only to ecosystem services, but to all entities that attract intrinsic and deontological values, including, for example, also historical, religious and artistic heritage sites.

6. Towards a pluralist framework for ecosystem service decision-making

In order to remediate the limitations of cost-benefit analysis highlighted in this paper, we favour the adoption of a pluralist framework for environmental decision-making,[13] within which a heterogeneous set of value-articulating and decision-making instruments is available to represent in a more comprehensive manner the multiple ways that people value and make decisions about the environment. Such a framework ought to recognise and account for the existence of plural values (both commensurable and incommensurable) and ethical attitudes (both consequentialist and deontological) towards nature (O'Neill et al., 2008). It also needs to acknowledge the interpersonal origins of decision-making and the uncertainty that permeates the scientific understanding of complex social-ecological systems, which implies that important public decisions over the environment cannot be adequately evaluated through scientific models alone, but rather require the deliberative interaction of a broad set of stakeholders (Vatn, 2005). A pluralist framework should also improve the transparency of the evaluation process by bringing underlying normative motivations and distributional issues to the forum, rather than living them hidden in the price mechanism (Spash, 2008). To summarise, a pluralist framework acknowledges that environmental choices are inevitably complex and multidimensional, involving heterogeneous groups of stakeholders and a broad range of competing interests, and often leading to intense confrontation and difficult choices (O'Neill et al., 2008). The task and challenge of policy-making should then be to reconcile these competing interests and values until a consensus emerges, rather than to turn decisions into abstract efficiency calculations.

6.1. Emerging tools for decision-making

Emerging tools in a pluralist framework for decision-making include multi-criteria assessment and deliberative approaches. Differently from cost-benefit analysis, in which economic efficiency is the unique objective, multi-criteria assessment methods evaluate projects in terms of multiple objectives (e.g. poverty alleviation, psychological health and ecological resilience) that can be differently prioritised. Several criteria are used to indicate the level of achievement of each objective, and these criteria are measured through different units of measure that are not necessarily converted to a single monetary metric. Each alternative policy option is given a score for each objective on the basis of its impacts on relevant criteria. These scores are then weighted according to the level of priority assigned to each objective, and an average of these scores is calculated for each alternative policy (Munda, 2004). Multi-criteria assessment therefore corrects the utilitarian and monistic approach of cost-benefit analysis, by allowing the prioritisation of multiple objectives and by utilising plural units of measure that respect the incommensurability of certain values (Munda, 2004). However, multi-criteria assessment does not yet constitute an unquestionable solution for public decision-making–there are several technical problems still affecting the aggregation procedure, as well as significant power asymmetries among participating stakeholders (Vatn, 2005).

While both cost-benefit analysis and multi-criteria assessment are based on a calculative logic, deliberative methods are based on discursive interactions between lay people, scientists and decision makers about how to address collective environmental issues. These methods are derived from political theories of communicative action that account for the fact that understanding and values are socially constructed (Habermas, 1984), and from political theories of discursive democracy that interpret democracy as 'deliberative governance' with the 'consent of citizens' (Rawls, 1971; Dryzek, 2000). Accordingly, the aim of deliberative methods is to enable the careful assessment of the information, interests and values that underlie a decision, and to channel the disparate conceptions of the social good into the construction of a consensus-based solution (Howarth and Wilson, 2006; Antunes, 2008). Where sufficient ecological knowledge is available, deliberative approaches can benefit from the use of biophysical models that project the level of ecosystem services provided under different scenarios, such as maps and trade-off curves, but without attempting their economic valuation (Tallis and Polasky, 2009).

Several deliberative tools have been developed since the 1970s, including focus groups, citizens' juries, consensus conferences and deliberative polls (Wilson and Howarth, 2002; Getzner et al., 2005). In practical terms, these approaches are still relatively nascent, and thus problems of inclusiveness, representation, competence and manipulation have not been fully resolved yet (Niemeyer and Spash, 2001; Carter et al., 2007; Reed, 2008). Nonetheless, these problems do not necessarily affect deliberative

[13] The pursuit of a pluralist approach to environmental decision-making is part of a more general call for methodological pluralism in the social and ecological sciences (Norgaard, 1989), and follows the footprints of parallel work on the pluralist governance of complex socio-ecological systems (Ostrom, 1998).

methods more than cost-benefit analysis and other welfarist approaches to policy evaluation. As we have argued in this paper, cost-benefit analysis can result in the exclusion of stakeholders who are not able or willing to define their values for environmental goods in monetary terms, it tends to under-represent the preferences of the poor, it is afflicted by ignorance an uncertainty about ecological processes, and it can be exposed to preference manipulation through social norms and advertisement. Moreover, while in cost-benefit analysis these problems tend to remain hidden in the price mechanism, deliberative approaches present the potential advantage to tackle them more directly through institutional design (Accocella, 1998; Spash, 2008). Therefore, the standard argument in defence of cost-benefit analysis that it should not be criticized unless a full-blown alternative exists is now less valid than ever, as a number of alternative approaches for decision-making are being refined. Claiming that "cost-benefit analysis still seems the best game in town" (Pearce, 2000: 74) or that "the TEV framework helps [. . .] in valuing ecosystem services as comprehensively as possible" (Heal et al., 2005: 5) means remaining blind to the plethora of emergent alternatives for environmental decision-making.

6.2. Should cost-benefit analysis be discarded altogether?

Can cost-benefit analysis be considered a useful component of a pluralist framework for environmental decision-making, or should it be discarded altogether? For heuristic purposes, perspectives on this issue may be divided into three groups: (i) those who believe that through appropriate technical advances cost-benefit analysis can be increasingly applied to the evaluation of policies that affect ecosystem services; (ii) those who believe that theoretical inconsistencies remain negligible in some environmental applications of cost-benefit analysis, but become unacceptable when ecosystem change involves significant uncertainty and potentially large and irreversible impacts; and (iii) those who believe that cost-benefit analysis should be discarded altogether because its weak theoretical foundations imply that its estimates are never fully reliable.

The majority of opinions would probably fit in the second group. When the subject of valuation is a simple good that is privately owned and traded on the market, then economic valuation may be adequate–for example, when calculating the monetary benefit of pollination services from a remnant tropical forest to a local coffee production (Ricketts et al., 2004). Similarly, cost-benefit analysis may be suitable to examine local projects that have marginal and spatio-temporally limited impacts on ecosystems and their services–for example, a municipality deciding whether to invest in natural or man-made structures for flood control along a short riparian tract (Gowdy, 2007). On the contrary, alternative deliberative tools should be used when decisions involve intangible dimensions of human well-being, intrinsic ethics, critical thresholds, scientific uncertainty, significant distributional biases and spatio-temporally extensive ecological impacts.

A current limitation of this middle-ground approach is that no consensus rules have been developed yet to guide the selective use of cost-benefit analysis. For example, it remains difficult to identify where ecosystem thresholds occur for different ecosystem services (Fisher et al., 2008). Similarly, it is not always possible to predict with certainty whether a specific policy may involve extensive ecological impacts, intangible dimensions of human well-being and strong distributional biases. For example, Gowdy (2007) refers to the construction of a new bridge as an example of a policy that carries limited impacts and trade-offs and that may therefore be safely evaluated through cost-benefit analysis. However, in some circumstances the construction of a new bridge has actually had extensive impacts on an ecosystem and its inhabitants. In Tanzania, the completion of the Mkapa bridge in 2003 increased the opportunities for economic development in remote regions of the country. However, in the absence of strong regulatory mechanisms, the bridge also contributed to escalating illegal timber harvesting and forest degradation in these regions (Milledge et al., 2007), with potentially serious repercussions on forest-dependent local communities and threats to a globally unique biodiversity. New research is required to clarify in what contexts cost-benefit analysis may be safely utilised.

7. Conclusions

Progress in our understanding of human behaviour and ecological change illustrates that the normative premise that underlies cost-benefit analysis, i.e. that all ecosystem services should be valued in terms of individual economic preferences, is inconsistent with the psychological, interpersonal and context-dependent origins of human behaviour and the non-linear character of ecological change. Individuals may perceive intangible benefits from nature, may not always have sufficient understanding of ecosystem services, and may not always decide independently from others on what value to attribute to ecosystem services. In these circumstances individuals may fail to value ecosystem services using a monetary unit of measure. Individuals also tend to value elements of nature according to different rules, depending on the institutional context within which they act, and this hinders the aggregation of all values on a single scale of measure. Meanwhile, the notion of economic value is of little use when an ecosystem approaches a critical ecological threshold and ecosystem services become non-substitutable and absolutely scarce. Finally, the unequal distribution of wealth in society may significantly influence the determination of market prices, biasing the analysis towards the preferences of higher-income groups and resulting in the under-provision of important ecosystem services to lower-income groups. Under this set of circumstances, willingness to pay may not constitute a valid indicator of the value that people hold towards ecosystem services, and as a consequence cost-benefit analysis may result in inaccurate estimates and offer an unreliable guide to decision-making. In addition, the margin of error in cost-benefit analysis can further increase when this tool is used to evaluate ecological change over geographically and temporally extensive scales.

In order to remediate to the limitations of cost-benefit analysis, rather than locking environmental policy-making into a monistic approach that evaluates policies uniquely in terms of monetary value and efficiency outcomes, we favour opening it up to a pluralist framework instead. Such a framework would be composed of a heterogeneous set of value-articulating and decision-making instruments that are appropriate to the type of ecosystem service analyzed, the dimensions of well-being and spheres of ethics affected, the spatial and temporal scales involved, and the likely conditions of ecological non-linearity, scientific uncertainty and distributional disparity. Cost-benefit analysis may remain a component of this pluralist framework, since it may be suitable to examine local projects that have marginal and spatio-temporally limited impacts on ecosystem services, but its applicability should be carefully assessed each time.

Rather than channelling all our efforts and resources into technical debates about how to convert all biophysical processes and human values into a monetary measure for cost-benefit analysis, we call for the development of increasingly realistic and comprehensive institutions for the evaluation of environmental choices. This may appear to be a Sisyphean challenge to those in entrenched positions adamantly supporting mainstream views on ecosystem service valuation and environmental decision-making.

We should not fall in those muddy trenches, but be responsive to the outcomes of progressive collaborative ventures between a broad set of humanistic and scientific disciplines.

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References

Accocella, N., 1998. The Foundations of Economic Policy: Values and Techniques. Cambridge University Press, Cambridge.
Aldred, J., 2006. Incommensurability and monetary valuation. Land Economics 82 (2), 141–161.
Andreoni, J., 1990. Impure altruism and donations to public goods: a theory of warm-glow giving. The Economic Journal 100, 464–477.
Anthoff, D., Cameron, H., Richard, T.S.J., 2009. Equity weighting and the marginal damage costs of climate change. Ecological Economics 68 (3), 836–849.
Antunes, P., 2008. Principles and concepts for participation and deliberation. Lecture at the "THEMES: Methods and tools for environmental appraisal and policy formulation". Summer School in Lisbon, 25th May–6th June 2008.
Arrow, K.J., 1951. Individual Values and Social Choice. Wiley, New York.
Arrow, K., Fisher, A., 1974. Environmental preservation, uncertainty and irreversibility. Quarterly Journal of Economics 88, 312–319.
Arrow, K., Dasgupta, P., Goulder, L., Daily, G., Ehrlich, P., Heal, G., Levin, S., Mäler, K.G., Schneider, S., Starrett, S., Walker, B., 2004. Are we consuming too much? Journal of Economic Perspectives 18 (3), 147–172.
Arrow, K., Dasgupta, P., Mäler, K.G., 2008. Resilience and sustainable development: building adaptive capacity in a world of transformations. In: Perrings, C. (Ed.), Ecological Economics, Volume 4. SAGE Publishing.
Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R.E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K., Turner, R.K., 2002. Economic reasons for conserving wild nature. Science 297, 950–953.
Barbier, E.B., 2000. Valuing the environment as input: review of applications to mangrove-fishery linkages. Ecological Economics 35 (1), 47–61.
Baumgärtner, S., Becker, C., Faber, M., Manstetten, R., 2006. Relative and absolute scarcity of nature. Assessing the roles of economics and ecology for biodiversity conservation. Ecological Economics 59 (4), 487–498.
Beckerman, W., Pasek, J., 1997. Plural values and environmental valuation. Environmental Values 6, 65–86.
Begley, S., 2002. Furry Math? Market has failed to capture true value of nature. The Wall Street Journal, New York, August 9.
Beltratti, A., Chichilnisky, G., Heal, G., 1998. Sustainable use of renewable resources. In: Beltratti, A., Chichilnisky, G., Heal, G. (Eds.), Sustainability: Dynamics and Uncertainty. Kluwer, Dordrecht.
Boadway, R., 1974. The welfare foundations of cost-benefit analysis. Economic Journal 84, 926–939.
Brekke, K.A., Howarth, R.B., 2000. The social contingency of wants. Land Economics 76 (4), 493–503.
Brekke, K.A., Howarth, R.B., Nyborg, K., 2003. Status-seeking and material affluence: evaluating the Hirsch hypothesis. Ecological Economics 45, 29–39.
Brekke, K.A., Kverndokk, S., Nyborg, K., 2003. An economic model of moral motivation. Journal of Public Economics 87, 1967–1983.
Bromley, D.W., 1991. Environment and Economy: Property Rights and Public Policy. Basic Blackwell.
Brooks, S., 2005. Landscape, politics and the historical geography of Southern Africa images of 'Wild Africa': nature tourism and the (re)creation of Hluhluwe game reserve, 1930–1945. Journal of Historical Geography 31 (2), 220–240.
Carpenter, S.R., Mooney, H.A., Agard, K., Capistrano, D., DeFries, R.S., Díaz, S., Dietz, T., Duraiappah, A.K., Oteng-Yeboah, A., Pereira, H.M., Perrings, C., Reid, W.V., Sarukhan, J., Scholes, R.J., Whyte, A., 2009. Science for managing ecosystem services: beyond the Millennium Ecosystem Assessment. Proceedings of the National Academy of Sciences (PNAS) 106 (5), 1305–1312.
Carter, C., Ekeli, K., Kenyon, W., O'Neill, J., 2007. Representing Diversity. Path Policy Research Brief 1. Macaulay Institute, Aberdeen.
Chee, Y., 2004. An ecological perspective on the valuation of ecosystem services. Biological Conservation 120 (4), 549–565.
Chiesura, A., De Groot, R., 2003. Critical natural capital: a socio-cultural perspective. Ecological Economics 44, 219–231.
Chichilnisky, G., 1996. An axiomatic approach to sustainable development. Social Choice and Welfare 13, 231–257.
Chipman, J., Moore, J., 1978. The New Welfare Economics 1939–1974. International Economic Review 19 (3), 547–584.
Ciriacy-Wantrup, C.V., 1968. Resource Conservation: Economics and Policies. University of California, Berkeley.
Coley, R., Kuo, F., Sullivan, W., 1997. Where does community grow? The social context created by nature in urban public housing. Journal of Environment and Behaviour 29, 468–494.
Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., Naeem, S., Limburg, K., Paruelo, J., O'Neill, R.V., Raskin, R., Sutton, P., Van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.
Daily, G.C., Soderqvist, T., Aniyar, S., Arrow, K., Dasgupta, P., Ehrlich, P.R., Folke, C., Jansson, A., Jansson, B., Kautsky, N., Levin, S., Lubchenco, J., Maler, K., Simpson, D., Starrett, D., Tilman, D., Walker, B., 2000. The value of nature and the nature of value. Science 289, 395–396.
Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., Shallenberger, R., 2009. Ecosystem services in decision making: time to deliver. Frontiers in Ecology and the Environment 7 (1), 21–28.
Danielsen, F., Sørensen, M.K., Olwig, M.F., Selvam, V., Parish, F., Burgess, N.D., Hiraishi, T., Karunagaran, V.M., Rasmussen, M.S., Hansen, L.B., Quarto, A., Suryadiputra, N., 2005. The Asian tsunami: a protective role for coastal vegetation. Science 310, 643.
Dasgupta, P., Mäler, K.G., 2004. Environmental and Resource Economics: Some Recent Developments. Discussion Paper vol. 186. The Beijer International Institute of Ecological Economics. The Royal Swedish Academy of Sciences, Stockholm, Sweden.
Dasgupta, P., Mäler, K.-G., Barrett, S., 1999. Intergenerational equity, social discount rates and global warming. In: Portney, P.R., Weyant, J.P. (Eds.), Discounting and Intergenerational Equity. Resources for the Future, Washington DC.
DeYoung, B., Barange, M., Beaugrand, G., Harris, R., Perry, R., Scheffer, M., Werner, F., 2008. Regime shift in marine ecosystems: detection, prediction and management. Trends in Ecology and Evolution 23 (7), 402–409.
Dittmar, H., 1992. The Social Psychology of Material Possession. St. Martins, New York.
Dryzek, J.S., 2000. Discursive Democracy: Politics, Policy and Political Science. Cambridge University Press, New York.
Etzioni, A., 1988. The Moral Dimension: Toward a New Economics. Free Press, New York.
Farber, S.C., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. Ecological Economics 41 (3), 375–392.
Farley, J., 2008. The role of prices in conserving critical natural capital. Conservation Biology 22 (6), 1399–1408.
Fisher, B., Turner, K., Zylstra, M., Brower, R., de Groot, R., Farber, S., Ferraro, P., Green, R., Hadley, D., Harlow, J., Jefferiss, P., Kirkby, C., Morling, P., Mowatt, S., Naidoo, R., Paavola, J., Strassburg, B., Yu, D., Balmford, A., 2008. Ecosystem services and economic theory: integration for policy-relevant research. Ecological Applications 18 (8), 2050–2067.
Fisher, J.B., Maggi, K., Romm, J., 2006. Scales of environmental justice: combining GIS and spatial analysis for air toxics in West Oakland, California. Health & Place 12, 701–714.
Frank, R.H., 1985. Choosing the Right Pond. Oxford University Press, New York.
Frederick, S., Loewenstein, G., 1999. Hedonic adaptation. In: Kahneman, D., Diener, E., Schwarz, N. (Eds.), Well-Being: The Foundations of Hedonic Psychology. Russell Sage Foundation, New York.
Frederick, S., Loewenstein, G., O'Donoghue, T., 2002. Time discounting and time preference: a critical review. Journal of Economic Literature XL, 351–401.
Freeman, M., 1993. The Measurement of Environmental and Resource Values: Theory and Methods. Resources for the Future, Washington D.C.
Frey, B.S., Stutzer, A., 2002. Happiness and Economics. Princeton University Press.
Getzner, M., Spash, C.L., Stagl, S., 2005. Alternatives for Environmental Valuation. Routledge, London.
Gollier, C., 2002. Discounting an uncertain future. Journal of Public Economics 85, 149–166.
Gómez-Baggethun, E., de Groot, R., Lomas, P.L., Montes, C., 2010. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecological Economics 69, 1209–1218.
Gowdy, J., 2004. The revolution in welfare economics and its implications for environmental valuation. Land Economics 80 (2), 239–257.
Gowdy, J.M., 2007. Toward an experimental foundation for benefit-cost analysis. Ecological Economics 63 (4), 649–655.
Gowdy, J.M., Howarth, R.B., Tisdell, C., 2010. Discounting, ethics, and options for maintaining biodiversity and ecosystem integrity, Chapter 6. In: Kumar, P. (Ed.), The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. Earthscan, UK.
Habermas, J., 1984. The Theory of Communicative Action. Beacon Press, Boston.
Hanley, N., 2000. Macroeconomic measures of 'sustainability'. Journal of Economic Surveys 14 (1), 1–30.
Hansson, S.O., 2007. Philosophical problems in cost-benefit analysis. Economics and Philosophy 23, 163–183.
Harsanyi, J.C., 1955. Cardinal welfare, individualistic ethics, and interpersonal comparisons of utility. Journal of Political Economy 63, 309–321.
Heal, G.M., Barbier, E.E., Boyle, K.J., Covich, A.P., Gloss, S.P., Hershner, C.H., Hoehn, J.P., Pringle, C.M., Polasky, S., Segerson, K., Shrader-Frechette, K., 2005. Valuing

Ecosystems Services: Toward Better Environmental Decision-making. The National Academies Press, Washington DC.
Holling, C.S., 2001. Understanding the complexity of economic, ecological and social systems. Ecosystems 4, 390–405.
Horowitz, J., McConnell, K., 2002. A review of WTA/WTP studies. Journal of Environmental Economics and Management 44, 426–447.
Howarth, R.B., Wilson, M.A., 2006. A theoretical approach to deliberative valuation: aggregation by mutual consent. Land Economics 82 (1), 1–16.
Kaldor, N., 1939. Welfare propositions of economics and interpersonal comparisons of utility. The Economic Journal 49 (195), 549–552.
Kahneman, D., Wakker, P., Sarin, R., 1997. Back to Bentham? Explorations of experienced utility. Quarterly Journal of Economics 112, 375–405.
Kalt, J., Zupan, M., 1984. Capture and ideology in the economic theory of politics. American Economic Review 74 (3), 279–300.
Kriström, B., 2006. Framework for assessing the distribution of financial effects of environmental policies. In: Serret, Y., Johnstone, N. (Eds.), The Distributional Effects of Environmental Policy. OECD, Paris, and Edward Elgar, Cheltenham.
Kumar, M., Kumar, P., 2008. Valuation of the ecosystem services: a psycho-cultural perspective. Ecological Economics 64 (4), 808–819.
Li, C.Z., Löfgren, K.G., 2000. Renewable resources and economic sustainability: a dynamic analysis with heterogeneous time preferences. Journal of Environmental Economics and Management 40, 236–250.
Lindbeck, A., 1997. Incentives and social norms in household behaviour. The American Economic Review 87 (2), 370–377.
Lipsey, R., Lancaster, K., 1956-1957. The general theory of second best. Review of Economic Studies 24, 11–32.
Little, I.M.D., 1949. The foundations of welfare economics. Oxford Economic Papers 1 (2), 227–246.
Marglin, S.A., 1963. The social rate of discount and the optimal rate of investment. The Quarterly Journal of Economics 77 (1), 95–111.
Martínez-Alier, J., 2002. Environmentalism of the Poor: A Study of Ecological Conflicts and Valuation. Oxford University Press, New Delhi.
Martínez-Alier, J., Munda, G., O'Neil, G.J., 1998. Weak comparability of values as a foundation for ecological economics. Ecological Economy 26, 277–286.
MA, 2003. Ecosystems and Human Well-being: A Framework for Assessment. Millennium Ecosystem Assessment. WRI, Washington DC.
Milledge, S.A.H., Gelvas, I.K., Ahrends, A., 2007. Forestry, Governance and National Development: Lessons Learned from a Logging Boom in Southern Tanzania. An overview. TRAFFIC East/Southern Africa, Tanzania Development Partners Group and Ministry of Natural Resources and Tourism, Dar es Salaam.
Munda, G., 2004. Social Multi-criteria evaluation. Methodological foundations and operational consequences. European Journal of Operational Research 158, 662–677.
Neumayer, E., 2003. Weak Versus Strong Sustainability: Exploring the Limits of Two Opposing Paradigms. Edward Elgar, Cheltenham.
Neumayer, E., 2007. A missed opportunity: the Stern review on climate change fails to tackle the issue of non-substitutable loss of natural capital. Global Environmental Change 17 (3/4).
Ng, Y., Wang, J., 1993. Relative income, aspiration, environmental quality, individual and political myopia: why may the rat-race for material growth be welfare reducing? Mathematical Social Sciences 26 (1), 3–23.
Niemeyer, S, Spash, C.L., 2001. Environmental valuation analysis, public deliberation, and their pragmatic syntheses: a critical appraisal. Environmental and Planning C: Government and Policy 19, 567–585.
Norgaard, R.B., 1989. The case for methodological pluralism. Ecological Economics 1, 37–57.
Norgaard, R.B., 2010. Ecosystem services: from eye-opening metaphor to complexity blinder. Ecological Economics 69 (6), 1219–1227.
Norton, B.G., 2005. Sustainability: A Philosophy of Adaptive Ecosystem Management. University of Chicago Press, Chicago.
Norton, B., Noonan, D., 2007. Ecology and valuation: big changes needed. Ecological Economics 63 (4), 664–675.
Nussbaum, M., 2000. Women and Human Development. Cambridge University Press, Cambridge.
Nyborg, K., 2000. Homo economicus and Homo politicus: interpretation and aggregation of environmental values. Journal of Economic Behaviour & Organization 42 (3), 305–322.
Olson, M., Bailey, M., 1981. Positive time preference. Journal of Political Economy 89 (1), 1–25.
O'Neill, J., Holland, A., Light, A., 2008. Environmental Values. Routledge, London.
Ostrom, E., 1998. Scales, polycentricity, and incentives: designing complexity to govern complexity. In: Guruswamy, L.D., McNeely, J.A. (Eds.), Protection of Global Biodiversity: Converging Strategies. Duke University Press, Durham.
Paavola, J., Adger, N.W., 2005. Institutional ecological economics. Ecological Economics 53 (3), 353–368.
Padulosi, S., Hodgkin, T., Williams, J.T., Haq, N., 2002. Underutilized crops: trends, challenges and opportunities in the 21st Century. In: Engels, J.M.M., Ramanatha Rao, V., Brown, A.H.D., Jackson, M.T. (Eds.), Managing Plant Genetic Resources. CABI-IPGRI, Rome.
Pascual, U., Muradian, R., Brander, L., Gómez-Baggethun, E., Martín-López, B., Verma, M., Armsworth, P., Christie, M., Cornelissen, H., Eppink, F., Farley, J., Loomis, J., Pearson, L., Perrings, C., Polasky, S., 2010. The economics of valuing ecosystem services and biodiversity. Chapter 5. In: Kumar, P. (Ed.), The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. Earthscan, UK.
Pearce, D., 2000. Cost-benefit analysis and environmental policy. In: Helm, D. (Ed.), Environmental Policy: Objectives, Instruments, and Implementation. Oxford University Press, Oxford.
Pearce, D.W., Turner, R.K., 1990. Economics of Natural Resources and the Environment. Harvester, Hemel Hempsted.
Pearce, D., Atkinson, G., Mourato, S., 2006. Cost-benefit Analysis and the Environment: Recent Developments. OECD, Paris.
Pigou, A.C., 1920. The Economics of Welfare. Macmillan and Co, London.
Rawls, J., 1971. A Theory of Justice. Harvard University Press, Cambridge.
Reed, M.S., 2008. Stakeholder participation for environmental management: a literature review. Biological Conservation 141, 2417–2431.
Rekola, M., 2003. Lexicographic preferences in contingent valuation: a theoretical framework with illustrations. Land Economics 79 (2), 277–291.
Ricketts, T.H., Daily, G.C., Ehrlich, P.R., Michener, C.D., 2004. Economic value of tropical forest to coffee production. PNAS 101 (34), 12579–12582.
Robbins, L., 1938. Interpersonal comparisons of utility: a comment. Economic Journal 48, 635–641.
Sagoff, M., 1988. The Economy of the Earth. Cambridge University Press, Cambridge.
Scheffer, M., 2001. Alternative attractors of shallow lakes. The Scientific World 1, 254–263.
Scitovsky, T., 1941. A note on welfare propositions in economics. Review of Economic Studies 9, 77–88.
Sen, A.K., 1977. Rational fools: a critique of the behavioural foundations of economic theory. Philosophy and Public Affairs 6, 317–344.
Sen, A., 1985. Commodities and Capabilities. North Holland, Oxford.
Sen, A., 1999. Development as Freedom. Oxford University Press.
Sen, A., 2000. The discipline of cost-benefit analysis. The Journal of Legal Studies 29 (2), 931–952.
Spash, C.L., 2006. Non-economic motivation for contingent values: rights and attitudinal beliefs in the willingness to pay for environmental improvements. Land Economics 82 (4), 602–622.
Spash, C., 2008. How much is that ecosystem in the window? The one with the biodiverse trail. Environmental Values 17, 259–284.
Spash, C.L., Hanley, N., 1995. Preferences, information and biodiversity preservation. Ecological Economics 12, 191–208.
Stern, N.H., 2006. The Stern Review: The Economics of Climate Change. Cambridge University Press, Cambridge.
Sugden, R., 1999. Alternatives to the neo-classical theory of choice. In: Bateman, I., Willis, K. (Eds.), Valuing Environmental Preferences. Oxford University Press, Oxford.
Tallis, H., Polasky, S., 2009. Mapping and valuing ecosystem services as an approach for conservation and natural-resource management. Annals of the New York Academy of Sciences 1162, 265–283.
TEEB, 2008. The Economics of Ecosystems and Biodiversity. European Commission.
TEEB, 2010. The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB. European Commission.
Teschl, M., Comim, F., 2005. Adaptive preferences and capabilities: some preliminary conceptual explorations. Review of Social Economy 63 (2), 229–247.
Turner, R.K., 2007. Limits to cost-benefit analysis in UK and European environmental policy: retrospects and future prospects. Environmental and Resource Economics 37, 253–269.
Turner, R.K., Daily, G.C., 2008. The ecosystem services framework and natural capital conservation. Environmental and Resource Economics 39, 25–35.
Turner, R.K., Paavola, J., Cooper, P., Farber, S., Jessamy, V., Georgiou, S., 2003. Valuing nature: lessons learned and future research directions. Ecological Economics 46, 493–510.
Tversky, A., Kahneman, D., 1986. Rational choices and the framing of decisions. In: Hogarth, R.M., Reder, M.W. (Eds.), Rational Choice: The Contrast between Economics and Psychology. University of Chicago Press.
Vatn, A., 2005. Institutions and the Environment. Edward Elgar, Cheltenham.
Wallington, T.J., Hobbs, R.J., Moore, S.A., 2005. Implications of current ecological thinking for biodiversity conservation: a review of the salient issues. Ecology and Society 10 (1), 15.
Walker, B., Pearson, L., Harris, M., Mäler, K.-G., Li, C.-Z., Biggs, R., Baynes, T., 2010. Incorporating resilience in the assessment of inclusive wealth: an example from South East Australia. Environmental and Resource Economics 45 (2), 183–202.
Wegner, G., Howell, K.M., Davenport, T., Burgess, N.D., 2009. The forgotten 'Coastal Forests' of Mtwara, Tanzania: a biologically impoverished and yet important ecosystem. Journal of East African Natural History 98 (2), 167–209.
Wilson, M.A., Howarth, R.B., 2002. Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation. Ecological Economics 41 (3), 431–443.
Winkler, R., 2006. Valuation of ecosystem goods and services part 1: an integrated dynamic approach. Ecological Economics 59 (1), 82–93.