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Economic Environmental Valuation: An Analysis of Limitations and Alternatives

BIOMOT Project Deliverable 1.1



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Preface

The BIOMOT project (MOTivational strength of ecosystem services and alternative ways to express the value of BIOdiversity) addresses the problem of building and sustaining motivation to act for biodiversity by means of a comprehensive rethinking of what value and motivation actually are for people. It has four principal objectives: to establish how economic methods to express the value of biodiversity can be adapted in such a way that they result in stronger motivations to act for biodiversity (Work Package 1); to establish what (economic and alternative) ways to express the value of biodiversity are at work in cases of successful governance and policy action for biodiversity (Work Package 2); to establish what (economic and alternative) ways to express the value of biodiversity are at work in cases of successful action for biodiversity carried out by political, businesses, NGO and other leaders (Work Package 3); and to establish, based on the foregoing results together with philosophical enquiry, a general “theory of motivation to act for biodiversity” and show the practical applications of this theory to enhance biodiversity action in the daily lives and practices of people and institutions at levels ranging from the local to the global (Work Package 4).

This report constitutes the first of Work Package 1’s three deliverables for BIOMOT, and the culmination of the first two of its four Tasks. Tasks 1.1 and 1.2 have consisted of a preliminary exploration and analysis of the concept of Total Economic Value and the role of economic valuation in the context of wider normative perspectives on biodiversity. This report presents the findings of this analysis, in particular focussing on the limitations of standard economic approaches to environmental and biodiversity valuation and how alternative, non-economic valuation approaches avoid these limitations. In Task 1.3, structured interviews are to be arranged with academic economists, applied economists and practitioners (in governments, NGOs and consultancies) that work with economic valuation of biodiversity in Europe. The purpose is to elicit the views of these experts regarding both the weaknesses of monetary approaches to biodiversity valuation and the possibilities and prospects of alternative methods of valuation. The results of Task 1.3 will be presented in the second of WP1’s three deliverables, and together with the findings presented below, will serve as a foundation for its final deliverable, to be entitled ‘Biodiversity Valuation for Biodiversity Action: New Perspectives, New Methods’.

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References

1 Introduction

The explanation for environmental problems has long been claimed by environmental economists as lying in the fact that neither environmental goods nor harms are priced in the market, and that therefore the solution to those problems is to bring environmental goods and harms into the market by pricing them (e.g. Pearce and Moran 1994; see O'Neill 2007: 21 and Foster 1997). Proponents of this view typically argue that while some environmental goods and harms can and have been readily priced by being commodified and brought into actual markets, others require environmental valuation methods for the determination of 'shadow' prices which can then be entered into cost benefit analyses (CBA). It is these latter economic methods for environmental and biodiversity valuation that are the subject of this report. In particular it identifies four respects in which these methods have problems: (i) the way in which they are founded on multiple false assumptions concerning the nature of environmental values and the valuing agent; (ii) their failure to do justice to the variety of our relations to nonhuman nature; (iii) their failure to strike a balance between the pragmatic acceptance of the expansion of market norms into the spheres of valuation and decision making, and resistance to that expansion; and (iv) their motivational capacity. This report examines a number of alternative non-economic forms of valuation which aim to address these problems.

Valuation methods are used by decision makers to appraise options for intervention. Our particular interest is in environmental valuation in general and biodiversity and environmental valuation in particular. Valuation exercises in the arena of environmental decision making typically issue in information on the costs and benefits of different options regarding the protection, management or exploitation of some environmental resource. This information may be in the form of a cardinal ranking of the options with monetary values attached, allowing decision makers to see the degrees of the differences in the net benefits between each option quantified; an ordinal ranking of the options, where no information is supplied regarding the degree of difference between each option; or qualitative information (with no ranking) regarding the costs and benefits of each option, which may itself be disaggregated into the performance of each option according to different criteria. In this report, then, we understand an environmental valuation to be an exercise in appraising two or more options in order to provide

information and guidance for the decision regarding which option to pursue.

The structure of this report is as follows. In Section 2 we will introduce economic methods for environmental and biodiversity valuation, and in particular the total economic value model. Sections 3 through 7 will advance five distinct but related classes of criticisms of economic environmental valuation, and in each we will suggest the way in which alternative valuation methods contain the resources to help address those criticisms. In Section 3 we will advance the first of our five classes of criticisms of economic valuation methods, namely, that such methods fail to respect considerations with regard to spatial explicitness and non-linearity in service provision, and threshold effects in ecosystem functioning. We will then explicate an original method – which we call the *bio-folio* approach – that can be used to modify the economic approach in such a way that mitigates this failure of standard economic approaches.

In Section 4 we will advance the second of our classes of criticisms, which is oriented around the claim that economic valuation fails to adequately register ethical and intrinsic values and the reasons for which people respect these values. We argue that *participatory and deliberative techniques* can go much of the way to accommodating these values and allowing participants in the valuation process opportunities to both articulate their own reasons for holding the positions that they do and to challenge the reasons that others advance for their positions. In Section 5 the criticism that we focus on is that economic valuation methods mistakenly assume value commensurability, and that the broader decision making context of which it is part mistakenly assumes that rational decision making requires value commensurability. We will claim that certain forms of multi-criteria analysis acknowledge the incommensurability of the values involved in environmental decision making and allow that practical judgement has an ineliminable role in environmental decision making.¹

¹ In sections 4 and 5 we have chosen to pair the ethical criticisms with participatory and deliberative techniques and the commensurability criticism with multi-criteria analysis. It should be noted, however, that our demarcation of these alternative valuation approaches is somewhat artificial in the sense that many of these approaches can be and have been combined, both with each other and with standard economic approaches. See eftec (2006: 16) for an instance of an argument in favour of combining valuation techniques, and Stagl (2007) for an enumeration of ‘hybrid methodologies’, such as deliberative monetary valuation (Spash 2001), social multi-criteria evaluation (Munda 2004), three-stage multi-criteria analysis (Renn and Webler 1998), multi-criteria mapping (Stirling 1997), deliberative mapping (Davies et al. 2003), and stakeholder decision/dialogue analysis (Burgess 2000).

In Section 6 we characterise economic approaches to environmental valuation as the ‘itemisation approach’, and advance a fourth class of criticisms of this approach. We firstly distinguish between two kinds of valuing attitudes – de re and de dicto – and argue that the itemisation approach mistakenly assumes that nonhuman nature is principally to be valued in a de dicto rather than a de re sense. Secondly, we argue that economic valuation overlooks the profoundly important temporal dimension of environmental value. Thirdly, we argue that it fails to justify any consideration of certain blocks on whether natural objects or places are substitutable for replicas or replacements. This results in a failure to respect this temporal dimension of value and our de re valuing attitudes. We suggest that participatory and deliberative techniques are well-placed to avoid these criticisms.

In Section 7 we address a key premise of Work Package 1 of BIOMOT: that economic valuation methods are motivationally deficient. We will distinguish two senses in which such valuation methods fail with respect to their motivational capacity. According to the first sense the motivational capacity of a valuation method is a function of the extent to which its output is able to motivate actors to protect or preserve that which is the subject of the valuation. In relation to this sense we will discuss the phenomenon of ‘crowding out’, whereby the introduction of external incentives such as payment diminishes intrinsic motivation. According to the second sense the motivational capacity of a valuation method is a function of the extent to which it has the resources to register the content or nature of existing motivations of different groups to act in pro-environmental ways. Finally, in our concluding remarks in Section 8, we will draw together the improvements to economic valuation methods that our analysis has revealed are needed.

2 Total Economic Value

We will examine four kinds of valuation methods in this report. The first of these – economic valuation – we will introduce now. The other three valuation methods – bio-folio, participatory and deliberative techniques and multi-criteria analysis – will, in different ways, address the weaknesses of economic valuation that we will highlight. We will take the total economic value (TEV) model as the paradigmatic example of economic valuation. It will serve to illustrate the principles of monetary valuation and the methods widely employed to derive these values.

2.1 Introduction to Total Economic Value

The total economic value of an ecosystem is the total net value of the change in the flow of ecosystem services to society occasioned by some marginal change to the conditions of that ecosystem (see Dziegielewska 2009 and Bateman et al. 2010). The value of an ecosystem to humans can only be done by way of a marginal change because of its infinite value in total (we cannot live without it). The changes in which economists are interested in in this context are those brought about by economic developments that affect ecosystems, with the relevant question for society being whether the benefits of ecosystem loss prevention exceed the costs of doing so. TEV is assessed in order that a full cost benefit analysis (CBA) of the proposed project can be executed. The value assessment of the ecosystem is said to be ‘total’ because not only the direct use value of the ecosystem before and after development is assessed, but also its non-use value.

Economists distinguish between use value and non-use value to show that there is additional value apart from direct and indirect use (Pearce and Turner 1990). Use value is further distinguished into direct use value, such as the use of a forest for the timber resources it provides (consumptive) or recreational visits to a nature area (non-consumptive), and indirect use value, such as the indirect use of a wetland as flood control. Non-use or passive values refer to the existence of a value even though individuals do not intend to use the resource but feel a ‘loss’ if it would disappear. This was first introduced by Krutilla (1967) and was contrary to the thinking at the time which focused only on the value of goods and

services from the natural environment if developed. Within the category of non-use values, economists distinguish bequest value, which is based on the value for current generations of being able to pass on an ecosystem to future generations, and existence value, which refers to the value individuals find in simply knowing that a certain ecosystem or species exists, even if they themselves will never see it or derive any direct benefit from its existence. A third category of value is option value, which refers to the way in which we might want to preserve ecosystems and their constituent biodiversity in order that potential unforeseeable future benefits (of any of the above kinds) that might be derived from them are preserved. TEV is the sum of the marginal change in direct, indirect and option values. A standard CBA, into which TEV appraisals typically feed, recommends outcomes that represent potential Pareto improvements based on the Kaldor-Hicks compensation test (Kaldor 1939; Hicks 1939), according to which an outcome represents a welfare improvement if the benefits accrued by the winners are such that they could hypothetically compensate the losers and still be better off. Therefore, if a TEV appraisal of a proposed development project concludes that the sum of the direct, indirect and option values associated with some marginal change in an ecosystem is larger than the costs of preventing this change, a CBA will recommend the proposal.

Since many ecosystem services, and many of the constituents of biodiversity that deliver them, are not priced within the market, economists require alternative methods of determining a monetary value for the benefits they deliver to society that can be fed into a CBA. These methods may be distinguished into revealed preference and stated preference methods (Nunes and Van den Bergh 2001; Fujiwara and Campbell 2011). Revealed preference methods involve observations of choices people make regarding market goods in order to estimate the monetary value of non-market goods. There are two main revealed preference methods. The hedonic pricing method is based on the assumption that non-market goods such as a nearby nature area affect certain market goods, such as house prices. The differences between the prices of houses in close proximity to the nature area and the prices of relevantly similar houses reveals individuals' willingness to pay (WTP) for this particular non-market good. The travel cost method is similarly used to reveal individuals' WTP for non-market goods such as nature areas, in this instance by calculating the costs borne by individuals who visit the area. Stated preference methods

employ questionnaires to elicit individuals' WTP for a non-market good or willingness to accept (WTA) compensation for its loss. Again, there are two main stated preference methods. Contingent valuation methods take a representative sample of the target population and elicit the monetary valuation by presenting them with the scenario of the non-market good being included in a hypothetical market in which they must pay to consume it (or in which they are compensated for its loss). Choice modelling methods present a range of alternative descriptions of the non-market good in which its attributes, including price, are varied. Respondents are then asked to rank these descriptions, and statistical techniques are used to estimate WTP/A.

Pricing techniques such as stated preference models provide information about individuals' WTP or WTA which are summed up in the final balance of CBA. Because practical reasons obviously preclude doing so, stated preference methods will inevitably only provide information about the preferences of specific groups of peoples. Therefore the outcome of a CBA depends on the characteristics of the group of people that is taken as the reference for valuation (in particular their income). The process of placing a monetary value on biodiversity through non-user WTP is performed in the same way as for user WTP, but the identification of people who do not use an environmental good directly but have a legitimate interest in its preservation is obviously more problematic.

2.2 Advantages of Total Economic Value

TEV is a popular concept as it answers the need to understand the economic consequences of environmental management. The economic significance of ecosystems can be expressed through TEV by providing evidence of both the costs and the benefits of environmental regulations. Furthermore, the costs and benefits calculated are expressed as a monetary value, allowing for easy comparisons using a single unit of measure. As such, TEV provides decision support through an improved understanding of problems and the distribution of benefits. Also, in conjunction with the ecosystem services concept, it facilitates managers in taking stock of trends and creating an inventory of actors and relationships for stakeholder analyses.

Even though some cost benefit analyses of environmental changes are criticized as deeply flawed, the exercise is, according to some, redeemed through its value in showing the world how valuable the natural world actually is (Bockstael *et al.* 2000). Whereas few economists believe that information about the net benefits of alternatives should be the sole basis for social choice, many would argue that efficient use of scarce resources should be an important consideration since in most areas of public policy objectives are generally defined corresponding to some mix of the three Es: effectiveness, equity and economic efficiency. An additional argument is that the information about the socio-economic values provided through a TEV assessment would help to develop incentives to preserve biodiversity. For example, the motivation behind the creation of hypothetical markets for ecosystem services is that people as consumers cease to think about the environment as free to exploit and gain a sense of scarcity of ecosystem services. Their preferences, which would hence reflect that sense of scarcity, could then in theory influence the TEV of future marginal changes of ecosystems. This theory forms the basis for designs often through market mechanisms that promote the sustainable use of ecosystem services, such as Payment for Ecosystem Services schemes and wetland or conservation mitigation banking.

While noting these arguments regarding the strengths and advantages of TEV, we will, in sections 3–7, present a series of criticisms of economic environmental valuation.

3 Ecosystem Sustainability and the *Bio-folio* Approach

The criticism put forth in this section is that economic valuation methods used to calculate TEV do not address the sustainable use of ecosystem services. With the introduction of the environment as ‘natural capital’, economics as a profession was given a vision of nature that allowed the profession the use of techniques from mainstream economics to evaluate nature in a way consistent with standard CBA (Heal 2007). Ever since, the associated approach of TEV has encountered criticisms in its attempts to fit characteristics of nature into its methodology. In reaction, Turner *et al.* (2010) has listed five considerations that are seen as critical by economists for ecosystem valuation to be meaningful for a policy perspective. These are: (i) spatial explicitness; (ii) nonlinearity in service provision; (iii) threshold effects in ecosystem functioning; (iv) use of marginal values and macro values; and (v) preventing double-counting. In Section 3.1, each of these considerations will be briefly addressed, and in Section 3.2 the implications for TEV will be explained. In Section 3.3 we will propose a method to take these considerations into account in economic environmental valuation.

3.1 Considerations for Sustainable Use of Ecosystem Services

3.1.1 Spatial Explicitness

The first consideration mentioned is spatial explicitness of ecosystem valuation, because the spatial scale of an ecosystem is arbitrary. An ecosystem is a system with biotic and abiotic elements interacting in a given area, meaning that ecosystem services are provided on local to global scales, and ecological processes play out in different spatial scales, from the micro to the global. Therefore, the scale over which an economic environmental valuation is performed needs to be set in advance. In addition, ecosystems have a spatial heterogeneity, meaning that point estimates of ecosystem service value cannot be aggregated to larger estimates of larger areas. Any valuation of ecosystem services requires a contextual analysis of the local ecosystem and local economic, political and cultural parameters to be appropriate for policy decisions (Toman 1998).

3.1.2 Nonlinearity in Service Provision and Threshold Effects

Considerations of nonlinearity in service provision and threshold effects are a necessity from the lack of predictive power of ecology. Erratic, unforeseen changes can occur in ecosystems, causing potential loss of ecosystem services. These considerations are closely linked to problems with the approach of valuing marginal changes in ecosystems. As long as the locations of thresholds in ecosystems remain unknown, an erratic change might be a single marginal change ahead. The valuation of marginal changes makes economic valuation grant society a snapshot view of the economic output of an ecosystem, while loss of species and ecological adaptability can happen unobserved while ecosystem functioning itself can remain largely unchanged (Scheffer *et al.* 2001; Walker *et al.* 2010). Feedback loops exist in ecosystems, causing the response of an ecosystem to external stresses to play out into the future, through seasons and generations of species. Nonlinearity in ecosystem service provision can also be a result of seasonal changes, of the generation times of the species on which the service depends.

3.1.3 Use of Marginal Values

Considerations of marginal change are linked to both considerations of scale and threshold effects. Policy decisions often concern themselves with trade-offs, which occur at the margin, which makes marginal change a requirement for CBA. Physical changes in ecosystems that are considered marginal changes in ecosystem service productivity differ across spatial scales, which makes a marginal change an arbitrary decision. At local scales, a great physical change may render ecosystem valuation meaningless due to its impact, while on a global scale the same physical change may have only a marginal effect (Turner *et al.* 2010). Furthermore, it is difficult to come to an understanding of whether a given change in an ecosystem is marginal, especially when the locations of thresholds are unknown. A small change may push an ecosystem through a regime change, causing potentially great changes in ecosystem service production.

3.1.4 Double-Counting of Ecosystem Services

Finally, double-counting is a long-recognized consideration for the economic valuation of nature, and a feature of the complexity of interactions in ecosystems. The Millennium Ecosystem Assessment (2005)

divides ecosystem services into four groups, where so-called supporting services such as soil formation and nutrient cycling form the foundation for the production of the other services, namely, provisioning, regulating and cultural. There is a causal chain in place where one service (an intermediate service) causes another service. In economic valuation caution is required not to value the intermediate service separately, and in addition its contribution to a final service benefit. For example, a forest can both supply logging services and ecotourism services, but in reality these services may be competitive where the benefit of logging may cause the ecotourism service to be compromised.

3.2 Inability of TEV to Accommodate the Sustainable Use of Ecosystem Services

From these considerations it follows that economic valuation of ecosystems must incorporate knowledge of the state of ecosystems to provide meaning for policy decisions. In this line of thought, sustainable use of ecosystem services into the future becomes an economic challenge that requires careful monitoring of the state of the ecosystems to maintain their service provision under the pressure of society using these ecosystems. The considerations discussed above can potentially be addressed in economic valuation of ecosystems by using the fields of science that supply us with information on what is required to maintain a sustainable production of ecosystem services.

So far, taking these specific considerations into account has been problematic, and highlights a mismatch between the assumptions out of which TEV is supposed to operate and realities of ecological systems. Economists were forced to deal with the reality that ecosystems behave in erratic ways (Holling 1992; Scheffer et al. 2001; Scheffer and Carpenter 2003), and that this information about the state of the ecosystems is not captured through economic valuation methods theoretically grounded in consumer sovereignty. Economic valuation methods aim to elicit the value of nature to humans and aggregate individual consumer preferences (Bockstael et al. 2000; Farber et al. 2002). However, if consumer preferences do not follow the requirements of ecosystem sustainability, decisions based on TEV will not follow these requirements either (Common and Perrings 1992). Assuming that ecosystem sustainability is preferred by consumers, this criticism claims that there is an information problem, in which consumers lack information about the consequences of actions and decisions on ecosystem sustainability (Chee

2004).

Also, because economic valuation tracks the benefits of a marginal change in an ecosystem, the economic method is unable to predict future erratic behavior and to maintain a state of non-volatile production. Local extinctions and loss of adaptability can happen unobserved while ecosystem functioning itself can remain largely unchanged, leading to unexpected state changes (Chillo et al. 2011; Sundstrom et al. 2012; Walker et al. 2010). The economic valuation methods used to calculate TEV do not address the ecological importance of species and their functioning for ecosystem sustainability.

The motivation behind possible adaptations to TEV to address these shortcomings lies in the commitment that sustainability values are more important than efficiency values around and below the thresholds of ecosystems to maintain ecosystem service production into the future. In order to increase TEV's meaningfulness for policy decisions in this regard, values of sustainability could be addressed when economic valuation is put in a framework that provides a longer time perspective and takes into account ecosystem functioning. We expand on such a framework below.

3.3 Including Ecosystem Sustainability in Economic Valuation through Portfolio Theory

We now address the question of how to combine the concept of economic valuation of ecosystems with knowledge of ecosystem sustainability. First, we address the knowledge of ecosystem sustainability available to use from ecology. Secondly, we propose a method to combine ecological theory with economic valuation of ecosystem services.

3.3.1 Ecological Notions of Ecosystem Resilience

According to Holling (1973), ecosystem resilience is the amount of perturbation an ecosystem can withstand before local extinctions occur and as a consequence cause the ecosystem to shift to another state. Consequently, so-called 'Holling Sustainability' defines an ecosystem as self-sustainable if it can maintain its self-organization through time by adapting to stresses imposed on it (Common and Perrings 1992). In maintaining ecosystem resilience, biodiversity is seen as protection against loss of productivity

or variability of provision. More precisely, as formulated by McCann (2000), biodiversity by itself is not the driver of ecosystem resilience, but ecosystem resilience depends on functional diversity, which is capable of differential response to environmental perturbations. The susceptibility of ecosystems to environmental perturbations is diminished by the presence of a greater variety of functional groups in an ecosystem, because this diversity will lead to the presence of more pathways for energy flow and nutrient recycling (Cadotte et al. 2011; Hobbs et al. 2007; Hooper et al. 2002; Peterson 1998).

Ecological resilience is expected to function across scales as well (Peterson 1998). Ecosystems are responsive to external biological input, such as migrating species and seed dispersal, and are connected through species that serve as linkages (Lundberg and Moberg 2003). So called 'mobile links' maintain a diversity of functional traits within ecosystems on a local scale (Loreau et al. 2002), and so increase ecological resilience by connecting habitats through food web linkages, genetic exchange or other processes. Because of the links between ecosystems, if a species is lost locally, it can be replenished from another nearby area. This way, biodiversity on larger scales functions as an ecological memory that enables disturbed local sites to reorganize (Bengtsson et al. 2002; Lundberg and Moberg 2003).

3.3.2 Insurance Value of Biodiversity and Portfolio Theory

Taking ecological theory into account, an exploration is required of how these ecological theories can be combined with economic valuation. Functional diversity and mobile links have an insurance value in insuring society against loss of ecosystem services through maintaining ecological resilience (Baumgärtner 2007). Insurance value could ideally be expressed as a level of functional diversity, where functional diversity makes up a stock of resilience in an ecosystem (Walker et al. 2010).

Problematically, the locations of thresholds in ecosystems are often unknown and hence the size of resilience stocks in an ecosystem are also often unknown. Therefore, other solutions must be found in which investment in species that are productive for TEV can be combined with investment in functional diversity and mobile links in such a way that a resilience stock is maintained to the best of our knowledge, despite the drive for optimization of TEV. One road to explore is portfolio theory. Portfolio theory explains the economic sense of expanding investment in biodiversity and allows us to select

specific assemblies of functional diversity to take into account in ecosystem management.

The idea behind portfolio theory is to maximize returns while minimizing risk through the creation of a portfolio of investments (Markowitz 1952; 1959). The overall effect of this diversification is a lower volatility in returns. In ecosystem management, using portfolio theory would translate into building a biodiversity portfolio of genes, species or ecosystems with different attributes in order to maximize the TEV of ecosystem services, while managing variability or loss of ecosystem services provision. Such usage was termed '*bio-folio*' by Figge (2004). Portfolio approaches to management compare the trade-offs between variation of economic value through time and the level of TEV for specific portfolios of biodiversity that the ecosystem includes (Figge 2004). In selecting assemblies of biodiversity for a portfolio, investment in biodiversity can be expanded to include a range of species populations that comprise both TEV and insurance value. The theories of functional diversity and mobile link species add information on ecological resilience to the portfolio framework, and can guide a selection of biodiversity whose addition to the portfolio is a necessity for insuring society against ecosystem service loss.

Furthermore, while portfolio theory does not solve the problem of assessing the locations of thresholds, it does allow us to set a limit on optimization of TEV. In choosing a species assembly for a particular location, the level of functional diversity to maintain a resilience stock at that location can be set as a boundary condition for species assemblies under consideration. This limits the number of potential species assemblies under consideration as a portfolio to those that uphold that level of functional diversity. This methodology answers the information problem in TEV, since it allows ecological theory to be a guide for biodiversity conservation in ecosystem management, in addition to optimization of TEV. A wide investment in biodiversity for future returns makes economic sense as explained through portfolio theory, and it allows us to select specific assemblies of biodiversity to maintain ecosystem services into the future (Admiraal et al. 2013).

By putting economic valuation into a framework that provides opportunities for a future-oriented perspective and inclusion of ecological knowledge, economic valuation could gain confidence amongst

the public, policy makers and practitioners that it is a method suitable to value ecosystems in a meaningful way. Of the considerations mentioned in the use of TEV, three of those considerations touch on sustainable use of ecosystem services. Taking these considerations into account is seen as critical by economists for ecosystem valuation to be meaningful from a policy perspective. This perspective is the sustainable use of ecosystem services into the future as an economic and social challenge. People value the continued existence of ecosystems and access to their services, for their own future welfare and for the welfare of their children and future generations. Addressing the sustainable use of ecosystem services requires a perspective that is future-oriented, and inclusion of ecological knowledge for decision making regarding ecosystem exploitation.

4 Ethical Values and Participatory and Deliberative Techniques

We turn now to a criticism of economic methods for biodiversity and environmental valuation that is oriented around the claims that they fail to adequately register ethical and intrinsic values and the reasons for which people respect these values. We will firstly outline two criticisms that are, as it were, external to the mechanisms of the TEV method itself, questioning as they do the very act of asking affected populations their willingness to pay or accept compensation for environmental goods. Next, we will present two arguments that are, as it were, internal to economic methods, claiming as they do that they fail to adequately register the rich normative reasons for which people value natural and cultural environments and their nonhuman inhabitants. Thirdly, we will argue that these failings are due to the particular way in which contingent valuation questions are asked, and the conceptions of rationality and rational decision making that they assume. We will lastly propose that participatory and deliberative valuation techniques have the capacity to meet many of these objections.

4.1 Contested Property Rights and Distributive Concerns

The first of our external ethical criticisms of economic environmental valuation is that it is arguable that many environmental goods – including places which contain and embody biodiversity – should not be treated as objects of liberal ownership which can be acquired and traded (O’Neill 2007: 26–27). To treat such goods as objects of liberal ownership is, among other things, to exercise property rights over them that involve exclusivity and alienability. But such rights are not necessarily appropriate for environmental goods. For example, for an individual or community for whom the goods have been handed on to their care and for their use by their predecessors, and who have hitherto understood that they were to hand it on to the care of their successors, the question of how much they would be willing to accept in compensation to break this chain of bequest, or how much they would be willing to pay to continue it, is justifiably felt as anathema to their sense of membership of a temporal community made up of themselves, their predecessors and their successors. This response – evidenced in protest bids in which respondents refuse to answer – may occur not only with regard to proposals to establish actual markets in which the environmental goods in question become tradable commodities, but also with

respect to hypothetical markets in which respondents are asked such questions in contingent valuation surveys (see e.g. Burgess et al. 1995).

The second of our external ethical objections to economic valuation is that in this approach ‘the poorer you are the less your preferences count’ (O’Neill 2007: 28). Where environmental goods are commodified in actual markets, or where contingent valuation survey respondents are asked to participate in hypothetical markets, poor individuals in affected populations are forced to sell cheaply and thus the preferences they register will be weak.² While it is possible to assign different weights to respondents’ answers in contingent valuation surveys depending on their income and other material circumstances, in practice this adjustment is rarely made. Further, even this response cannot account for protest bids in which the respondent refuses to put any price on the good because they think the question is inappropriate or offensive for reasons of the kind outlined above.

4.2 Constitutive Incommensurabilities and Intrinsic Value

The first of our criticisms that are internal to the methods used in economic environmental valuation relates strongly to the ethical objection that environmental goods should not be treated as commodities to be bought and sold because of the manner in which they can represent or embody certain fundamental values and commitments. It is that the use of money as a neutral common scale for measuring members of the affected population’s preferences is deeply problematic in virtue of the fact that the payment and receipt of money is a social act with certain social meanings and associations. In particular, in certain circumstances, to accept a price is an act of betrayal and to offer a price is an act of bribery (Raz 1986: 345ff; O’Neill 1993: 118–22). For example, to offer money in return for friendship is at best to misunderstand the nature of friendship and at worst to treat the (potential) friend as a commodity which can be bought and sold. With regard to sites of biodiversity, on the one hand many are embodiments of a community’s history and people’s relationships to them can be expressive of respectfulness to their predecessors in that community, and on the other hand environmental concern is often motivated by ethical commitments to one’s children in particular or future human generations in

² This is what Sandel (1998: 94–96) calls the ‘coercion’ of the market.

general. That people understand sites of biodiversity and the social meaning of money in these ways is evidenced by the occurrence of protest responses in contingent valuation surveys, where respondents refuse to put a price on the particular sites. To refuse to put a price on goods that embody or represent commitments to one's community or to particular individuals is an act expressive of those commitments. Even more strongly, to refuse to put a price on goods of this kind is partly constitutive of those commitments, or as Raz (1986: 345) puts it, it is an instance of a 'constitutive incommensurability'. That is, payment and certain commitments are not only incommensurable with one another, but this incommensurability is partly constitutive of the commitment. Economic approaches fail to understand that environmental goods, such cultural landscapes, can strongly represent or embody commitments of this kind, and as such that refusal to place either an actual or hypothetical price on them to express this commitment is a rational response. As such, the standard practice of disregarding protest bids means that economic valuation exercises are illegitimately excluding such value commitments from their data.

The second of our internal criticisms of economic valuation approaches is that they cannot accommodate the intrinsic value of natural environments and nonhuman individuals. One of the central claims articulated and defended by environmental ethicists in the past 40 years has been that nonhuman nature possesses intrinsic value (Callicott 1980, 1989; Rolston III 1986, 1990). There are significant objections to this claim (Light 2002; Morito 2003; Norton 1995; Weston 1996), but we will not examine them here. Rather, since it is a claim that is defended by many environmentalists and articulated in major environmental treaties (e.g. UN Rio Declaration on Environment and Development and the Convention on Biological Diversity), we consider it worthwhile to examine the extent to which economic valuation is able to account for and express this value.

Different authors in the environmental ethics literature have used the term 'intrinsic value' in a variety of different senses. Moreover, these different senses have sometimes been conflated. O'Neill (1992) distinguishes three main ways in which it has been used: (i) to denote the value a state or activity has in itself as opposed to the value it has as a means to some further end (non-instrumental value); (ii) the value an object has solely in virtue of its intrinsic – or non-relational – properties; and (iii) the value an

object possesses independent of its being valued by any valuing agent (objective value). For our purposes here, the relevant senses of ‘intrinsic value’ are (iii) and a particular version of (i). This version has it that for a nonhuman organism, species or ecosystem to have intrinsic value is for it to have value in itself and not merely as a means to the ends of others. It is claimed that if an entity possesses this kind of value then it has moral standing, that is, it is owed moral consideration by moral agents (O’Neill, Holland and Light 2008: 115). The entities that enjoy moral standing vary from theory to theory, from all and only sentient beings (sentientism, e.g. Singer 1986), to all and only individual living beings (biocentrism, e.g. Attfield 1987 and Taylor 1986), to collective entities such as species and ecosystems (ecocentrism, e.g. Callicott 1980 and Rolston 1990). Senses (i) and (iii) are independent of one another; one can reject the notion of objective value – which is a meta-ethical claim regarding the mind-independent, or realist, status of that value – while still maintaining that certain entities possess non-instrumental value, which is a substantive ethical claim.

A central feature of economic valuation means that it cannot be relied upon to register intrinsic value (in either of the two relevant senses). This feature is its restriction to only being able to directly accommodate the preferences of current market agents. Its methods – current market value, stated preference and revealed preference – of providing valuations of nonhuman nature is to measure the strength of the preferences for nature of current market agents. But what if the preferences of current market agents do not reliably track the intrinsic value – in either the moral standing or objective value senses – of nonhuman nature? If this is the case, then economic valuation methods cannot be relied upon to account for and express that value. It is at least arguable (although ultimately an empirical question) that the preferences of current market agents fail to reliably track intrinsic value in either sense. While many environmentalists might believe nonhuman nature possesses value independently of any human valuing agent, it is likely that many individuals do not hold such a belief. Similarly, it is likely that many individuals do not believe that nonhuman nature, or at least a significant subset of it, has non-instrumental value in the sense necessary for it to have moral standing.

Even if these claims are not correct and the preferences of current market agents do indeed adequately

reflect the intrinsic value of nonhuman nature in either or both of these senses, the manner in which the TEV approach to biodiversity valuation accommodates the possibility of intrinsic value leaves it vulnerable to preference changes in populations such that it no longer adequately reflects this value. The idea of existence value in hypothetical valuation is commonly associated with a discrete attractive or compelling segment of the ecosystem rather than with capturing existence as a set of complex ecosystem relations (Vatn and Bromley 1994). But the inclusion of existence value in the TEV framework fails to address this criticism insofar as it remains grounded in the value that the specific element of nonhuman nature has for the agent who values its existence.

4.3 Reason-blindness and the Market Conception of Rational Decision Making

The third set of criticisms we will now present moves on from the above ethical criticisms to the conception of rational decision making that is presupposed by economic valuation approaches. These criticisms can be made clear by firstly considering if all the above concerns could be allayed if it were the case that commitment to the ethical and intrinsic values of nonhuman nature were universal and, further, if the technical apparatus of TEV were routinely adjusted for inequalities amongst respondents and to adequately account for any protest bids to register constitutive incommensurabilities and other kinds of ethical objections. We argue that economic approaches would remain flawed in virtue of the following feature: preference-aggregating methods such as CBA are what O'Neill (2007: 28–32) calls 'reason-blind'. That is, contingent valuation surveys are sensitive to the intensity of the respondent's preferences but cannot record the soundness of the reasons for which those preferences are held. This is due to the core assumption of the model of the consumer behind contingent valuation, which is that we are autonomous utility maximizers.

However, individual choices are influenced by the opportunity for others. There are distinct social aspects to (most) choices and this is exemplified by norms, which are internalised, socially defined solutions when individual acts are competing or in conflict. Thus there is behaviour motivated by individual utility and there is behaviour founded on moral reasoning about the right thing to do. This distinction between the 'I' and the 'We' is parallel to that between consumer and citizen (Vatn 2009).

Individuals might well hold different preferences depending on the two roles. Studies in social choice theory and in public policy have suggested that individuals may have multiple preference orderings and apply different preferences in different contexts. An important implication is that if some, or all, respondents in an economic valuation behave as citizens, the aggregate of their WTP will yield a meaningless money value because TEV is defined as the aggregation across *individual* utility (see Nyborg 2000).

Hence we reject the way in which the utilitarian conception of rationality is dominant in economic approaches to valuation and decision making, and offer an alternative below. To illustrate one reason why it is flawed, consider both the question and the manner of questioning in contingent valuation surveys. The question – how much are *you* willing to pay or accept in compensation for this good? – encourages the respondent to base their response only on the benefits that they receive from the good as a consumer rather than on what, as a citizen, they think society should do with the good (Sagoff 2008: 46–47, 67–86). Further, the manner in which respondents are questioned – individually, privately, without any critical scrutiny from others – makes it easier to base responses in self-interested concerns rather than appeals to the common social good or, indeed, to the intrinsic value of nonhuman nature (Elster 1998; Rawls 1996: 66–71; Goodin 1996). It is far from clear that preferences should be treated as exogenously given and fixed in this way, rather than as open to transformation through interaction with the views of others, nor that all preferences – whether they are rooted in self-interest or in ethical concern – should be treated as on a par with one another in being legitimate candidates for satisfaction by the market or by government intervention justified by appeal to a surrogate market.

A further issue is that commonly economic analysis serves to provide background information to a public debate. This is fundamentally different from a final ranking of policy alternatives as with CBA. While the latter requires that normative views are taken into account, for example regarding equity, a democratic debate requires factual information to be distinguished from normative judgement (Nyborg 2000).

Below, we suggest an alternative model of valuation and decision making that addresses these, and the foregoing, criticisms.

4.4 Participatory and Deliberative Techniques

To summarise the criticisms of economic approaches to environmental and biodiversity valuation advanced in this section, we have claimed that a grounding in the market norm that the role of public policy is to efficiently satisfy preferences, and the assumption that preferences are fixed and exogenously given and not open to rational assessment, underpins the calculative, algorithmic preference-aggregating procedures of CBA and the stated and revealed preference methods to determine the intensity of those preferences. But we have argued that such methods are open to ethical objections of two kinds. Firstly, the methods themselves are morally objectionable in certain contexts in their demand for respondents to consider asserting property rights over goods which they do not consider to be theirs, and in the way that, without adjustment, the methods ensure the distribution of environmental harms will fall disproportionately on the poor. Secondly, economic valuation methods cannot accommodate either protest bids which are partly constitutive of important ethical commitments, and they are vulnerable to neglecting the intrinsic value of nonhuman nature. Further, we have claimed that these methods are flawed in their conception of the status of different kinds of preferences; in their exclusion of the possibility of their transformation; and in their relation of preferences to decision making. We submit that participatory and deliberative techniques can go some way to addressing all of these criticisms.

Firstly, the appeal to participatory and deliberative techniques (PDTs) in environmental valuation and appraisal is an instance of a wider appeal to the normative political theory of deliberative democracy (see Cohen 1989; Dryzek 1990; Miller 1992), which is the view that ‘democratic decision-making should consist not in the utilitarian aggregation of preferences, but in public debate on the public good’ (Jacobs 1997: 221). PDTs thus start from an openness to the view that preferences are not fixed and exogenously given, but may be transformed by exposure to the arguments of fellow deliberators. They also start not from the position that the role of public policy is to satisfy those given preferences, but rather that it is to be formed and emerge from rational debate regarding the ends of the policy, in

addition to how best to meet those ends. This constitutes a conception of rational decision making that is an alternative to the market-based, instrumental rationality of economic approaches, namely, a forum-based, procedural rationality, whereby a rational decision is the outcome of fair procedures and deliberation which meets the norms of rational discussion.

Secondly, PDTs are not reason-blind. Instead of each respondent undergoing private questioning, groups of respondents, chosen either at random or selected to be representative of the affected population, are brought together to deliberate the proposal prior. Consider the PDT that takes the form of a citizens' jury (see Stewart et al. 1994), which is a group of 12–25 individuals, aided by an impartial moderator, who receive information and hear arguments from both neutral and interested witnesses, then deliberate and vote on the proposal. The participants are asked to explain their reasons for the votes (Jacobs 1997: 222). The public nature of the debate ensures that the reasons offered must meet the 'publicness condition' (O'Neill 2001: 484). For a reason or argument to meet the publicness condition it must be able to survive being made public, and as such the imposition of this condition in deliberative fora (i) makes it less likely that appeals to self-interested concerns will be made and (ii) makes it more likely that wider and more general ethical concerns regarding nonhumans and future generations are made, thus ensuring their interests are represented (O'Neill 2007: 180; Goodin 1996: 846–847). Moreover, as Kenyon et al. (2010: 222) put it in order to contrast this method with contingent valuation surveys, citizens' juries ask the 'right question', namely, what, in these circumstances, is best for society? That is, instead of the private questioning of contingent valuation surveys eliciting the views of the respondents *qua* consumer regarding her willingness to pay, citizens' juries encourage participants to answer the question *qua* citizen (Aldred and Jacobs 2000). The equivalent of protest bids, which render the reasons for a zero bid invisible, can be articulated in this context. Moreover, contingent valuation methods mistakenly assume that respondents have pre-existing private preferences for environmental goods which the survey elicits. Citizens' juries, by contrast, allow participants to form new preferences and subsequently adjust them in the light of reasoned dialogue (Söderholm 2001: 489).

Potential problems with PDTs should not be overlooked. In particular, similar concerns to the way in which income inequality can affect the results of unadjusted WTP surveys, so too can inequality regarding the capacity and confidence of participants in deliberative fora to contribute to the discussion and articulately and cogently express their positions (O'Neill 2007: 149–150). A second problem concerns the legitimacy of the claims of participants to speak on behalf of those they are representing, including marginalised groups, nonhumans and future generations (ibid.: 181–184). Addressing both of these problems and ensuring that all groups have equal representation and equal capacity to participate in deliberative institutions is likely to preclude any convergence on a given option in many cases, since environmental conflicts involve a plurality of values such that it may not be possible to realise all values in all cases. This need not be seen as a weakness of well-functioning PDTs, but rather a strength insofar as it reveals conflict instead of silencing dissenting voices in order to manufacture consensus. It is to the issue of value plurality and conflict to which we turn next.

5 Value Incommensurability and Multi-criteria Analysis

The TEV approach to biodiversity valuation is intended to enable the rational resolution of value conflicts by feeding information regarding the preferences of affected populations into the decision making procedure of CBA. But we will argue here that this approach is mistaken in two fundamental respects. Firstly, it mistakenly assumes both the commensurability of all values associated with biodiversity and of biodiversity values with other social values. Secondly, CBA is motivated by the mistaken assumption that the rational resolution of value conflicts requires value commensurability. We will examine each mistake in turn, before turning to the ways in which certain forms of multi-criteria analysis may form a component of valuation and appraisal methods that avoid these mistakes.

5.1 The Mistaken Assumption of Value Commensurability

Firstly, we will show how economic methods assume environmental values are commensurable with one another and with other, non-environmental values, and argue that this assumption is mistaken. For our purposes here, and following Hsieh (2007), we say that abstract values (e.g. equality or beauty) are commensurable or incommensurable with respect to one another, and concrete bearers of value (e.g. sites of biodiversity or public policy decisions) are comparable or incomparable with respect to one another. In cases of practical conflict, if two different outcomes – say, the outcome in which a site of biodiversity is left alone and the outcome in which a portion of it is destroyed for some economic development – are to be comparable with one another, it must be the case that the various values they bear are commensurable with one another. Values may be strongly commensurable or weakly commensurable with one another. For strong value commensurability to obtain, the thesis of value monism must be true. Value monism is the thesis that apparently different kinds of value, such as knowledge, companionship and justice, are reducible to one ultimate value. This ultimate value enables the strong commensurability of the apparently different kinds of value by providing a unique best ranking of them (O'Neill 1993: 103). We reject the thesis of value monism and therefore hold that strong value commensurability does not obtain. Value pluralism – which maintains the irreducibility of central values – is a considerably more plausible thesis than value monism. This is the case because there are two

problems that any candidate ultimate value must overcome in order for value monism to be established. Firstly, it must be shown that there are no other values that should be accorded anything other than contributory or instrumental value status to that candidate ultimate value. Secondly, even if this challenge could be met, it must be shown that the candidate ultimate value is not itself merely a covering value for an irreducible plurality of components. For example, the assumption underlying economic theory and its valuation approach is welfarist, i.e. that value monism is true and the ultimate value to which all other values are contributory is human well-being. But theories of human well-being are either pluralist or monist regarding its content. If they are pluralist – claiming that human well-being consists in a plurality of welfare goods such as knowledge, friendship and pleasure – then value monism is false. This leaves monist theories of human well-being, but such theories are themselves implausible, since candidate welfare goods to which all other goods must be understood as contributory are irreducibly plural in character. For example, as O’Neill, Holland and Light (2008: 75) argue with regard to the commonest candidate value for monist theories of human well-being, namely, pleasure,

contrary to what is commonly assumed, pleasure is not a single value. A long cool drink at the end of a long hot walk, a conversation with a good friend, watching buzzards wheeling in the sky, achieving a longtime ambition to run a marathon within a particular time – all are a source of pleasure, but they do not share in some common property of pleasurable which we can add and subtract. There is a prima facie case for value pluralism even from within a hedonist perspective.

Even considering the possibility of strong commensurability within the limited context of environmental decision making alone reveals its implausibility. It is a strongly arguable proposition that there are a plurality of values associated with biodiversity, for example, aesthetic value, scientific value, recreational value and cultural heritage value (see Takacs 1996) This pluralism is made clear in contexts in which these values conflict. For example, managing a biodiversity site such as a coppice woodland in such a way that its cultural heritage value is preserved may conflict with its management for a particular rare species, and managing it for either purpose may conflict with its possible management as a productive timber resource. To understand what conservationists are trying to achieve in such a context as the maximisation of a single ultimate value – let us call it ‘environmental value’ – is to grossly misrepresent

their endeavours. Value monism does not obtain within the environmental domain, let alone with respect to the wider sphere in which environmental decisions are made, and therefore strong commensurability and the possibility of determining unique rankings of options by reference to a single ultimate value is ruled out.

For weak commensurability to obtain it must be the case that ‘while there may be no single value in terms of which all states of affairs and objects can be ranked, there does exist a single comparative term in terms of which they can be ordered’ (O’Neill 1993: 104). That is, this form of value commensurability can obtain even if the thesis of value pluralism is true, and the TEV approach to biodiversity and environmental valuation may still be viable. The single comparative term, such as ‘better than’, ‘at least as good as’ or ‘is more valuable than’ can be understood as allowing us to trade-off a given amount of one value against another. For example, we might say that a loss of a certain amount of aesthetic value can be compensated for by a certain amount of gain in cultural heritage value. CBA assumes the weak commensurability of values insofar as it is ‘committed to the existence of a single measure that orders all objects and states – persons’ willingness to pay at the margin for the satisfaction of their preferences’ (O’Neill 1993: 110). Monetary value provides the common scale against which different biodiversity and other social values can be measured. The TEV approach to biodiversity and environmental valuation assumes we can rank the outcomes of decisions and policies in terms of the degree to which they satisfy preferences, and the method it uses to measure aggregate satisfaction is individuals’ willingness to pay at the margin for these goods in either actual or hypothetical markets.

The argument we presented above in Section 4.2 regarding constitutive incommensurabilities and the social meaning of acts of monetary exchange already provides one problem for the approach of using money as the measuring rod of preference satisfaction. A further problem is that if different attributes of an environmental good are incongruous – that is, attached to orthogonal dimensions – one metric (monetary value) will be unable to capture all relevant information. As was argued in Section 4, certain kinds of moral commitments that inform biodiversity values cannot be reduced to the satisfaction of the preferences of current market agents. This moral aspect of environmental changes introduces one

important basis for such incongruity that restricts trade-off possibilities and alters perceptions of value in ways that differ from the comparative calculus at the margin (Vatn and Bromley 1994). Thirdly, the metaphor of the trade-off misrepresents the way in which value conflicts and decision making in environmental contexts is, and should be, approached. Again, conservationists do not understand their efforts to be a matter of maximising a certain set of values by comparing the values realised by each option against a common scale. This is a peculiarly consequentialist framing of the problem which omits entirely other approaches which are theoretically richer and more psychologically credible, such as considering which particular obligation and claims have the strongest importance in a given context or which virtues a community wishes to exhibit (O'Neill, Holland and Light 2008: 81). These alternative accounts do not presuppose weak value commensurability or the framing of value conflicts as necessitating trade-offs.

5.2 The Mistaken Assumption that Rational Decision Making Requires Value Commensurability

The second mistake which we argue the TEV approach to biodiversity and environmental valuation makes is that the decision procedure into which the TEV model feeds – that of CBA – is founded on the mistaken assumption that the rational resolution of value conflicts requires value commensurability. As O'Neill, Holland and Light (2008: 71) claim,

Part of the promise of...standard economic decision-making tools such as cost-benefit analysis, is that they offer a procedure for resolving value conflicts through the employment of a common measure which allows the losses and gains within each option to be aggregated, and then the total value offered by each option to be compared. It promises the possibility of reducing social choice to a matter of a calculus – a method by which anyone, given a set of data about the outcomes of alternative actions, can work out mechanically which outcome is best.

If values are incommensurable, it is assumed, then rational decision making is not possible. But this assumption is mistaken. Firstly, there are alternative accounts of what makes decisions rational which do not require commensurability. A procedural account of rationality holds that a rational decision is one which is the outcome of deliberation that meets the norms of rational discussion (O'Neill 2007: 30; see also Simon 1979: 68). An expressive account of rationality holds that a rational decision is one which

‘adequately express one’s rational attitudes towards the people and things one cares about’ (Anderson 1993: 18). Both of these accounts of rationality are consistent with and can be accommodated by deliberative techniques. However, our more general claim here, and the one that is most pertinent to the alternative valuation and decision making tool that we will introduce below, is that value incommensurability does not preclude rational decisions in contexts of conflicts between those values. The alternative, as O’Neill (1998: 126) characterises it, is to ‘use your judgement after the best possible deliberation’. This judgement should be tutored and informed, and based upon developed capacities of perception and knowledge founded in education and experience (O’Neill 1993: 117). This approach is neither calculative nor algorithmic, but deliberative. O’Neill, Holland and Light (2008: 85) argue that alternatives to algorithmic decision making recognise that there may not be a complete ordering of lives when engaged in choosing between policy options such that we can identify the best option, the second best option and so on, but only a partial ordering whereby what we have is ‘a set of admissible solutions which themselves are not ordered. It is possible that one might simply have a variety of options, each with their own bundle of goods, each coherent and making sense, and with no ordering between them.’ We suggest below that some forms of multi-criteria analysis recognise this and allow room for the exercise of deliberation and judgement.

5.3 Multi-criteria Analysis

Multi-criteria analysis (MCA) is an approach and a set of techniques to appraise different options for achieving objectives and thereby to serve as an aid and support to decision making.³ MCA may be used if the desire is to find a single best option, but also if it is only required to draw up a shortlist of options for further consideration, or to rank or categorise options (Munda 2008: 57). A typical MCA proceeds by first outlining the objectives, and then distinguishing a set of options for achieving these objectives. Next, relevant criteria for measuring the extent to which each option achieves these objectives are specified, and then each option is assessed against these criteria. This exercise results in a performance matrix, with rows for each option and columns for each criterion, with the performance of each option entered in the cells. The performance data may come in the form of objective measures

³ The description of MCA in this section is based on Chapter 4 of Department for Communities and Local Government (2009).

such as distance or weight; observed measures such as current price; or informed, expert opinion. This data may be merely in binary terms, where it is indicated if a criterion is satisfied or not, but it is often converted into cardinal numbers, typically on a scale of 1–100.

The first point to note about MCA, therefore, is that in distinguishing performance criteria it explicitly acknowledges the plurality of dimensions of value involved in environmental decision making. However, while we consider this to be a virtue in relation to the monism of economic approaches, many forms of MCA do not take these values to be incommensurable, since MCAs may be *compensatory*, in which trade-offs between the criteria are undertaken in order to provide information on how each option performs overall and in relation to each other. These require a numerical scale to be established for each criterion to measure the performance of each option, and an importance weighting to be assigned to each criterion. A simple weighted average of scores for each option can then be calculated (assuming that the score for each option can be judged independently of the score for any other criterion). Further, trade-offs between different criteria can be undertaken, where a poor performance on one criteria may be compensated for by a good performance on another. To conduct and analyse these trade-offs, MCA techniques use various mathematical methods to aggregate the scores of each option on all the criteria.

But other forms of MCA do not assume value commensurability. These *non-compensatory* forms attempt no aggregation or trade-offs between the different criteria (Martinez-Alier et al. 1998: 283-284). Non-compensatory techniques may be used if trade-offs are blocked on ethical grounds, for example. Moving to a compensatory MCA may also be deemed unnecessary if one option dominates all others, i.e. if one option performs at least as well as any of the others on all criteria and better on at least one criterion. However, if, as in most complex cases, there is no dominant option, and if trade-offs are not inappropriate, compensatory MCA techniques are normally used.

The improvement over standard economic methods that MCA embodies resides not only in the extent to which it acknowledges value plurality and (for some forms) value incommensurability, but also the

extent to which the judgement of those carrying out the MCA is called upon at many points in the process. For all forms it is called upon in identifying the objectives and specifying and distinguishing the performance criteria; in specifying measures for the performance of each option according to each criterion; in grouping or clustering criteria; and in judging if double-counting is occurring and how to avoid it. Further, in compensatory forms it is called upon in the task of weighing the importance of the criteria in relation to one another. In forms where it is decided not to assign numerical scores and weights to the criteria, a decision maker may also be required to use his or her judgement in assessing the performance matrix and the mixture of performance data it contains in order to evaluate the options. Of course, the development of MCA is a response to the shortcomings of humans' capacity to process significant quantities of information mixed across qualitative and quantitative dimensions, so it must be acknowledged that all of these kinds of judgements are prone to the biases and cognitive limitations which MCAs are designed to overcome. But while we acknowledge this we do not consider the exclusion of all judgement and its replacement by algorithmic procedures to be an appropriate response, nor do we accept the assumption implied by this response, namely, that judgement under conditions of value incommensurability cannot be rational. For example, we have already seen that it is reasonable to judge as preferable an option which performs at least as well as any of the others on all criteria and better on at least one criterion.⁴ But it is not clear that value pluralists who take incommensurability seriously should be barred from making the reasonable practical judgements even in cases where there is no dominant option. Neither should it be assumed that to do so requires assigning weights to the criteria in advance and abstracted from a careful consideration of the manner in which each option performs such that trade-offs can be engaged in. To say in advance, for example, that criteria relating to aesthetic value should count for half as much as criteria relating to cultural heritage value is of dubious worth, since there is a plurality of ways in which aesthetic and cultural heritage values can be realised. What is required is a careful assessment of the particular options in question, including the ways in which they realise different values, which in turn can subtly influence the way in which these different values should be weighted. For example, the form of MCA known as 'multi-criteria mapping' (Stirling 1997) allows individual specialists and stakeholders to revise their weightings of each criteria, as well as the

⁴ See Sen (1993: 32) on 'dominance' ordering.

scores they have assigned, in light of the ranking outcome of their initial efforts. This form of MCA is combined with forms of PDTs in a process called ‘deliberative mapping’ (Davies et al. 2003), which facilitates deliberation and mutual learning between lay persons and specialists. That MCA allows room for preference transformation and judgement is one of its strengths, and one of the ways in which the criticism of economic methods developed in this section can be addressed.

6 History, Substitutability and De Re Valuing Attitudes

The fourth class of criticisms of economic approaches to biodiversity and environmental valuation that we will advance concerns a particular characterisation of economic approaches to environmental value which reveals some further aspects of the economic valuation which we wish to draw attention to.

O'Neill and Holland (1999: 460) outline this 'itemisation approach' as follows:

A list of goods is offered that correspond to different valued features of our environment, such that increasing value is a question of maximizing one's score on different items on the list, or at least of meeting some 'satisfying' score on each. We have something like a score card, with valued kinds of objects and properties, valued goods, a score for the significance of each, and we attempt to maintain and where possible increase the total score, the total amount of value. The approach involves a form of consequentialism: we assess which action is best in a given context solely by its consequences, by the total amount of value it produces.

We will argue that this approach (i) misunderstands a central way in which people value nonhuman nature, (ii) overlooks an important dimension of environmental value, and (iii) justifies policies which fail to respect this dimension of value. While there is not, as for the criticisms advanced sections 4 and 5, a class of established alternative valuation methods and decision tools that can address those criticisms, we suggest that PDTs are better equipped than standard economic valuation to avoid them.

6.1 *De Re and De Dicto Valuing Attitudes*

The first of our expressions of the way in which the itemisation approach, and by extension the economic approach to environmental valuation, misunderstands the value of that which forms the subject of their valuations appeals to a distinction between *de dicto* and *de re* attitudes. An attitude can be a belief, desire, valuing, emotion, or other mental state directed towards some object or state of affairs. *De re* attitudes are about particular things, whereas *de dicto* attitudes are about types of thing that satisfy descriptions. The difference can be most clearly seen by considering some examples. Take the claim:

1. John believes that the chief executive of News Corporation is rich.

On a de re reading, John believes of a particular person (Rupert Murdoch) that he is rich. On a de dicto reading, what he believes is that whoever is the chief executive of News Corp is rich. These beliefs are quite different: John might have acquired the de re belief from having met or knowing facts about Rupert Murdoch; he might have acquired the de dicto belief from just knowing facts about chief executives and News Corp (without knowing anything about Rupert Murdoch in particular). Notably, we are unable to determine simply from John's assertion that he believes that the chief executive of News Corporation is rich whether he has the de re or de dicto belief. In other words, what he says is ambiguous and needs to be disambiguated by further questioning.

The problem of ambiguity can be seen in the following example. How should we interpret:

2. Ralph values a site of high biodiversity.

Does Ralph value a particular site (one, perhaps, with which he is personally acquainted), or does he just value a site that happens to satisfy the description 'high biodiversity' (rather than any particular site)? The verbal report (2) does not provide enough information to determine whether the de re or de dicto reading is appropriate. The difference can also be expressed in predicate logic. On the de dicto reading, we should analyse (2) as:

3. Ralph values $\exists x$ (x is a site of high biodiversity).

On the de re reading we should analyse (2) as:

4. $\exists x$ (x is a site of high biodiversity and Ralph values x).

Many of our ethical values and motivations seem rooted in de re concerns (Smith 1994; Williams 1981; Hare 2007): it is particular persons, for example, that seem to be the focus of many of our moral

interests and subsequent motivations to act to help. Bernard Williams gives the following example: suppose that your spouse and a stranger both fell into a river putting their lives at risk. One does not normally, faced with such a situation, consider ‘what would be the right thing to do in these circumstances’; instead, one is motivated to act to save one’s spouse. That is, one is motivated by a *de re* concern for one’s spouse rather than a *de dicto* concern about people in general and the right thing to do. To be motivated by a *de dicto* concern in this circumstance would be more characteristic of someone who is *alienated* from people (Railton 1984). In contrast, we normally have *de dicto* concerns about fungible commodities to which we have no distinctive historical or emotional ties: money, tools, and so on. We usually value tools for the jobs that they perform, and money for the use to which it can be put. Plausibly, the way in which people often value natural and cultural environments is based on their personal experiences and ties to particular places that they have encountered: the place where they were brought up, the woodlands where they roamed as a child, the places that have some special historical or cultural significance for their community. Approaches that treat biodiversity as a fungible resource and its value in a *de dicto* sense fail to capture this essential component of our relationship to the environments and the value we find in them.

The economic approach to biodiversity and environmental valuation is one such approach; it mistakenly assumes that biodiversity is to be valued, and is as a matter of fact principally valued, in a *de dicto* sense rather than a *de re* sense. This assumption is most clearly seen in the economic notion of ecosystem services. The Millennium Ecosystem Assessment (2005) defined ecosystem services as ‘the benefits people obtain from ecosystems’. Fisher and Turner (2008) expanded on this definition and proposed the standard definition used in economic studies that ‘ecosystem services are the aspects of ecosystems utilized (actively or passively) to produce human well-being’. Thus if there are no takers there is no economic value. It is the delivery of a service to humans such that their well-being is maintained or enhanced that is of value, and people’s preferences can be relied upon to reveal this value.

More generally, the itemisation approach to environmental values and what it is to maintain or increase them rests on the mistaken assumption that the value we find in nonhuman nature can be captured by a

list of physical attributes which ensure the delivery of certain services to human society. Rather, we argue, we also value particulars: particular woodlands, particular rivers, particular meadows, even particular nonhuman individuals. Given this, it is not the case that all particulars which instantiate a certain valuable set of physical attributes will be equally valuable to us. In particular, as we will argue below, it will not be the case that there will be no loss of value if a valued particular - such as the woodland where local people regularly take an evening walk - is substituted with a new woodland albeit with the same physical attributes, such as the same species composition and structure. In the next section, we provide one reason why this might be so which appeals to the role of history in our appreciation of nonhuman nature, a role which the itemisation approach overlooks, before exploring further the way in which these arguments suggest that certain blocks on substitutability.

6.2 The Role of History in Environmental Value

The itemisation approach leaves out a dimension of profound significance in our relations to and valuing attitudes towards particular environments, namely, the temporal dimension. The itemisation approach is static and ahistorical, and as such contains no reference to, and no mechanism with which to register, the importance of time and history. As O'Neill and Holland (1999: 462) argue:

Time and history do not enter into problems of biodiversity policy solely as technical constraints on the possibility of recreating certain landscapes with certain physical properties; for example, particular habitats are valued precisely because they embody a certain history and processes. The history and processes of their creation matter, not just the physical attributes they display.

A distinction which helps to illuminate this point is between an end-state view of the value of nonhuman nature and a process-based view. According to the end-state view, what matters is only the state of the biodiversity or ecosystem insofar as it affects the delivery of ecosystem services. If two constituents of biodiversity or two ecosystems are in the same state and therefore are able to deliver a service as effectively as one another, then their value is equivalent. That is, the level of services provided by this biodiversity within a given period can be seen as a 'flow' extracted from an underlying 'stock' of biodiversity assets. This is a purely *de dicto* evaluation of biodiversity: constituents are valued not as particular objects, but for those properties they possess that enable them

to deliver relevant services. However, this view of environmental value misses the value that is to be found in the history and processes which brought that state about. Environments embody both natural and cultural histories (O'Neill 2007: 86). The fact that a certain place is a product of certain historical processes, be they geological, ecological or cultural, and the way in which that place embodies that history and those processes, is often one of the sources of its value. The itemising approach misses this significance.

6.3 Natural Capital and Substitutability

The way in which the economistic approach to environmental values misunderstands the importance of time and history, and consequently what it is to respect these aspects of environmental value, can most clearly be seen in the increasing dominance of the notion of 'natural capital'.⁵ Capital can be either natural (e.g. resources, genetic information, ecosystem functions) or manmade (e.g. infrastructure, skills, knowledge). There is a debate within the literature on sustainability regarding the degree to which we ought to allow manmade capital to substitute for natural capital in what we bequeath to our successors. Our concern, however, regards the degree of substitutability *within* the category of natural capital, i.e. whether, for example, we can substitute one woodland or wetland for another without loss of value. The criterion which determines the strength of the constraint on substitutions is, according to this approach, technical and practical feasibility. That is, the block on the substitutability of some items of natural capital, such as an ancient woodland, is merely a function of the technical and practical difficulty of the manmade replication or recreation of them. If an item of natural capital cannot be replaced, with all its physical attributes, in a reasonable timeframe, then it is regarded in this framework as non-substitutable. If it can, then substitution is justifiable, since this will leave the itemised list of valued objects and properties unchanged.

Biodiversity offsetting provides an example of this position. It has been developed in a number of countries, and appears poised to become part of mainstream planning policy in England. The policy essentially consists of the requirement for developers, after they taken all reasonable measures to avoid,

⁵ See, for example, the recent (2013) report of the UK Government's newly-established Natural Capital Committee.

minimise and mitigate biodiversity losses on their development site, to compensate for or offset those losses which are unavoidable by securing habitat expansion or restoration elsewhere.⁶ Biodiversity losses on the development site are quantified as biodiversity units, and the number of units that those losses score is a function of the area of those losses; the condition that the habitat was in, rated as poor, medium or good; and the distinctiveness of the habitat, rated as low, medium or high. Then the proposed habitat creation or restoration scheme to offset these losses has its biodiversity units calculated, which is a function of its area, distinctiveness and condition, but also of the risk that the scheme will not come to fruition, the length of time it will take to come to fruition, and how well-integrated the project is in the wider biodiversity strategy in the region. All sites of biodiversity are assumed to be – or at least made to be – commensurable with one another, and it makes them so by assigning each a simple numerical score of biodiversity units. One biodiversity site is thus substitutable for another.

This reveals the extent to which the economic understanding of the value of nonhuman nature results in the absence of any in principle blocks on the substitutability of particular natural objects and places. But, in principle, replicas of particular natural objects and places cannot bear the same history of that which they are intended to replace. A newly planted woodland, for example, cannot bear the same value as an older woodland which may have a significant history of association with neighbouring communities. It will take not only time for the newly planted woodland to become a place at all, and for members of the community who frequent it to develop the meaningful relationships with it that make it a place, but it can never be the same place. Landscapes and particular places within them can be constitutive of social bonds over time to the temporal communities of which we are members. The replaced woodland may have been worked by the predecessors of those members of the local community who currently visit it, and it may only be the woodland which enables this sense of community to be felt and understood. This will also bear on its aesthetic value: landscapes which embody history, both natural

⁶ See Defra (2011) for details, and Hannis and Sullivan (2012) for a recent critical overview.

and cultural, and which evoke a sense of that history, as Alan Carlson (2002) argues, bear thick, expressive aesthetic qualities in addition to their thin, formal aesthetic qualities. The newly planted woodland has to rely on its formal aesthetic qualities alone. A richer and more psychologically plausible understanding of our relations to nonhuman nature reveals that however technically proficient at replicating the physical attributes of ecosystems we become, justifiable blocks on substitutability will remain.

6.4 Requirements of Valuation Methods to Accommodate De Re Valuing Attitudes

We have argued here that the itemisation approach to environmental value, which is one way to characterise the economic approach, mistakenly assumes that nonhuman nature is principally to be valued in a de dicto rather than a de re sense, that it overlooks the profoundly important temporal dimension of environmental value, and that it justifies the biodiversity offsetting policy which allows no principled blocks on the substitutability of particular natural objects or places and thus fails to respect this temporal dimension of value and our de re valuing attitudes. As O'Neill (2007: 88) argues, 'decision procedures like cost-benefit analysis are apt to foster forms of development and planning that are ahistorical and placeless.' What is required, then, are valuation methods which are able to accommodate appeals to de re valuing attitudes; to the importance to communities of the way in which the narrative and history of their predecessors are embodied in cultural landscapes; and to the way in which offset schemes, however faithful to the habitats they are attempting to compensate for, cannot replicate the places that are lost which embody those histories precisely because the replicas cannot in principle have the same histories. PDTs that provide opportunities for representatives of affected populations to articulate these concerns can avoid the foregoing criticisms of economic valuation methods. Further, an acknowledgement of history, narrative and process allows an alternative conception of how to frame environmental decision making. The itemising approach has it that the goal of environmental policy is to maximise the value of the individual goods on our list. A narrative approach changes this to a question of how it would be most appropriate to continue the story of a place, given its history and narrative up to the present (O'Neill 2007: 143). Economic, unlike deliberative, approaches provide no forum in which such a rich question could be considered.

7 The Motivational Failure of Economic Valuation Methods

Our concern in this section is to consider the *motivational capacity* of economic valuation methods. It is a premise of the BIOMOT project that despite increasingly sophisticated economic valuations consistently presenting substantial monetary figures for environmental goods, policy makers and publics remain insufficiently motivated to act in such a way that these goods are adequately protected, as evidenced by the fact that after many thousands of valuation exercises claiming that biodiversity and other environmental goods are worth the equivalent of substantial monetary sums, biodiversity loss in Europe continues (European Environment Agency 2010). This leads to the hypothesis that there is something deficient about the motivational capacity of standard economic valuation methods. We will distinguish two senses in which economic methods may be said to be flawed with respect to their motivational capacity. Before doing so, we wish to make clear that when commenting on the motivational capacity of a valuation method there must be clarity on precisely *who* it is intended or expected to motivate. Many valuation exercises are typically only seen by – and are only intended for – decision makers, and not the public. However, some are used also more generally in public discourse as ways of displaying the value of the natural world in monetary terms, with the aim of increasing wider motivations to preserve biodiversity. These two audiences need to be kept distinct.

7.1 Intrinsic Motivation and Crowding Out

Our first understanding of the motivational capacity of a valuation method is that it is a function of the extent to which the quantitative or qualitative assessment or summary of the value of a particular environmental resource is apt to motivate those who are presented with it to act for the protection of the resource in question. That is, it is the *output* of the valuation exercise that has the capacity to motivate pro-environmental actions. In the case of economic methods, that output is typically a monetary figure. Of course, the way we characterised valuation exercises above was that they consist essentially in an ordering of options. Understood in this narrow way of an ordering of a restricted set of options without the content provided by a monetary figure or any other quantitative or qualitative information, we should

not expect a valuation to have the power to motivate those appraised of its assessment to act for biodiversity in general or even the environmental good that is the subject of the valuation. However, monetary figures are used to cardinally rank the options under consideration according to the degree to which they satisfy human preferences, thus providing content to the ordering that could plausibly have motivational power. Indeed, it is clear that those who commission and undertake valuation exercises are hopeful that the large monetary figures attached to the marginal changes of the particular environmental goods and resources that are the subject of their valuations will make clear not only the value of that environmental good but also the value that environmental goods have in general to our economic well-being, and will thus motivate actors to protect the good in question and behave in pro-environmental ways in general. However, now that we have identified one understanding of the motivational capacity of economic valuation methods to be a function of the monetary figures they issue in, we must consider a general criticism of the motivational capacity of standard economic instruments with regard to pro-environmental actions known as ‘crowding out’. We will introduce crowding theory in general, before returning to the implications for economic valuation methods.

The claim here is that *extrinsic* motivations can ‘crowd out’ *intrinsic* motivations. ‘[O]ne is said to be intrinsically motivated to perform an activity when one receives no apparent reward except the activity itself’ (Frey (1997: 13), after Deci (1971: 105)). In contrast, one is said to be extrinsically motivated to perform an activity when one acts to receive a reward beyond the performance of the activity itself. Frey (1997) argues that the ‘use of monetary incentives [as well as external interventions other than rewards, such as commands and regulation] *crowds out* intrinsic motivation under identifiable and relevant conditions’ (p. xi). In contrast to psychologists, who place great emphasis on such motivation in explaining and predicting behaviour, economists, Frey claims, attach little importance to it, being ‘convinced that incentives applied from the outside are more important, and by far the greatest motivator is money.’ The (relative) Price Effect is a central component in economists’ explanations of behaviour. As Frey explains (p. 20), it ‘states that human beings *ceteris paribus* increase an activity whose reward or price, in comparison with other relevant prices, rises.’ The claim that increases in monetary incentives may decrease participation in an activity previously sustained by intrinsic motivation

is therefore in direct contradiction to the principle of the Price Effect.

Frey (pp. 16–17) distinguishes three related and overlapping psychological processes to which the phenomenon of the crowding out of intrinsic motivation by extrinsic incentives may be attributed. Firstly, individuals' intrinsic motivation may diminish when they perceive their self-determination to have been reduced by an external intervention such as the introduction of a monetary incentive. Secondly, intrinsic motivation may be undermined when an external intervention is perceived to imply a lack of acknowledgement of the intrinsic motivation that previously sustained the activity, thus damaging the individual's self-esteem. Thirdly, an individual may abandon their intrinsic motivation in response to being deprived of the opportunity to exhibit their intrinsic motivation to others.

Frey posits (p. 18) that these psychological processes that provoke the crowding out effect take place when particular psychological conditions obtain, namely, that the individual has a subjective perception that the external intervention is *controlling*. He further distinguishes eight conditions which determine whether an external intervention may be perceived as controlling (pp. 26–33), which include where the task is interesting for the agent, where the message implied by the intervention lacks acknowledgement of the agent's intrinsic motivation, and where the reward is closely contingent on the agent's performance.

External interventions may also crowd in intrinsic motivation, where they foster self-esteem and promote agents' self-determination, thus being perceived as supportive rather than controlling. A motivational *spill-over effect* into other domains may also occur following an external intervention. This is to be borne in mind when considering that the Price Effect may overcome the negative effect of crowding out if prices or rewards are sufficiently increased. However, while it might be overcome in the particular sphere in which it is applied, it might nonetheless be the case that intrinsic motivation is undermined in other, related spheres. For example, external intervention in certain areas of environmental policy might have the desired effect in those areas while diminishing the intrinsic motivation in other areas of environmental activity.

With this general explanation of motivation crowding theory in place, we can see that if intrinsic motivations are important in biodiversity and environmental preservation, then the attempt on the part of economists to motivate its protection by placing a monetary value on it may be counterproductive due to this crowding out effect. With regard to the case of environmental policy, Frey examines (pp. 56–66) where the use of incentives such as marketable permits and charges for polluting activities may undermine and crowd out what he calls ‘environmental morale’, i.e. intrinsic motivation to act in an environmentally friendly manner. But even if it is the case that intrinsic motivation is indeed important with respect to pro-environmental behaviour, it is not immediately clear that this claim would be relevant to the kinds of economic valuation exercises that are the subject of this report. For economic valuation exercises, as stated above, are executed for the purposes of providing information for policy makers regarding the relative costs and benefits of the options under consideration with respect to non-market environmental goods. Therefore, although they typically involve contingent valuation questionnaires in which respondents are asked how much they are willing to pay for an environmental good or how much they are willing to accept in compensation for its loss, and although a key principle underlying CBA, the main decision tool into which economic valuations feed, is that the benefits of an option outweigh the costs if it is the case that the winners could adequately compensate the losers and still be better off (the Kaldor-Hicks compensation test), it remains the case that despite all this reference to payments and compensation, at no point are actual payments made; it is all purely hypothetical. Therefore, strictly, at no point in the valuation process is there a situation in which crowding out can occur.

However, Neuteleers and Engelen (2013) argue that what they call ‘commodification in discourse’ or ‘talking money’, of which economic valuation is an instance, can have a similar effect to actual payments and the expansion of real markets, and lead to the crowding out of intrinsic motivation for pro-environmental behaviour. To illustrate their claim in the context of the intrinsic motivations of public-spiritedness and civic duty, the authors cite Frey and Oberholzer-Gee’s (1997) study which saw the level of acceptance amongst members of two Swiss communities to the nearby siting of a nuclear waste repository drop from 50.8% when no monetary compensation was mentioned to 24.6% when

compensation was mentioned. To turn to economic valuation methods, they argue that framing the question with which respondents to contingent valuation surveys are presented in terms of their private willingness to pay for an environmental good as a consumer focuses their consideration and their answer on a purely economic mode of valuing the good, to the partial or total exclusion of the numerous non-economic modes of valuing that characterise people's ordinary and common relationships with nonhuman nature. Intrinsic motivation typically results, they claim, from non-economic modes of valuing, and extrinsic motivation typically results from economic modes of valuing. To the extent, therefore, that economic valuation exercises crowd out non-economic modes of valuing, the intrinsic motivation which is responsible for a significant proportion of pro-environmental behaviour may be undermined by the use of these methods.

We can consider this claim with regard to three groups: the respondents to contingent valuation surveys; decision makers; and the wider public. With regard to the first group, it may perhaps be an exaggeration to say that its members may be subject to the crowding out of their intrinsic motivation beyond the context of their participation in the survey, since the survey will only be of short duration and any given respondent is unlikely to be a participant of multiple surveys. With regard to the second group, certain decision makers may regularly be presented with cost benefit analyses which frame environmental goods in monetary terms, and so may more frequently engage in the economic mode of valuing such goods, thus crowding out intrinsic motivation. With regard to the third group, to the extent that the wider discourse regarding environmental protection has in recent years been dominated by reports on the value of 'natural capital' and the economic costs of 'business as usual', as Neuteleers and Engelen (2013: 14) suggest, '[i]t is not at all implausible that such an economic discourse comes to permeate the public debate and settle inside the minds of the broader public.' This is an instance of what they call (ibid.: 5) an 'interpersonal' motivation spill-over effect, where '[c]ommodifying one person's activities [e.g. decision makers] can crowd out the intrinsic motivation of other people [e.g. the broader public]'.

Ultimately, the above argument requires empirical study of the relative importance of intrinsic and extrinsic motivation in the environmental sphere and the hypothesis that economic valuation in particular

and commodification in discourse in general have the same or similar crowding out effects as actual payments. However, our discussion is intended to demonstrate that there are reasons for making the spread of market norms into valuation and decision making an important focus of research when assessing the failures of environmental policies that are increasingly framed in economic terms.

7.2. The Failure to Register the Content of Existing Pro-environmental Motivations

If the first failure of economic methods with respect to their motivational capacity concerns the *output* of those methods, the second concerns the *process*. This is the failure of the process to register the content or nature of the motivations that typically drive people to act in pro-environmental ways. This second understanding of the motivational capacity of a valuation method, then, has it that the motivational capacity of a valuation method is a function of the extent to which the method can comprehend and distinguish existing motivations to act in ways that protect or preserve that which is the subject of the valuation. A valuation method registers the content or nature of existing motivations insofar as the process by which the valuation is arrived at has mechanisms or procedures that can distinguish between the motivations of either participants in the valuation process whose views are contributing to the final valuation, namely, either the respondents in contingent valuation surveys or the participants in PDTs. As with the previous understanding, the motivational capacity of a valuation method cannot be assessed *per se* but only in relation to a particular individual or group, since it may be the case that a valuation method registers the content of the motivations of one individual or group while failing to register the motivations of others. We argued in Section 4, for example, that standard economic valuation methods are limited in their capacity to register ethical motivations to act for environmental goods. For individuals or groups for whom this class of motivations is important, this valuation method will, according to this understanding, lack motivational capacity.

The two understandings of the motivational capacity of a valuation method that we have outlined are connected in the following way: insofar as a valuation method registers the content of the existing motivations of an individual or group to act for biodiversity (sense 2), it is more likely that the information that the valuation issues in will serve to renew or strengthen those motivations (sense 1).

8 Conclusion: Towards Improving Economic Environmental Valuation Methods

This report constitutes a preliminary analysis of where improvements to environmental valuation methods must be made, and it thus lays the groundwork for the fulfilment of Work Package 1's task: the improvement of those methods. By way of conclusion and the establishment of the terms in which this improvement should be judged, therefore, it is appropriate to ask the question: improvement in what respect? We distinguish four kinds of improvements which the analysis contained in this report has revealed the need for.

Firstly, valuation methods require improvement with respect to the false assumptions on which monetary valuation methods are founded. Our analysis has revealed a number of assumptions that are questionable at best and straightforwardly false at worst: that money is a neutral measuring rod for people's preferences; that preferences are exogenously given and not endogenously formed by the techniques used to determine their strength; that the reasons that can be articulated for why we hold the preferences we do are not relevant for assessing how far these preferences should be taken into account; that values associated with biodiversity are commensurable; that rational decision making requires value commensurability; that nonhuman nature is principally to be valued in a *de dicto* rather than a *de re* sense; that the history that nonhuman nature embodies, both natural and cultural, can be disregarded in determining its value; that the value of nonhuman nature is such that its constituents can be substituted for one another without loss of value; and that the Price Effect holds in all spheres. Each of these assumptions leads to weaknesses in the economic approach to valuation, and it should be a matter of concern that the principal way in which environmental valuation takes place is built upon such doubtful foundations.

Secondly, valuation methods require improvement with respect to the justice they do to the variety of our relations to nonhuman nature. As we have argued, the single measure of willingness to pay or accept compensation which is utilised by economic valuations fails to register the richness of the normative relations that we stand in to particular environments. But the problem runs deeper than the

failure to register these relations, for this measure is also at fundamental odds with these relations insofar as to put a price – even a hypothetical price – on an environmental good that has been bequeathed to us is arguably to betray both the predecessors who bequeathed it and the successors who have a claim on us to care for it so that they may inherit it from us. At least some of the forms of the alternative valuation techniques we have examined avoid the use of money as a valuation tool, and PDTs in particular can register the variety of valued relations that participants’ stand in to particular environments, including reference to history and *de re* valuing attitudes.

Thirdly, valuation methods require improvement with respect to the balance they strike between the pragmatic acceptance of the expansion of market norms into the spheres of valuation and decision making, and resistance to that expansion. Monetary environmental valuation assumes the claim made in the introduction – that the source of environmental problems lies in the absence of markets for environmental goods and harms – and offers shadow prices to correct for this absence. In doing so, even if it avoids introducing actual markets and real prices, it introduces market norms into spheres which should arguably be preserved from such norms. As O’Neill (2007: 43) characterises the pragmatist argument for the worth of economic environmental valuation, the shadow prices that they result in are ‘rhetorical devices in arguments with governments to persuade them to intervene in markets to protect environmental goods’.⁷ The expression of a valuation in monetary terms is necessary, it is argued, because of the institutional framework in which political decisions are made and the way in which market norms have been adopted within the political sphere. One response to this pragmatist argument is that rather than to accept the expansion of market norms into two spheres which were previously considered to lie outside those boundaries, namely, environmental goods and political decision making, by developing techniques that ensure the best prices are secured for environmental goods such that decision makers give them due weight, it is rather more appropriate to resist this expansion (O’Neill 1997: 550). While we recognise the force, even if we do not fully accept the conclusions, of the pragmatic argument for determining shadow prices for environmental goods, we believe that leavening valuation methods where appropriate with techniques that avoid monetary

⁷ See Schokkaert (1995) for a pragmatic defence of economic valuation methods despite its problems.

measures and accommodate the articulation of non-economic modes of valuing can go some way to addressing concerns regarding the expansion of market norms.

Fourthly, we contend that valuation methods require improvement with respect to their motivational capacity. The focus of this criticism falls on the exclusive use of monetary values to measure preferences and to rank policy options, and the way in which even though there no actual markets or payments involved the use of this measure may nonetheless crowd out non-economic modes of valuing nonhuman nature and the intrinsic motivations which result from them. Further, with no mechanism to register the content or nature of existing pro-environmental motivations, the dominance of economic modes of valuing is compounded.

Appendix I Q Methodology

A particular methodology related to PDTs to which we wish to draw attention is Q methodology. Q methodology was developed in the 1930s by the psychologist William Stephenson (1953) and most applications are within this field. However, it can be used in a wide array of decision contexts in which there is conflict, and is thus also applied in political science, in particular in the USA (Brown 1980); health services research (Brown 1996); and more recently in public discourses regarding natural resources, environmental policy and governance. It is a statistical approach that provides a systematic and rigorous quantitative mean that helps researchers and stakeholders develop an understanding of the different discourses that exist in the context of a contentious topic. It enables the researcher to find the main ordered patterns in subjective viewpoints within and across individuals rather than patterns across respondents' characteristics (such as by age bracket, profession etc). Thus it is not the respondents who are the focus of the approach but the 'construction of their opinions'. Generally there are fewer discourses than there are participants in the survey and the method is particularly suited to investigate any patterns that are shared across individuals. It is particularly suited to studying those public policy topics around which there is much debate and contestation, such as biodiversity and the natural environment.

Our interest in Q methodology in this context is the way in which it can be used in support of PDTs. Insofar as such techniques require careful choice of participants in the associated mini-publics, this method can be used to identify different viewpoints in order to make sure each of these is represented in the public participation process. In these applications the method serves to address recruitment, composition and mandate issues in discursive representation (Davies et al. 2005; Dryzek and Niemeyer 2008).

Barry and Proops (1999, 2000) and Addams and Proops (2000) were the first to use the method to investigate social discourses on environmental policy and governance. Since then a diverse range of issues within environmental policy have been studied with the help of Q methodology, including forest

management, global climate change, value plurality among conservation professionals, shifting environmental perspectives among farmers and travellers preference for middle-distance transport (e.g., Steelman and Maguire 1999; Dasgupta and Vira 2005; Sandbrook et al. 2011; Davies and Hodge 2012; Van Excel et al. 2011). In more recent years, the method has become well established in the field of natural resources management and policy as is evidenced for example by the workshops on Q methodology and forest policy held at the US Geological Service in 2007 (www.fort.usgs.gov/QMethodology). An interesting application of Q methodology from this workshop investigates prevailing multi-spatial value discourses among local residents related to a national forest in Colorado.

Since we are using Q methodology in Work Package 1 to elicit the views of economic practitioners and applied economists regarding both the weaknesses of monetary approaches to biodiversity valuation and the possibilities and prospects of the alternative methods of valuation assessed in this document, we will outline the three stages involved in a Q methodology study. Stage one involves developing a set of statements to be sorted; stage two requires participants to sort the statements along a continuum of preference; and in stage three the data are analyzed and interpreted. The research is performed on relatively small samples. This person sample is non-random; it is a selection based on theoretical relevance — i.e., people who have well-formed and distinct viewpoints. These respondents are then asked to rank-order a number of the statements on a Likert point scale ranging from most disagree to most agree with. The number of statements should be no more than 60 (usually about 30) statements and is called the Q set.

The Q set is the actual research instrument. The initial statements, of which there could be hundreds, is called the *concourse* and can be collected in various ways, for example through a survey and/or the scientific and/or popular literature. The total collection of items is in fact not restricted to words and could include photographs etc. Once the whole *concourse* has been gathered, a subset of statements is selected to form the Q set: the group of statements to be rank-ordered by the test subjects. This selection of the Q set is usually done employing a two dimensional *concourse* matrix. This matrix then

helps with cell sampling of the Q set. What goes on the x - and y -axis of this matrix depends on the research topic.

The respondents perform the sorting while having the whole Q set in front of them. By sorting the respondents give their subjective opinion about the statements, and by doing so reveal their personal profile. Q sorters typically receive randomly numbered opinion statements, sorting instructions and an answer sheet to record the chosen order of statements. Commonly an answer sheet is used (Q grid) which forces the Q sort into the shape of a quasi-normal distribution (Figure 1). There are fewer statements that can be placed at the extreme ends and more that are allowed to go into the middle area. The middle represents the grey zone, or almost neutral, reaction.

For the data analysis, each person's rank-ordered sort of statements is transformed into an array of numerical data. Each person's array of numerical data is then inter-correlated with the arrays of all the others. Respondents correlate to others with similar opinions based on their Q sorts. The correlation matrix is then subjected to factor analysis to obtain groupings of data arrays that are highly correlated. Thus the results are the grouping of opinion profiles based on the similarities and differences in which the statements are arranged by the participants. This determines the factors that represent clusters of participants with similar opinions. Factor loadings show each participant's association with each of the identified opinion types. There are several free software programs to convert the statements to an online or offline interface and other free software packages or the factor analysis of Q sort data.⁸ Recent publications primarily use PQ method for the statistical analysis (Sandbrook et al. 2010; Hermans et al., 2012).

The consensus and conflict statements can provide highly relevant insights both from a theoretical and a practical perspective. From a practical perspective they can serve to identify potential coalitions and can help to describe the polarised nature of the debate around the specific topic investigated in particular. Q methodology further can help enable, according to Barry and Proops (1999), both a more democratic

⁸ See <http://schmolck.userweb.mwn.de/qmethod/webq/> and <http://schmolck.userweb.mwn.de/qmethod>

and more effective form of policy making in virtue of the way in which it can ‘identify for policy makers the ways environmental issues are perceived by various groups, allowing the identification of common issues or perspectives in the population’ (p. 345) and allow those who ‘ultimately have to live with the consequences of environmental policy to be included in determining or identifying a problem (which can then be addressed by policy makers), it is more likely that the policy will be acceptable (or legitimate) and thus more likely to be effective’ (pp. 344–345). Further, Q methodology can be used as a support tool for PDTs insofar as it can be used to select participants in such techniques on the basis that they represent a particular widely shared discourse (Dryzek and Neimeyer 2008: 486), or a discourse that is particularly relevant to environmental issues, such as ecocentric discourses (Davies et al. 2005: 611).

Van Excel et al. (2005) report that that Q-sorters often spontaneously indicate they have enjoyed participating in the study and that they experienced it as instructive. They argue that this is because after completing their Q sort, respondents can oversee their opinion or preference regarding the subject of the study reflected on the score sheet lying in front of them, and can make changes if they disagree. These aspects of recognition and flexibility generate a sense of control of their contribution and of reliability of the study as a whole. Thus Q sorting perhaps requires greater involvement than standard survey analysis, but apparently does so in a very pleasant and comprehensible manner.

Q methodology studies must be carefully designed to avoid researcher bias, and it must be borne in mind that since it can only analyse discourses it may overlook ‘positions that are poorly articulated, or indeed (for example, in the case of animals) not directly articulated at all’ (Davies et al. 2005: 612). Further caution is required if Q methodology is used to select participants in a deliberative process that participation in the process is not illegitimately gained by strategic sorting of the Q set, and that participants are not already biased by their exposure to the statements in the Q set (ibid.). A more general criticism of Q methodology is advanced by Dryzek and Neimeyer (2008: 487) when they claim the method could be seen as substituting ‘substitute social science for political process, with the risk of empowering an unaccountable social scientific elite.’

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