Environmental Cost-Benefit Analysis

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Abstract

Environmental cost-benefit analysis, or CBA, refers to the economic appraisal of policies and projects that have the deliberate aim of improving the provision of environmental services or actions that might affect (sometimes adversely) the environment as an indirect consequence. Vital advances have arisen in response to the challenges that environmental problems and environmental policy pose for CBA. In this article, we review a number of these developments. Perhaps most notably this includes continuing progress in techniques to value environmental changes. Growing experience of these methods has resulted in, on the one hand, ever greater sophistication in application and, on the other hand, scrutiny regarding their validity and reliability. Distributional concerns have led to a renewal of interest in how appraisals might throw light on questions about equity as well as efficiency, and there have been substantial new insights for discounting costs and benefits in the far-off future. Uncertainty about what is lost when environmental assets are degraded or depleted has resulted in a number of distinct proposals although precaution is the watchword in each. Just as importantly, there is a need to understand when CBA is used in practice and why environmental decisions are often made in a manner apparently inconsistent with cost-benefit thinking.

Contents

1. INTRODUCTION

RP: revealed preference

SP: stated preference

Although the principles of cost-benefit analysis (CBA) remain largely the same, the practice of carrying out appraisals has undergone a transformation over the past two or so decades.[1] Nowhere is this more the case than for environmental applications: that is, cost-benefit appraisals of policies and projects that have the deliberate aim of improving the provision of environmental services or actions that affect (sometimes adversely) the environment as an indirect consequence. Current best practice in *environmental cost-benefit analysis* reflects, in its turn, an array of conceptual and empirical developments primarily in environmental economics (but also in other fields of economic inquiry such as health economics). It is the objective of this article to review the most important of these recent developments.

In doing so, we identify what justly can be considered to be the core activity in environmental CBA: the estimation of monetary values for environmental changes, environmental valuation. This has given rise to a proliferation of methods that have sought to uncover, in a variety of ways, the value of environmental impacts (and nonmarket impacts more generally). As we discuss, critical scrutiny of these methods has also been prominent. Although developments in techniques of nonmarket valuation have been a defining feature of environmental CBA, a broader array of evolving issues has also come to the fore in extending economic appraisal to contemporary environmental policy challenges, perhaps most notably climate change and biodiversity loss. Some of these issues can be summarized as stemming from distributional concerns with regards to how human well-being and wealth are distributed across generations as well as within generations: respectively inter- and intragenerational equity. Other insights have emerged in response to reflections on whether the extent and nature of the uncertainty and irreversibility that characterize certain environmental problems might require that decision making needs to be weighted more heavily in favor of precaution. Finally, there are important issues of how CBA operates in the real world and in relation to its critiques. Just as important are questions of political economy: What reasons explain why actual policy design diverges from the optimal design of those policies according to CBA?

2. ENVIRONMENTAL VALUATION

Two main approaches have been developed to place an economic value on nonmarket goods and services: revealed preference (RP) and stated preference (SP) methods. There are many possible uses for such information.

[1]There is an extensive academic literature on CBA, some of which may not use the term cost-benefit analysis but instead refers to benefit-cost analysis, policy evaluation, or project appraisal. Numerous texts and manuals have appeared, many covering the general field of CBA [e.g., Sugden & Williams (1); Just et al. (2); Boardman et al. (3)]. Other contributions have covered detailed procedures for estimating shadow prices (i.e., the true social value of costs of benefits as opposed to the distorted market prices we might observe in reality), especially in developing countries [e.g., Ray (4); Londero (5)]. These texts remain highly relevant in environmental contexts where, for example, policy distortions remain substantial, such as perhaps in the evaluation of biofuels. More recently, texts have appeared specifically in the environmental context [e.g., Johansson (6); Hanley & Spash (7); Pearce et al. (8)].

Within the context of CBA, the objective is to generate original (or primary) data on the total economic value that the public places on environmental changes that arise as a result of some policy proposal.[2] A complementary approach, that of benefits (or value) transfer, draws on the accumulated findings within such valuation studies conducted to date and so applies existing monetary values (or secondary data perhaps after some adjustment) to new policy contexts. The techniques and recent developments are discussed in the sections that follow.

2.1. Stated Preference Methods

Stated preference (SP) methods is an umbrella term under which are found a range of survey-based methods that use constructed or hypothetical markets to elicit preferences for specified policy changes [Bateman et al. (9)]. By far the most widely applied SP technique is the contingent valuation (CV) method. Over the past 40 years, this technique has been increasingly used by academics and policy makers owing to its flexibility and ability to estimate the economic value of environmental goods that are not traded in markets. Using an appropriately designed questionnaire, a hypothetical (or contingent) market is described where the good in question (which might be an improvement in water quality, reduction in a risk to human health, or protection of an ecosystem) can be traded. This contingent market defines the good itself, the institutional context in which it would be provided, and the way it would be financed. Respondents are then asked to express their maximum willingness to pay (WTP) or minimum willingness to accept (WTA) compensation for a hypothetical change in the level of provision of the good. The method can be used to estimate all the benefits (use and nonuse) associated with a change in the level of provision of a good or service. CV is firmly rooted in the economic theory of individual choice. For thorough reviews of the CV method see Mitchell & Carson (10), Bateman et al. (9), Champ et al. (11), and Alberini & Kahn (12). This asserts that elicited WTP or WTA values are monetary measures of respondents' preferences, corresponding to Hicksian welfare measures. That is, for a proposal that results in an environmental improvement, the change in well-being that an individual enjoys can be measured by his or her WTP for, or his or her WTA compensation to forego, that improvement. For an environmental deterioration, the change in well-being is the WTP to avoid that outcome or WTA compensation to tolerate it. Choosing between WTP and WTA, in a given context, arguably remains a contentious issue, although in large part the choice is determined by whether people affected by a policy have a property right to the current or the new level of provision of the good in question. In addition, CV (and SP approaches more generally) combines these economic precepts with insights from psychology and survey research.

The CV method has been extensively applied in both developed and developing countries and the range of environmental issues addressed is wide: water quality, outdoor recreation, species preservation, forest protection, air quality, visibility, waste management, sanitation improvements, biodiversity, health impacts, natural resource damage, and environmental risk reductions to list but a few. Carson's (13) bibliography contains over 5500 published and unpublished CV studies undertaken in just under 100 countries. Much of the impetus for this body of work was the conclusions of the special panel appointed by the U.S. National Oceanic and Atmospheric Administration (NOAA) in 1993 [Arrow et al. (14)] following the *Exxon Valdez* oil spill in Alaska in

CV: contingent valuation

WTA: willingness to accept

WTP: willingness to pay

[2] It is usual, in environmental economics, to divide this notion of total economic value into use and nonuse (or passive use) values. Use values relate to actual use of the good in question (e.g., a visit to a national park), planned use (a visit planned in the future), or possible use. Actual and planned uses are fairly obvious concepts, but possible use could also be important because people may be willing to pay to maintain a good in existence in order to preserve the option of using it in the future. Nonuse value refers to willingness to pay to maintain some good in existence even though there is no actual, planned, or possible use (e.g., willingness to pay to preserve a species as a bequest for future generations or simply for its own sake).

CE: choice experiment

1989. The panel concluded that, subject to a number of recommendations, CV studies could produce estimates reliable enough to be used in a (U.S.) judicial process of natural resource damage assessment. But very few CV studies have actually been associated with litigation, as resource equivalency methods (where damaged resources are replaced like-for-like perhaps on the basis of physical criteria) became the favored approach for damage assessments after 1996 [Smith (15)]. Rather, in the past 15 years, CV has mostly been applied to the evaluation of projects and policies and has been used in over 50 countries by a range of government agencies and international organizations [Carson (16)].

The rise in popularity of CV has been accompanied by a very active debate and critical assessment of the merits and limitations of the technique and its underlying conceptual framework, mostly within the economics profession [e.g., Hausman (17), Diamond & Hausman (18), Portney (19), Hanemann (20), Boyle & Bergstrom (21), Carson et al. (22)] but also beyond [e.g., Fischoff & Furby (23), Schkade & Payne (24), Sagoff (25)]. In fact, according to V. Kerry Smith, "Contingent valuation has prompted the most serious investigation of individual preferences that has ever been undertaken in economics" (15, p. 46). Some critiques are philosophical in nature and, as such, discuss concepts of value and the appropriateness of relying on private economic value systems to inform public policy [see Haddad & Howarth (26)]. But the largest body of literature that has emerged from the CV debate has focused on technical aspects, constructing rigorous tests of robustness across a variety of policy contexts and investigating and correcting for the presence of bias or theoretical anomalies [Bateman et al. (9), Mitchell & Carson (10); Bateman & Willis (27)].

Nevertheless, the controversy is unlikely to disappear as many critics appear to hold CV up to a criterion of perfection [Boyle (28)]. Clearly, this is unrealistic not least because perfection does not characterize any empirical methodology or, indeed, actual market decision. Moreover, taken as a whole—and notwithstanding notable caveats—empirical findings so far largely support the validity and reliability of CV estimates—from well-executed studies—of the value of nonmarket goods. The elaborate methodology that has earned its place in the environmental economist's tool-kit today is the culmination of this substantial body of research. That said, a healthy dose of skepticism is important in the application, use, and interpretation of any empirical methodology such as CV [Boyle & Bergstrom (21)]. Hence, the intense scrutiny to which CV has been subjected does point to the need for practitioners to follow, in some way, guidelines for best practice. Although the NOAA guidelines continue to be a focal point, there are a number of more recent guidelines or state-of-art reference points useful to practitioners and to those who are focused on the use of CV in policy evaluation [e.g., Bateman et al. (9), Champ et al. (11), Alberini & Kahn (12)].

2.1.1. Recent developments

Choice experiments. Choice modeling refers to a family of survey-based methodologies, which has been used extensively in the marketing and transport literatures to model preferences for bundles of characteristics of goods and to isolate the value of individual product characteristics typically supplied in combination with one another. The conceptual microeconomic framework for these analyses lies in Lancaster's (29) work, which assumes that the well-being consumers derive from goods can be decomposed into its composing characteristics. Recent extensive reviews can be found in Bateman et al. (9), Louviere et al. (30), Bennett & Blamey (31), and Hensher et al. (32). Curiously, there does not seem to be a uniform use of terminology on these multiattribute valuation techniques in the literature. Contingent ranking, contingent choice, conjoint analysis, and choice experiments (CEs) are all terms that have been used to describe similar SP data where choices, ranks, or matches between alternatives are involved. Perhaps the most widely used variant in environmental economics is the CE technique [Bateman et al. (9)]. In a CE survey, respondents are required to choose their most preferred out of a set of alternative policy options. For example, in the case of a policy to improve the quality of some water body, such as a river, the attributes might be a boost to the river's ecology (perhaps as indicated by fewer fish deaths), decreased health risks to those who are exposed to the water such as swimmers and rowers, and increased visual amenity. Each alternative choice in a CE is characterized by a number of such attributes, which are offered at different levels across options. A price or cost variable is typically one of the attributes, and WTP can be indirectly inferred from the choices made. CE can be seen as a generalization of the dichotomous choice CV approach[3] [Carson (16)], sharing a common theoretical framework as well as a common basis in terms of econometric analysis.

CE boasts a number of attractions [Hanley et al. (33)], but primarily it is the ability to deal with situations where changes are multidimensional, and where trade-offs between these dimensions are of particular interest, that makes the strongest case for its inclusion as a technique of environmental valuation. This comes at a cost, however, most notably the cognitive difficulty associated with complex choices between bundles with many attributes and levels. Although task complexity may lead to learning, it may also result in, for example, fatigue effects and the use of heuristics (rules of thumb) to make choices. This, in turn, may increase the variance of values estimated as well as the occurrence of random errors in choices and inconsistent or irrational responses [Foster & Mourato (34)]. Moreover, CEs are typically much more complex to design and to analyze statistically than CV experiments [Bateman et al. (9), Hanley et al. (33)]. And of course, as an SP method, CE (and its variants) faces many of the criticisms and problems raised with CV. But unlike CV, the CE format has received relatively little testing with respect to its vulnerability to empirical anomalies in environmental applications. As a result, a wide range of potential benefits suggested for CE methods (vis-á-vis CV) remains little more than informed speculation [Hanley et al. (33)].

Entreaties and hypothetical bias. After decades of theoretical debate, methodological advances and empirical applications of the CV method, the central concerns remain arguably the same: the hypothetical nature of the contingent market and the absence of real economic commitments (ask a hypothetical question, get a hypothetical answer). So-called hypothetical bias—arising, for example, when people's stated WTP differs from true WTP owing to the hypothetical nature of the good—has been the subject of much scrutiny. The conventional wisdom is that individuals are likely to overstate their economic valuation of a good in a hypothetical context [e.g., Neill et al. (181); although see also Dickie et al. (35)]. The evidence further suggests that hypothetical bias (*a*) increases with the size of elicited values (with the implication that for small hypothetical values common in CV studies, hypothetical bias may not be a major problem) [Murphy et al. (36)]; (*b*) is lower for WTP than for WTA formats [List & Gallet (37)]; (*c*) is higher in the presence of uncertainty [Champ et al. (38)], unfamiliar policy changes, and less knowledgeable respondents [e.g., List & Gallet (37)]; and (*d*) is higher for voluntary contributions (than compulsory payment mechanisms such as taxes) perhaps because of the free-riding associated with actual contributions [Murphy et al. (39)].

In recent years, a growing number of studies have begun to explore the use of entreaties such as "cheap talk" to address the problem of hypothetical bias [e.g., Murphy et al. (36), Cummings & Taylor (40), List (41), Poe et al. (42), Lusk (43)]. The term cheap talk (CT) arises from its use in the information, bargaining, and game theory literatures where it

[3]This referendum-style method asks respondents whether they would be willing to pay or not some specified amount.

typically refers to the "cost-less transmission of information and signals" [Cummings & Taylor (40)]. In the valuation literature, it usually refers to a script, typically quite lengthy, which is added to the valuation scenario in order to directly draw respondents' attention to the problem of misstating true values as a result of the hypothetical setting. Instead, respondents are asked to focus their responses as if they were in a real-life setting, hence the entreaty nature of this added text. In the first published study to test CT, Cummings & Taylor (40), in a laboratory setting, found that a long eight-paragraph script was effective at reducing hypothetical bias for three different public goods, with responses under the CT treatment being indistinguishable from responses involving actual payments. Subsequent studies of CT show a number of patterns emerging [see, for example, Murphy et al. (39), List (41), Poe et al. (42), Lusk (43)]: (*a*) despite mixed evidence, on the whole CT appears to be effective at lowering stated values; (*b*) shorter scripts tend to be less effective; (*c*) inexperienced consumers are more receptive to CT scripts than experienced or knowledgeable respondents; (*d*) CT appears to be more effective at higher payment levels where hypothetical bias may be more pronounced; and lastly (*e*) at least one study, which looked at the potential for CT to result in overcorrections, suggested no evidence of this effect. Despite its relative success at reducing hypothetical bias, the potential of entreaties for dealing with other types of CV bias remains unexplored. One exception is a recent study by Atkinson et al. (44), which used an entreaty to address the problem of protest responses in two CV experiments with encouraging results. More work is needed in this area to assess the usefulness of entreaties in correcting for other types of bias.

Deliberative group valuation. Scrutiny, emanating largely from outside the environmental economics literature, has evolved into proposals for more communal and deliberative approaches to economic valuation. The rationale is that this may be able to capture and quantify collective values that are additional to individual or household values, typically elicited in CV or CE, and/or may help to minimize problems such as some protest behaviors [Haddad & Howarth (26)]. Sagoff (45) was one of the earlier proponents of the idea of deliberative, discursive, jury-like approaches as a way to clarify public values by group consensus, whereas Prato (46) has proposed community-based collaborative decision-making processes. One practical effect of such proposals has been deliberative valuation workshops or "market stalls" [e.g., Macmillan et al. (47), Alvarez-Farizo et al. (48)] where participants are given the opportunity to discuss the individual or collective value of the proposed change in a group context. Current evidence on deliberative and group approaches to valuation is limited, with only a small number of studies and relatively small sample sizes. For example, Macmillan et al. (47) had 52 people in their workshops, and Alvarez-Farizo et al. (48) had 24 people. Proponents hope, however, that these approaches might be used in applications involving poorly informed consumers as well as unfamiliar or complex changes such as biodiversity loss [Macmillan et al. (47)]. Investigating such claims through wide-scale testing therefore is a rich topic for further research.

Valuing health risks. Environmental policy affects human health in a number of ways. First, by reducing environmental risks to lives, it may save lives, i.e., reduce premature mortality. Second, it may improve the health of those living with a disease, such as a respiratory illness, and so result in a morbidity benefit. Valuing these mortality and morbidity effects in monetary terms can provide extremely useful information for policy. Past evidence indicates that the benefits of reducing human health effects (mortality and/or morbidity) often exceed the costs of pollution control by considerable margins. Over the last 30 years, SP studies (together with RP methods) have been used extensively to calculate both individual WTP to secure reductions in the risk of death arising from a policy and WTP to avoid particular health outcomes.

At this time, substantial progress has been made particularly in the valuation of mortality risks.

For convenience, WTP for mortality risk reductions is normally expressed in terms of the value of statistical life (VOSL). This implies dividing the WTP for a given risk reduction by that risk reduction to obtain the VOSL [Pearce et al. (8), Bolt et al. (49), Krupnick (50)]. Various countries adopt single (or ranges of) values for the VOSL and then use them in policy appraisal. The U.S. Environmental Protection Agency (EPA), for example, has used for many years a VOSL range of \$0.6 million to \$13.5 million, with an average of \$4.8 million (1990 US\$) on the basis of an assessment of the existing U.S. literature [Robinson (51)]. However, although there is a very large body of research on health values for North America and Europe, there is a dearth of evidence for developing countries in general.

Notwithstanding this apparent lack of geographical breadth in the empirical record, considerable strides have been made in recent years in terms of clarifying both the meaning and size of the VOSL [see Pearce et al. (8) for an overview]. Recent research, for example, has shown that the age of the respondent who is valuing the risk matters [see Krupnick (52) for a review]. Although age may or may not be relevant in valuing risks that are immediate (e.g., life is threatened in the here and now), in fact, the literature is ambiguous on this; age has been found to affect the valuation of future (or latent) mortality risks. This is an important finding given that environmental contexts are associated with both immediate and future health risks. That is, the value of a reduction in the risk of death can be very different according to whether a policy affects mostly older individuals already suffering from some underlying health condition (and who thus face a very present risk of dying as a result of exposure to say poor air quality) or younger individuals who might otherwise have enjoyed a normal life expectancy (in the absence of cumulative exposure to poor air quality). Thus, the standard practice of applying the VOSL to value all reductions in mortality risk without regard to the age of those who benefit may result in erroneous value estimates. In general terms, a policy that lowers the general level of exposure to air pollution should be evaluated in terms of the (lower than immediate VOSL) valuations associated with younger people's valuations of future risks plus older persons' valuation of that risk as an immediate prospect.

VOSL: value of statistical life

An alternative approach that takes into account the age of persons saved by a particular policy and that may capture the shorter life expectancy phenomenon is the value of a statistical life-year (VSLY) approach. The VSLY is calculated by dividing the value of a statistical life by the discounted remaining life expectancy, thereby converting VOSL estimates into a value per life-year saved. VSLY can then be multiplied by life-years saved, i.e., the remaining life expectancy, to value the statistical lives of persons of different ages. However, the VSLY approach still assumes that the value per life-year saved is independent of age and, in addition, is sensitive to the rate used to discount the value of future life-years [Krupnick (52)]. There also remains debate about whether valuing life expectancy changes is a superior approach to valuing reductions in the risk of death.

A rather distinct age-related issue is that some environmental risks fall disproportionately on the very young. The calculus of WTP seems to break down here because children may have no income to allocate between goods, including risk reduction. They may be ill informed about or be unaware of risks and may be too young to articulate preferences. One implication is that adults' (or parents') valuations of the risks on behalf of children need to be estimated. The literature on which to base such judgments is only now coming into existence (53). Preliminary findings suggest that the resulting values of WTP may be higher for adults valuing on behalf of children than they are for adults speaking on behalf of themselves.

Valuing ecosystem services. Ecosystem services refer to the wide range of benefits that people derive from the multitude of resources and processes that are supplied by natural ecosystems

(54, 55). The Millennium Ecosystem Assessment (55), the most comprehensive survey to date of the state of the planet, indicates that nearly two thirds of the services provided by nature are in rapid decline. An evolving issue, therefore, is the valuation of ecosystem services [e.g., Daily (54)]. In the case of forests, for example, advances have been made in measuring the economic values associated with timber and nontimber products, carbon sequestration and storage, recreation, and watershed regulation. Progress has been more limited in estimating the nonuse values of forests, and a lively debate surrounds the value of genetic material in forests for pharmaceutical research [Pearce & Pearce (56)]. Yet, biodiversity and ecosystem services are perhaps among the most complex environmental concepts to define, measure, and value, and there remain major methodological issues to be addressed [Daily et al. (57)].

Efforts to estimate the total economic value of ecosystem change need to be distinguished from efforts to value all ecosystems. It is not sensible to speak of the total value of a type of ecosystem and even less sensible to speak of the total value of all ecosystems. Unfortunately, some of the recent literature on ecosystem valuation claims to do just this [e.g., Costanza et al. (58) and, for a critique, Bockstael et al. (59)]. Nor is it clear that bottom-up approaches—whereby each type of service is valued separately and then the values are added to get some idea of the total economic value of the ecosystem—are capturing the whole value of the ecosystem. Put another way, the value of the system as a whole may be more than the value of the sum of its parts perhaps because of complex ecological interactions [Arrow et al. (60)]. This is not just a problem for those approaches that seek to evaluate ecosystems as a whole. For example, a small economic value for any one service might suggest it could be dispensed with, yet its removal could reverberate on the other services through complex changes within the ecosystem. Thus, the bottom-up valuation procedure could be misleading as a guide to the social value of marginal changes in the provision of ecosystems as well.

It follows that ecosystem valuation is not a straightforward exercise, and it seems fair to say that the literature has progressed only a limited distance in tackling these issues [Pearce et al. (8)]. Therefore, an important way to extend SP and other valuation methodologies is in developing their suitability to aid decision making in complex multidimensional areas, such as in managing the flows of ecosystem services [Alberini & Kahn (12)]. Nevertheless, a considerable challenge is presented in translating such values (suited as these are to valuing relatively small changes in ecosystem provision) to the types of large-scale change that are argued to characterize current threats to ecosystems and biodiversity.

2.2. Revealed Preference Methods

RP methods look at "surrogate markets," that is, analyze or infer preferences for nonmarket goods as implied by past behavior in an associated market. These methods seek to quantify the market footprint of nonmarket changes. The principal advantage of RP over SP techniques is the fact that they are based on actual behavior and hence typically enjoy higher credibility among policy makers. Inevitably, RP methods have limitations. The first is the inability to estimate nonuse values, as they are based on market footprints of some form of use-related behavior. The second is their inability to estimate values for levels of quality that have not been experienced and revealed by the market. Nor have the assumptions that underpin RP been subject to anything like the scrutiny that has accompanied developments in say CV approaches [Bishop (61)]. The two main RP techniques are the travel cost method and the hedonic price method; both are discussed in the following sections.

2.2.1. Travel cost method. The travel cost (TC) method has been used to value spatial nonmarket goods, particularly outdoor locations used for recreational purposes (e.g., parks, woodland, beaches, lakes). Typically, the recreational area is an unpriced good. However, the basis of the TC approach is the recognition that individuals produce recreational experiences through the input of a number of factors, which may, in some way, command prices. Among these factors are the recreational area itself, travel to and from the area, and in some cases, staying overnight at a location. Such information is usually collected through surveys carried out at the recreational site and through secondary data, although complications abound, such as the treatment of multiple purpose trips and the estimation of the value of time. Overviews of the TC method can be found in Habb & McConnell (62) and Parsons (63). Recent treatments using this technique have looked at multiple-site TC and, in doing so, analyze visitors' choices between a number of substitute sites, which differ both in location (i.e., some have higher access costs) and site qualities, for any particular recreational trip. By observing how different visitors choose between sites with different qualities and different costs of access, it is possible to use econometric techniques to estimate how each of the quality variables and the cost variable contribute to the utility of a visit.

An example of an application of this technique can be found in Day (64) where a multiple-site TC model was used to estimate the recreation demand of local residents for four renowned games reserves—Hluhluwe, Umfolozi, Mkuzi, and Itala—in KwaZulu-Natal, South Africa. The costs of traveling to the different game reserves vary for different households living at different distances from the sites. Recreational trips to the game reserves were characterized by three dimensions of choice: which reserve to visit (each of the four reserves provides differing opportunities for viewing game), how long to stay (short versus long stay), and which type of accommodation to use (basic hut, luxury chalet). Hence, an option was defined as the choice of a particular type of accommodation in a particular reserve for a particular length of time. The three dimensions of choice facing visitors were modeled using a three-level nested logit model. Data on a random sample of 1000 visitors were derived from the KwaZulu-Natal Parks Board's reservation database (e.g., trip duration, accommodation type, postal code), and accurate measurements of travel costs and travel times were sought using detailed mapping techniques.

HP: hedonic price

2.2.2. Hedonic price method. The starting point for the hedonic price (HP) method is the observation that the price of a large number of market goods is a function of a bundle of characteristics. For instance, the price of a car is likely to reflect its fuel efficiency, safety, and reliability. The HP method uses statistical or econometric techniques to isolate the implicit price of each of these characteristics. Recent reviews of the hedonic approach can be found in Habb & McConnell (62) and Taylor (65). Two types of markets are of particular interest for the HP method: (*a*) property markets and (*b*) labor markets. In the case of the latter, the HP method has also been used to estimate the value of avoiding risk of death or injury by looking for price differentials between wages in jobs with different exposures to physical risk [Taylor (65), Krupnick (50)]. In the former, hedonic studies of the property market therefore have been used to identify the value of nonmarket goods (or bads) affecting house prices, such as road traffic, aircraft noise, air pollution, water quality, proximity to landfill sites, and planning restrictions on open spaces in and around urban areas. The merits of such values will depend on a number of factors, including the existence of a competitive property market and econometric problems. For example, a possible difficulty belonging to the latter problem is multicollinearity. As an illustration, if local air quality and proximity to good schools are highly correlated, then it becomes all the more difficult to separate out the specific influence of each. This HP process itself involves collecting large amounts of data on prices and characteristics of properties in an area, and applying statistical techniques to estimate a hedonic price function, relating each characteristic of interest to the house price (such as those relating to the property itself, local environmental quality, proximity to local amenities, and public services). In the U.K. case, WTP values for reductions in transport-related noise—estimated from a hedonic study—are now used officially in road and rail appraisal.

2.2.3. Recent developments

Use of geographical information systems. A very promising development in the RP literature is the increasing use of geocoded (spatially referenced) data and geographical information systems (GISs). Bateman et al. (66) outline the potential benefits of this approach in the context of both recreational demand research and hedonic analysis. A GIS offers one means of obtaining considerable improvements in recreational demand modeling via greater accuracy in calculating travel times and description of available substitutes to a particular recreational resource. To the extent that data availability permits, a GIS presents a way of routinely and comprehensively tackling such geographical issues through detailed mapping and so on [see Day (64)]. Given the intrinsic spatial relationships between properties, a GIS can also be very usefully applied to improve the estimation of independent variables in the HP model (such as distance between properties and particular locations or amenities of interest). Moreover, geocoded property value datasets and GIS allow the use of spatial econometric methods in HP studies [Taylor (65)]. Day et al. (67) illustrate a recent example of the application of these techniques in the context of the valuation of transport noise externalities using the hedonic approach.

Combining revealed and stated preference data. There is a growing realization that RP and SP information is highly complementary and can be used in joint estimation of preferences [Louviere et al. (30), Habb & McConnell (62), Cameron (68), Ben-Akiva & Morikawa (69)]. After adequate scaling, RP and SP data relating to people's discrete choices can be pooled (stacked) to allow joint estimation of the parameters associated with the attributes. Swait & Louviere (70) developed a procedure to account for the relative variance of each type of data and allow joint estimation under a single preference structure. Combining RP and SP data enhances the unique strengths of these respective data while minimizing their limitations. That is, the approach takes advantage of efficiency gains from estimating the same parameters with information from different choices. Examples of this approach in the environmental field include Adamowicz et al. (71) and Kling (72).

2.3. Benefits Transfer

Although advances in methods that seek to generate primary data on the value of environmental goods and services have been a striking feature of modern CBA, routine use of valuation arguably will rely just as much on using secondary data, i.e., the results of existing studies but applied to new (yet related) policy settings—so-called benefits (or value) transfer. This involves taking a unit value of a nonmarket good estimated in an original or primary study and using this estimate (perhaps after some adjustment) to value benefits that arise when a new policy is to be implemented. Benefits transfer is itself the subject of a rapidly growing literature [see, for example, Boyle & Bergstrom (73), Desvousges et al. (74), Navrud & Ready (75)] and, if a valid procedure, has the potential to reduce the need for costly and time-consuming original studies. Indeed, the holy grail of benefits transfer is the consolidation of original data on nonmarket values in transfer databases where values can be taken "off the shelf" and applied to new policies and projects as needed [such as the Web-based Environmental Valuation Reference Inventory (EVRI), **http://www.evri.ca**].

This raises considerable challenges, not least ensuring there is an abundance of good quality studies to populate these databases across the array of environmental changes that are of interest to decision makers. Just as importantly, the validity of benefits transfer remains open to scrutiny. Indeed, a number of contributions have sought to test the accuracy of benefit transfer exercises. Ready et al. (76), for example, undertook identical CV surveys in five European countries to elicit WTP to avoid health effects thought to be associated with air pollution. A test of the validity of benefits transfer was permitted by a comparison of whether the WTP values across these countries, for some identical health outcome, were similar. Similarity is judged using specific (statistical or other) criteria. Put simply, such tests indicate whether transferring a WTP value—in this case from one country to another—is justifiable (and how much error is entailed in that transfer). Reviews by Brouwer (77) and Rosenberger & Loomis (78) summarize the findings of a number of these types of tests for recreational resources, water quality improvements, and landscape amenities. Brouwer & Bateman (79) investigate the temporal reliability of transfer values; that is, at some point, values taken from older vintages of original studies are likely to become too unreliable to transfer to contemporary policy settings. Distilling an overall message from these tests, however, is not straightforward. In some cases, transfer error ranges are small, and in other cases, these ranges are extremely large indeed. In the case of health value transfers across national boundaries, for example, Barton & Mourato (80) find large transfer errors for two comparable CV surveys eliciting WTP to avoid ill-health symptoms associated with exposure to polluted coastal water in Portugal and Costa Rica. By contrast, Alberini et al. (81) find that transferring health values from the United States to Taiwan provides a reasonable approximation of the findings of an original study conducted in Taiwan.

The evidence to date suggests that there is a need for still more research to secure a better understanding of when transfers work and when they do not as well as to develop methods that might lead to improved transfer accuracy. Intuitively, more sophisticated transfer approaches might be the answer. Such techniques seek to control for as many important differences as is possible between an original study site (or sites) and a new policy site, e.g., relating to the attributes of the environmental goods as well as the socioeconomic characteristics of the populations at each site. The most ambitious of these approaches uses meta-analysis, a statistical analysis of the results of a (typically) large group of WTP studies. Studies have taken place focusing on urban pollution, recreation, the ecological functions of wetlands, values of statistical life, noise, and congestion [see, for example, chapters in Navrud & Ready (75)]. Another development is the use of GIS in, for example, the context of recreational value transfers [Lovett et al. (82)]. In practice, however, a number of studies suggest that simpler approaches, e.g., where WTP is adjusted to take account only of per capita income differences, perform just as well in terms of transfer accuracy [Alberini et al. (81), Ready et al. (76)]. Understanding more about the relative merits of simple and sophisticated transfer exercises is of considerable practical importance. The implication, however, of focusing mostly on intricate approaches is that benefits transfer—as is the case for primary valuation methods—might largely become the preserve of the highly trained specialist rather than a tool that can be routinely used by a broader assortment of practitioners.

Invariably, benefits transfer introduces greater uncertainty into appraisals in that it entails additional assumptions and judgments to those contained in original studies. The key issue is whether the transfer is still, on balance, informative, a question that was asked in a comparatively early contribution by Brookshire (83). As a practical matter, some degree of imprecision may not matter, and rather what is needed are rules of thumb relating to the hurdle of accuracy that any transfer must overcome instead of a benchmark using overly stringent statistical criteria [Ready et al. (76), Kristofersson & Navrud (84)]. Such pragmatism needs to operate within some formal understanding of what constitutes best practice. It is surprising that there are no generally accepted practical transfer protocols to guide practitioners [but see Champ et al. (11), Brouwer (77), and Navrud & Ready (75)]. Given the potentially central role in environmental decision making of benefit transfers, and its possible problems, it seems reasonable to predict—as well as to hope—that such guidelines will emerge in the near future. That said, the problem surrounding how to generalize the use of nonmarket or environmental values does not start itself with the issue of benefit transfer (and its conduct). There is also the question of whether an adequate and standardized database of environmental values is being established particularly in an academic literature which up to now has been concerned largely with methodological developments (A. Randall, personal communication). Fulfilment of this empirical task may well lack the glamour attached to further methodological investigation of, for example, SP approaches but is nonetheless just as crucial (and perhaps more so) in terms of facilitating policy applications through benefits transfer.

3. EQUITY AND CBA

Although the origins of CBA can be traced back to middle of the nineteenth century, the body of modern-day welfare economics, which underlies CBA, was established in the 1930s and 1940s [for the history of CBA, see Persky (85)]. In particular, the Kaldor-Hicks *compensation principle* established the idea of hypothetical compensation as a practical rule for deciding on policies and projects in real-life contexts [Hicks (86, 87), Kaldor (88)]. This compensation principle established the rule that benefits (gains in human well-being) should exceed costs (losses in human well-being) for policies and projects to be approved. In recommending those proposals that confer the largest net benefits, CBA appears to neglect questions of distribution and, moreover, views the worth of a proposal in terms of the outcomes that it generates rather than say other dimensions such as justice [Randall (89)]. Given that equity and justice concerns often dominate discourse about social decisions, it has often struck critics as bordering on the perverse that CBA has chosen to focus its attention so squarely on efficiency.

Placing the spotlight on efficiency, however, does not necessarily reflect a judgment that equity, or distribution, does not matter. Rather it assumes that economic efficiency underlying the Kaldor-Hicks compensation test can or should be separated out from the issue of who gains and loses: the distributional incidence of costs and benefits. There are a number of reasons for this separation. In the round, net losers currently could be net winners for efficient projects implemented at some later date, a tenet that Persky (85) calls the *classical creed*. Perhaps the most prominent view is that, given the apparently robust theoretical foundations of the Kaldor-Hicks test, CBA should be confined to maximizing the cake so there is more to share around according to some morally or politically determined rule of distributional allocation. Put another way, equity or distribution matters but not for the purposes of recommending specific projects or policies. And although the emphasis on efficiency, on the face of it, might seem one-dimensional, from the standpoint of CBA's role as just one input into the decision-making process, this single-mindedness could be important. This is because one can always rely on the political process raising the equity issue, but not necessarily the efficiency issue, and so, in this sense, conventional CBA redresses the balance [Pearce et al. (8)].

Despite this prevailing view, now classic contributions to the modern theory of CBA [such as Little & Mirrlees (90), Dasgupta et al. (91), Squire & van der Tak (92)] have all embraced the notion that ways in which societal well-being can be described (the underlying social welfare function in CBA) are arbitrarily large, and some alternatives may involve giving different weight, on the basis on plausible societal preferences, to different distributional outcomes. Others have come at the problem from a somewhat different angle, noting that, in practice, equity and efficiency issues are hard to separate. For example, future generations cannot be compensated, potentially or otherwise, for climate change liabilities that we are currently creating, or at least the transfers that would comprise these reparations cannot be guaranteed, given that we have no control on the actions of intervening generations [Lind (93)]. Others, in a similar vein, have questioned the ease of transfers across international boundaries [Hepburn (94)]. A further tack points out that those transfer mechanisms that do exist are not costless, and so an empirical question is, What is the best way of addressing distributional concerns (i.e., inefficient transfers or inefficient projects) [Harberger (95)]?

What all these critiques amount to is a chipping away at the bedrock of the Kaldor-Hicks compensation test that supports the standard CBA approach. Put another way, if the separation assumption (of efficiency and equity) can no longer be sustained, or at least if it is less clear-cut, then this could serve as a rationale to consider distributional issues more fully in project selection or design. It still leaves room for debate about what form any distributional analysis should take. Kriström (96) characterizes the options as belonging to a hierarchy ranging from relatively simple steps such as identifying and cataloging how project-related costs and benefits are distributed to recalculating a project's net benefit using assigning explicit distributional weights to the benefits received and costs incurred by different societal groups. This gives rise to a revised social decision criterion that the project is worthy, on balance, if the sum of its equity-weighted net benefits is at least positive: $\sum_i a_i NB_i \geq 0$, where NB_i are net benefits (simply, benefits minus costs) and a_i are equity weights that vary across i individuals, households, or societal groups. An important feature of conventional CBA is now apparent. It assumes weights of unity are assigned to the net benefits of individuals regardless of who it is that receives a unit of benefit or suffers a unit of cost [Drèze & Stern (97), Brent (98)]. Starting with Fankhauser et al. (99), there has been a resurgence of interest in equity weights in the literature on the distribution of the burden of climate change damage across countries [see also Azar (100), Tol (101), Anthoff et al. (102)].

Debate about what form these weights should take has surrounded different conceivable functions and forms that describe social welfare and, more specifically, the measure of inequality aversion (reflecting in turn a judgment about how much more, in terms of its impact on social welfare, a dollar of climate change damage suffered by a poor person is worth than a dollar of damage endured by a rich person).[4] Comprehensive reviews of the literature on estimates of inequality aversion can be found in, for example, Pearce & Ulph (103) and Cowell & Gardiner (104). The findings to date suggest that equity weighting could make a significant difference to decision making using CBA, but the range of plausible values that these weights could take is possibly large. It is, in large part, this ambiguity that makes cost-benefit analysts reluctant about using equity weights. In other words, the concern is not that equity weights are irrelevant, but that, as a practical matter, we know too little about what values these weights should take. Most of these data thus far have been obtained by studies of revealed behavior of individuals when saving and governments with regard to redistributive policies combined with thought experiments as to what seems reasonable. SP methods have been less commonly used, although a handful of related studies have shown the potential for using these methods in the context of environmental equity [Atkinson et al. (105), Scarborough & Bennett (106)]. In the interim, some practitioners advocate looking at switching values: that is, by asking, What set of weights would be required to tip the balance between recommending that the project go ahead or not go ahead [Gramlich (107), Kanninen & Kriström (108)]? This does not make the controversy of weights go away but shifts it once more, albeit at a later stage, to the political domain.

[4]For example, one commonly used way to calculate equity weights can be written: $a_i = (\bar{Y}/Y_i)^e$, where $\bar{Y}$ is average or mean income per capita; Y_i is income of the ith individual or household (or group); and crucially e is the elasticity of the marginal utility of income or society's valuation of an increment to that individual's income. Intuitively, this elasticity is said to reflect society's degree of inequality aversion. That is, if $e = 0$, then each unit of benefit is awarded the same weight (i.e., unity) regardless of who it accrues to. Higher values of e boost the weight a unit (e.g., a dollar) of benefit is given for those with below average income and shrink it for those with above average income.

4. DISCOUNTING AND THE FUTURE

Discounting involves attaching a lower weight to a given unit (say $1) of future benefit (or cost) than to an equivalent present unit.[5] The weights are determined by time itself and by a discount rate, which is expressed as a percentage. Discounting is justified by the assumption that it is what people do, because they are impatient and the fact that capital is productive (i.e., can be invested now for some future return), and so accords with the basic value judgment of CBA that people should have what they want and prefer. CBA was primarily formulated for appraisals of projects whose lifetimes did not stretch further than the short to medium term [Hepburn (109)]. Environmental problems, however, typically have a longer time horizon and the effect of discounting is often to give very low (and possibly practically zero) weight to events in the far-off future. CBA then, by discounting, appears to make these problems, such as climate change, disappear: this has led to a debate over the so-called "tyranny of discounting" [Markandya & Pearce (110)] or "the discounting dilemma" [Hepburn (109), Groom et al. (111)].

The response of many economists to this debate has been to defend the moral rationale for the principle of discounting in part by pointing out the apparent ethical implications of not discounting. Much of this defense rests on the implication of the impoverishment of the current generation, which is destined ever more to reduce and postpone its own consumption [Koopmans (112), Olson & Bailey (113), see for a discussion, Groom et al. (111)], although this point has nuances [see, for example, Asheim & Buchholz (114)]. Yet the long-standing recognition within the economics profession of the possible limitations of CBA in evaluating proposals with distant time horizons can be found in Lind (115) in the context of U.S. energy policy. Proposals for an environmental discount rate, reflecting irreversible environmental losses, can be traced back to contributions from Krutilla & Fisher (116). More recently, an edited volume by Portney & Weyant (117) has explored a diverse number of alternatives to the discounting problem. Emerging from this renewed interest in a way around the seeming impasse is an increasingly influential view that the assumption of a constant discount rate in the conventional approach should be replaced, in some form, by time-declining discount rates. This call for a radical change in the practice of discounting comes from a number of different quarters and includes posing the discounting problem as a social choice or moral problem [Chichilnisky (118), Li & Löfgren (119)] or more simply as a reflection of individual preferences toward the future [Henderson & Bateman (120), Frederick et al. (121)]. But perhaps the most prominent rationale has at its heart uncertainty about future interest rates [Weitzman (122, 123)] or the state of the economy [Gollier (124)].

Although much of this discussion is rather abstruse, it has a very real implication: the socially correct discount rate, to be used by governments in investment and policy appraisal, is one that declines with time. The effect of this is to slow down the increase in the discount factor across time (relative to the case where the discount rate is constant). In other words, the discount rate itself changes with time, getting smaller as time progresses. The practical effect of this recent "... revolution in thinking about discounting ..." [Pearce et al. (8), p. 186] can be dramatic for representing long-term issues in CBA. For example, Newell & Pizer (125) conclude that the introduction of time-declining discount rates has the effect of doubling their estimate of the social costs of carbon, and Groom et al. (111) show that certain candidate forms of declining discount rates might change decisions about U.K. nuclear energy options while others do not. The implications of declining discount rates in a wider

[5] Discounting implies that the weight, w_t, to be attached to a gain or loss in any future year, t, is less than 1. More specifically, the discounting formula is: $w_t = \frac{1}{(1+s)^t}$, where s is the (social) discount rate. This discount factor, w_t, therefore shrinks as t gets larger (i.e., as gains and losses become more distant).

variety of countries are explored in Hepburn et al. (126).

Time-varying discount rates also have their own problems, and chief among these is time inconsistency where plans that are made at one point in time are contradicted or reversed by later behavior [Strotz (127)].[6] The indications are that certain forms of time-declining discount rates solve one problem (the tyranny issue) by creating another [Heal (128), Hepburn (129)]. Even so the crucial issues are the relative size of the gains from resolving the tyranny problem and the problems from the apparent resulting adoption of time-inconsistent policies. Groom et al. (111), for example, speculate whether these problems are more significant than policy inconsistencies and reversals elsewhere in the political domain.

The controversy that discounting can cause is nowhere more evident than in the dramatic findings of the Stern Review on the economics of climate change [Stern (130)] and the ensuing debate surrounding those conclusions. Much of this debate has centered on just one, albeit focal, facet of the review: its findings on the social costs of carbon, which indicate that, in the absence of concerted action, the world risks what might amount to a decline of between 5% and 20% of its global consumption now and forever. Compared to the apparently significantly lower costs of action to combat climate change also presented, it is little wonder that the review concluded that "the benefits of strong, early action far outweigh the economic costs of not acting" (130, p. vx). Subsequent debate has focused on the evidence that underpinned this central conclusion [Nordhaus (131), Weitzman (132), Dasgupta (133), Maddison (134), Tol & Yohe (135), Mendelsohn (136)], including the role of the social discount rate (SDR) and the fundamental reasons for discounting: pure time preference and the utility value of future increments in consumption.[7] Although this debate seems a rather narrow focus, at its heart are fundamental issues for the use of CBA in this grand setting.

SDR: social discount rate

The first is the increasing recognition that the climate change problem is perhaps truly nonmarginal in the sense that the entirety of decisions affecting this problem could end up shifting the development path that the world economy is on (with, e.g., business as usual global emissions of greenhouse gases possibly leading to considerably lower future consumption levels than now) [Weitzman (132), Hoel & Sterner (137)]. This has significant implications for the choice of discount rate that might be used to evaluate climate policy, and which itself depends on the path of future consumption. Thus, this basic insight complicates the climate discounting problem considerably as, for example, the Stern Review shows not least by moving the analytical focus away from the cost-benefit practitioner's comfort zone of basing the SDR on current market rates.

A second important facet of this discussion about discounting has turned on whether, in selecting component values for the SDR, a descriptive approach or a prescriptive approach should be used [Dietz et al. (138)], a distinction that also can be found in (139). Put another way, should climate-related investments be appraised respectively in the light of information about preferences for the future as revealed in actual economic decisions, or is there room for the analyst to make explicit moral judgments about intergenerational equity [for a discussion of the ways in which ethics and economics are interwoven in the climate change context, see Dietz et al. (140)]? The Stern Review (130) opts for the latter approach, for example, in arguing that while one factor determining the pure rate of time preference—the (small) risk that

[6]As an illustration, time consistency requires that generation A chooses a policy and that generation B acts in accordance with it. Generation B does not revise what generation A planned. If generation A's plans are revised by generation B, then generation A will not have optimized its behavior—what it intended for generation B will turn out to have been wrong [Pearce et al. (8)].

[7]More specifically this discussion focuses on the consumption discount rate (s). This is written as $s = \rho + \eta.g$, where ρ is the pure rate of time preference (or utility discount rate), η is the elasticity of the marginal utility of consumption, and g is the growth rate of per capita consumption.

QOV: quasi-option value

future generations will not exist—is a reason to discount another factor, pure impatience—preferring benefits now rather than later simply because these come sooner—is not. A low value for the pure rate of time preference follows naturally from explicitly taking this ethical perspective. The substance of Nordhaus (131) and Weitzman (132) is that there is, on the face of it, very little evidence that this moral reasoning is reflected in people's actual behavior and choices, and thus, the empirical evidence is that the pure rate of time should take a higher value. Similarly, Dasgupta argues that the cost-benefit conclusions follow from "... views on intergenerational equity rather than ... new climatic facts ..." (133, p. 4). Although this claim is disputed by, for example, Dietz et al. (138), it is clear that the discounting debate must confront profound questions about intergenerational equity, to quote Beckerman & Hepburn, "on which reasonable minds may differ" (141, p. 198). Moreover, in the view of those authors, the tools of environmental CBA have a more proactive role to play in deliberating between seemingly intractable viewpoints by, for example, the use of SP methods and so on to assess social preferences toward different climate change–related outcomes [see, for example, Saelon et al. (142)]. Even though fresh perspectives on the discounting debate might be worthwhile, there is also increasing recognition that a meaningful reflection, in cost-benefit terms, on the implication of future but uncertain climate catastrophes might require additional analytical tools [Weitzman (132)]. It is to these issues of risk and uncertainty that we now turn.

5. UNCERTAINTY AND IRREVERSIBILITY

Although costs and benefits are rarely known with certainty, the flip side of this—namely, uncertainty—whether about a physical impact or its monetary value is a question of extent as well as its nature. In some cases, a risk probability distribution might be known, yet such distributions differ according to their degree of sophistication. In other cases, those of genuine uncertainty, there is no known probability distribution. The ways in which risk and uncertainty have been integrated into CBA have not changed much over the years. In practice, this has meant use of sensitivity analysis or the calculation of expected values (where the decision maker is assumed to be risk neutral) and expected utilities (where the decision maker is assumed to be risk averse). However, the presence of uncertainty combined with other features of a decision, e.g., irreversibility about say biodiversity loss, has given rise to at least two interesting developments. These approaches vary in their linkage to conventional economics, but what they have in common is the provision of substance to the notion of a precautionary principle in guiding environmental decision making.

The first development arises from the existence of uncertainty and irreversibility—perhaps because funds committed cannot be uncommitted or because other effects of the policy cannot be reversed—combined with the potential to learn by delaying a decision. In environmental economics, this is mostly known as quasi-option value (QOV), although elsewhere, in financial economics, it has been called option value or real options [see, for example, Kolstad (143), Dixit & Pindyck (144)]. QOV is not a separate category of economic value [Freeman (145)]. Instead, it is the difference between the net benefits of making an optimal decision and one that is not optimal because the latter ignores the gains that may be made by delaying a decision and learning during the period of delay. Thus, a development option, which, for example, involves the permanent conversion of tropical forestland to agricultural land, must be debited with the potential forgone costs of not waiting to learn more about the benefits of forest conservation. How large the gain is from this process is essentially an empirical question, but, in the context of environmental policy, Pindyck (146), for example, speculates that the implications could be dramatic. A number of empirical studies have sought to use the notion of QOV to explain WTP/WTA disparities [Zhao & Kling (147)] in plantation forestry [Yap

(148)] and conservation decisions. For example, in a study of Costa Rica's tropical forests, Bulte et al. (149) show that once uncertainty and irreversibility are included, the country has too little forest cover. Nevertheless, in that instance, the authors also conclude that QOV remains empirically unimportant relative to other considerations, such as accounting for the presence of global externalities provided by Costa Rica's forests, a finding consistent with another study that sought to elicit QOV [Albers et al. (150)].

One interpretation of the QOV approach is that it urges more caution about losing environmental assets such as ecosystems. Although the approach uses formal economics to do this, other frameworks have sought to combine CBA explicitly with conservation criteria. One idea is a safe minimum standard (SMS) [Ciriacy-Wantrup (151), Bishop (152)]. What the emphasis on SMS does is to reverse the onus of proof, away from assuming that development is justified unless the costs to the environment are shown to be very high, to a presumption that conservation is the right option unless the sacrifice that it entails is very high. Farmer & Randall (153) and Randall (154) argue that the appeal of the SMS lies in it being an approach that may well command broad moral consensus for making decisions. From this perspective, CBA prevails, but as a sustainability threat draws closer, the SMS becomes the priority. This thinking appears to have influenced conservation policy in the United States (the Endangered Species Act) as well as Europe (the EU Habitats Directive) [Berrens (155), Pearce (156)]. A handful of contributions, beginning with Barbier et al. (157), have also proposed a cost-benefit rule, which is subject to a constraint that natural capital is kept intact overall, perhaps, for example, by requiring that no further degradation or loss of ecosystems should be tolerated. What this does is make the principle of sustainability applicable to a portfolio of projects. Thus, to the extent that any one project degrades or destroys an ecosystem, this must be "covered off" by improvements or additions to ecosystems elsewhere, i.e., so-called shadow or compensating projects. Roach & Wade (158) provide an empirical investigation of this resource compensation or equivalency in the context of habitats. In the United States, the legal notion of public trust has been used to support such measures where if certain resources are damaged then "... compensation takes the form of in-kind services..." [Kopp & Smith (159, p. 2)]. As with the SMS, even though resource compensation provides some trade-offs between costs and benefits, it plainly circumscribes cost-benefit thinking in a substantial way.

6. CBA, ITS CRITICS, AND THE REAL WORLD

We have noted previously the extensive academic literature on the theory and practice of CBA. Detailed official guidance on how to carry out CBA is much rarer and tends to be confined to those countries where CBA is part of the process of regulatory impact analysis (RIA), (or, sometimes, appraisal or assessment). The Organisation for Economic Co-operation and Development (OECD) has issued its own guidelines on RIA (160) and also maintains an Inventory of RIA Procedures (161). In the United Kingdom, RIA is mandatory for regulations, although guidance on how to value costs and benefits in monetary terms is given by the U.K. Treasury (162) in *The Green Book: Appraisal and Evaluation in Central Government*. Interestingly, that document also contains pioneering official guidance on using a declining discount rate for projects or policies whose lifetimes extend beyond 30 years. In the environmental policy context in the United States, CBA is widely used, and indeed, the U.S. EPA has its own extensive guidelines (163) for preparing economic analyses of regulations. The European Commission (EC) is committed to applying some form of cost-benefit test to its Directives (see, for example, Article 130r of the Treaty on European Union, 1992), yet in practice, formal CBA has until relatively recently been infrequent, especially in the environmental context. For EC regional schemes (structural and cohesion funds), however, a guidance document on CBA does exist. This focuses mainly on conventional project appraisal issues but does have a section on valuing environmental impacts in money terms, and this stress on the environment, and nonmarket impacts more generally, has more recently been expanded (164).

Environmental CBA in developing countries is becoming increasingly applied, particularly as a result of appraisal procedures at the World Bank for scrutinizing investments with the Bank's Operational Policy on the Economic Evaluation of Investment Operations providing the formal impetus [see, for example, Belli et al. (165)]. A rather different array of projects, under the heading of markets for environmental services, also reflects this trend in utilizing cost-benefit thinking in the Bank's environmental investment decisions. Under these arrangements, the sufferer (or beneficiary of conservation), perhaps through some intermediary, pays a polluter to change his or her behavior. There are well-known obstacles to these Coasean bargains, and moreover, the evidence is that so far the financial flows arising from these deals are comparatively small [Pearce (166)], notwithstanding some prominent examples such as those made under Costa Rica's Forest Law.

Despite ample evidence that cost-benefit thinking is increasingly prominent as an aid to making actual environmental decisions, there are also plenty of signs that the existence of procedures cannot be taken as an indication that CBA is used. Silva & Pagiola (167), for example, take a detailed look at how environmental valuation methods have been used to appraise more than 100 projects within the World Bank's environmental portfolio. Taking stock of these findings brings to mind the glass half-full or half-empty analogy. That is, there has been a significant increase in the use of environmental valuation in recent years. Yet, in the majority of projects surveyed, covering energy, transportation, water, and agricultural sectors, no valuation was employed. In a recent review of 74 RIAs issued by the U.S. EPA from 1982–1999, Hahn & Dudley (168) find that, although all of these regulations monetized at least some costs, only about half monetized some benefits. Fewer still (about a quarter, on average, across the period) provided a full monetized range of estimates of benefits, even though the number doing so increased notably over the sample period. More generally, OECD (161) states that despite the desirability of CBA, it is not used in many of its member countries because of the difficulties of placing money values on a comprehensive range of costs and benefits. Whether this reflects genuine complexities or provides a flag of convenience for downgrading the consideration of benefits is another matter. Ignoring CBA, or doing it poorly, has consequences, however. Devarajan et al. (169) report on a study investigating 1200 World Bank projects in 58 countries finding that the cost of not carrying out (formal) economic appraisal is possibly high, especially if policy distortions are widespread. Deininger et al. (170) estimate that each dollar spent on analysis prior to World Bank project initiation led to an increase of \$4–8 in terms of development impact. In a review of recent European Directives, Pearce (156) finds that few if any of these initiatives was subject to economic appraisal, and moreover, it is likely that, for the United Kingdom at least, these policies, such as the most recent amendments to the EU Bathing Water Directive, are unlikely to pass a cost-benefit test. It is encouraging, therefore, that, at least in the United Kingdom case, official cost-benefit appraisals of these Directives have emerged subsequently (see Environmental CBA in Action).

Discussions about the extent of the use of CBA, as well as the quality of the assessments that are undertaken, segue naturally into reflections on the political economy of CBA. The fact that decisions are often inconsistent with CBA can be squared with the reality that, in practice, CBA is only one input to the decision, and in various circumstances, other considerations trump economic thinking. It also may be that decision makers have taken an extreme stance on the criticisms of CBA, or its underlying welfare economics foundations, that exist. Gowdy (171) provides a useful and thorough summary of many of these criticisms. Debate has focused on, for example, CBA's underlying value judgment that individuals' preferences should count in any social decision-making rule with the environment proving to be a particularly controversial area, where, for example, claims for the primacy of citizens' preferences [Sagoff (172, 173)] are prominent. It has also been suggested that CBA is often, in reality, downgraded because it is not deliberative in the sense of ensuring groups have influence over some decision (in addition to having their costs and benefits counted in a cost-benefit appraisal) [Turner (174)] (see also Section 2.1.1, above). Others, however, go the other way: To the extent that groups or individuals are poorly informed about the environment, there are too many risks to allowing uninformed views to hold sway over decisions.

Though a great deal of the criticism of CBA is based on misunderstanding or overly simplistic characterizations, there is also much that has been written that could pose a genuine cause for reflection among cost-benefit practitioners. Both List (175) and Hanley & Shogren (176) have commented on the discrepancy between, on the one hand, the current esteem in which cost-benefit approaches appear to be held in public decision making and, on the other, the emerging evidence within the academic literature on anomalies about individual preferences that might diminish the "power of CBA as a source of advice" (176, p. 27). List (175) shows that preference anomalies are likely to be a matter of degree and, in part, are determined by experience, which is often lacking when environmental goods are not traded directly, a view shared by Sugden (177). For Hanley & Shogren (176), echoing Randall (89), the difficulties that these inconsistencies present do nothing to eliminate the basic recommendation that knowing whether the benefits of an action outweigh its costs is one critical element of making a good decision or not. Moreover, a CBA that can accommodate, in a sensible fashion, recent thinking about the behavioral complexity of those individuals whose preferences cost-benefit analysts are seeking to unravel might well be one component of the best of all worlds. Clearly, this raises difficulties in that this increases the complexity of CBA and decision makers might be constrained in obtaining the requisite expertise needed to realize the strengths of this more nuanced approach to appraisal [Pearce et al. (8)].

Just as likely, and perhaps more so, however, is that government and its constituent decision makers, rather than merely maximizing social well-being as assumed in CBA textbooks, are faced with political realities that necessitate the reconciliation of conflicting interests, and giving prominence to CBA might not be the best way of serving those ends [Pearce et al. (8)]. One way of conceptualizing this problem is to look at a political welfare function as opposed to the social welfare function implicit in CBA [Grossman & Helpman (178), Aidt (179)]. The former is a more general version of the latter and so reflects not just the government's

ENVIRONMENTAL CBA IN ACTION

Although the academic literature on environmental CBA is abound with contributions on theory and methods, it is often to the gray literature that we must turn for comprehensive and practical examples of CBA in action. In the United Kingdom, for example, all regulations are subject to a Regulatory Impact Assessment (RIA), which, in turn, must consider costs and benefits and their distribution. One such example is the recent appraisal of the U.K. implementation of the European Commission (EC) revised Bathing Water Directive (BWD) (see **http://www.defra.gov.uk/corporate/consult/bathingwaters/impact-assess.pdf**). The objective of this policy is the tightening of standards governing the quality of recreational waters, such as those around the U.K. coastline. The RIA itself is a detailed document outlining, for example, the physical impact assessment (on bathing water) and its valuation. The latter itself was based on the findings of a stated preference study of the willingness to pay of citizens for improvements in bathing water quality (and other related attributes). Interestingly, part of the RIA publication is a statement, to be signed by the relevant HM Government Minister, attesting to the veracity of the appraisal and, furthermore, that with respects to the policy outcome: "… I am satisfied … that the benefits justify the costs."

concern for the well-being of individuals but also the weight it attaches to the well-being of particular interest or pressure groups, the support of which the decision maker needs to stay in power or to realize some policy proposal. Although this helps to understand the political realities of decisions, it does not change the role of CBA, i.e., explaining how a decision should look if the economist's (social welfare function) approach is adopted [Pearce et al. (8)].

7. CONCLUSIONS

Cost-benefit analysis (CBA) has been developed over a long period of time, and most economists would argue that, even if decisions are not finally made on the basis of CBA, decisions should be informed by CBA such that it is at least a prominent input to decision making. Environmental CBA, or more precisely the more extensive use of CBA as a practical tool for decisions with environmental impacts, is a somewhat more recent development. Part of the reason for this is the sheer advance in the use, and understanding, of methods to uncover, in a variety of ways, the value of environmental impacts (and nonmarket impacts more generally). Some of these techniques estimate original values perhaps by looking at actual behavior (i.e., RPs) or intended behavior (i.e., SPs). Other techniques make use of (or transfer to use the jargon) the findings of existing studies and apply these to a new policy or project context, i.e., benefits transfer.

The routine estimation of monetary values reflecting changes in environmental assets as well as environmental services is only one part of the story of recent developments in environmental CBA. The uncertainty that characterizes environmental losses has led to an emphasis, in some quarters, on how precaution could enter into decision making in several ways such as a (strong) sustainability constraint—perhaps requiring resource compensation or a SMS that must be observed—or as an QOV where a development option must be debited with the potential forgone costs of not waiting to learn more about conservation benefits. Distributional concerns have also been prominent, and environmental problems, such as climate change, that effect the well-being of generations far into the future have been perhaps the chief impetus for new groundbreaking interpretations of the discounting dilemma. Similarly, concerns about the distribution of costs and benefits among rich and poor have also breathed new life in distributional CBA such that certain minimum requirements for practice have emerged, although some applications go much further.

Nearly all of these developments have been accompanied by critical analysis. For example, in the case of environmental valuation, the increasing use of these methods has resulted in, on the one hand, ever greater sophistication in application and, on the other hand, scrutiny regarding their validity and reliability. Though much of the evidence to date suggests that there is a virtuous circle between translating the lessons from such scrutiny into practical guidance for future applications, challenges remain, especially in the case of benefits transfer. While this suggests that CBA is not without its limitations as a procedure for decision making, this does not absolve policy makers from neglecting this tool in choosing policy and projects. Indeed, there is evidence that the absence of considering costs and benefits in decisions, including those which affect the environment, has tangible and adverse consequences. Environmental CBA, or interim versions of it such as benefit assessment, appears currently to be enjoying an upsurge of influence, although even the modest aim for cost-benefit thinking to be an input to decisions is far from fulfilled. Thus, there remains more to do in terms of understanding why CBA is relied upon to inform some environmental decisions but not in others with the oft-cited data difficulties only being one piece in this puzzle. A related challenge is to ensure that CBA, in official practice, is a satisfactory (if pragmatic) reflection of the more important recent developments in the academic literature.

SUMMARY POINTS

1. Although cost benefit analysis, or CBA, has a long tradition, the extensive use of CBA as a practical tool for guiding decisions with environmental impacts is a more recent development.
2. In large part, progress in environmental CBA has been made possible by the sheer scale of advance in the use, and understanding, of methods to uncover the monetary value of environmental (and related) impacts. This progress, however, has been accompanied by searching and varied debates about the merits of these methods.
3. There has been renewed interest in how to integrate equity or distributional concerns within cost-benefit thinking (especially in relation to how environmental burdens and benefits are distributed), with guidelines for minimum standards of practice emerging.
4. Discounting (or giving less weight to) future benefits and costs remains controversial as evidenced by ongoing disputes surrounding the social costs of future climate change.
5. This point notwithstanding, novel breakthroughs in thinking about how to value the future have offered fresh hope for reconciling discounting practice with a concern for future generations.
6. The uncertainty that characterizes environmental losses has led to a varied search for ways in which the cost-benefit approach can be combined with precaution about the loss of environmental assets.
7. Environmental CBA appears to be enjoying an upsurge in its influence, although even the modest aim for cost-benefit thinking to be an input to public policy decisions is, in many instances, far from fulfilled.
8. While the uptake of CBA has been accompanied by a challenging critical debate about methods and its correct place in environmental decision making, the consideration of costs and benefits of actions remains a crucial input to policy.

FUTURE ISSUES

1. Continued developments in environmental valuation methods are to be welcomed, although in all likelihood such advances are subject to diminishing returns. At least as important is the need to establish empirical values that can be used or transferred across a variety of policy contexts.
2. Genuine progress has been made in valuing health risks. There is now a pressing need to understand more fully the value of ecosystem services, taking into account the complexities that ecological systems present.
3. Interest in distributional CBA is growing, but many practitioners are still to be convinced of its significance for appraisals.
4. Lively debates about the equity weights that might be attached to gains and losses of different groups across space and time could be aided with greater recourse to stated preference (SP) methods.

5. Integrating CBA with concerns for precaution about the loss of, for example, ecosystems and broader societal objectives, such as sustainable development, remains work in progress.
6. There remains more to do in terms of understanding why and how CBA is relied upon to actually inform some environmental decisions but not others.
7. A related challenge is to ensure that CBA, in official practice, is a satisfactory (if pragmatic) reflection of the more important developments in the academic literature.

DISCLOSURE STATEMENT

The authors are not aware of any biases that might be perceived as affecting the objectivity of this review.

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LITERATURE CITED

1. Sugden R, Williams A. 1978. *The Principles of Practical Cost-Benefit Analysis*. Oxford: Oxford Univ. Press
2. Just R, Hueth D, Schmitz A. 2004. *The Welfare Economics of Public Policy: A Practical Guide to Project and Policy Evaluation*. Cheltenham: Edward Elgar
3. Boardman A, Greenberg D, Vining A, Weimer D. 2006. *Cost-Benefit Analysis: Concepts and Practice*. Upper Saddle River, NJ: Prentice Hall
4. Ray A. 1984. *Cost-Benefit Analysis: Issues and Methodologies*. Baltimore, MD: Johns Hopkins Univ. Press
5. Londero EH. 2003. *Shadow Prices for Project Appraisal: Theory and Practice*. Cheltenham: Edward Elgar
6. Johansson P-O. 1993. *Cost-Benefit Analysis of Environmental Change*. Cambridge: Cambridge Univ. Press
7. Hanley N, Spash C. 1993. *Cost-Benefit Analysis and the Environment*. Cheltenham: Edward Elgar
8. Pearce DW, Atkinson G, Mourato S. 2006. *Cost-Benefit Analysis and the Environment: Recent Developments*. Paris: Organ. Econ. Co-Op. Dev.
9. Bateman I, Carson RT, Day B, Hanemann M, Hanley N, et al. 2002. *Economic Valuation with Stated Preference Techniques: A Manual*. Cheltenham: Edward Elgar
10. Mitchell RC, Carson RT. 1989. *Using Surveys to Value Public Goods: The Contingent Valuation Method*. Washington, DC: Resour. Future
11. Champ PA, Boyle KJ, Brown TC, eds. 2003. *A Primer on Nonmarket Valuation*. Dordrecht: Kluwer
12. Alberini A, Kahn J, eds. 2006. *Handbook on Contingent Valuation*. Cheltenham: Edward Elgar
13. Carson RT. 2008. *Contingent Valuation: A Comprehensive Bibliography and History*. Cheltenham: Edward Elgar. In press
14. Arrow K, Solow RM, Portney P, Leamer E, Radner R, Schuman H. 1993. Report of the NOAA panel on contingent valuation. *Fed. Regist.* 58:4016–64
15. Smith VK. 2006. Fifty years of contingent valuation. See Ref. 12, pp. 7–65
16. Carson RT. 2000. Contingent valuation: a user's guide. *Environ. Sci. Technol.* 34:1413–18
17. Hausman J, ed. 1993. *Contingent Valuation: A Critical Assessment*. Amsterdam: North-Holland

18. Diamond PA, Hausman JA. 1994. Contingent valuation: Is some number better than no number? *J. Econ. Perspect.* 8:45–64
19. Portney PR. 1994. The contingent valuation debate: Why economists should care. *J. Econ. Perspect.* 8:3–18
20. Hanemann WM. 1994. Valuing the environment through contingent valuation. *J. Econ. Perspect.* 8:19–43
21. Boyle KJ, Bergstrom JC, eds. 1999. *Doubt, Doubts, and Doubters: The Genesis of a New Research Agenda*? Oxford: Oxford Univ. Press
22. Carson RT, Flores NE, Meade NF. 2001. Contingent valuation: controversies and evidence. *Environ. Resour. Econ.* 19:173–210
23. Fischoff B, Furby L. 1988. Measuring values: a conceptual framework for interpreting transactions with special reference to the contingent valuation of visibility. *J. Risk Uncertain.* 1:147–84
24. Schkade DA, Payne JW. 1993. Where do the numbers come from? How people respond to contingent valuation questions. See Ref. 17, pp. 271–93
25. Sagoff M. 1994. Should preferences count? *Land Econ.* 70:127–44
26. Haddad B, Howarth R. 2006. Protest bids, commensurability and substitution: contingent valuation and ecological economics. See Ref. 12, pp. 133–49
27. Bateman IJ, Willis KG. 1999. *Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU and Developing Countries*. Oxford: Oxford Univ. Press
28. Boyle KJ. 2003. Contingent valuation in practice. See Ref. 11, pp. 111–69
29. Lancaster K. 1966. A new approach to consumer theory. *J. Polit. Econ.* 74:132–57
30. Louviere J, Hensher DA, Swait JD. 2000. *Stated Choice Methods: Analysis and Application*. Cambridge: Cambridge Univ. Press
31. Bennett J, Blamey R. 2001. *The Choice Modelling Approach to Environmental Valuation*. Cheltenham: Edward Elgar
32. Hensher DA, Rose JM, Greene WH. 2005. *Applied Choice Analysis: A Primer*. Cambridge: Cambridge Univ. Press
33. Hanley N, Mourato S, Wright R. 2001. Choice modelling approaches: a superior alternative for environmental valuation? *J. Econ. Surv.* 15:433–60
34. Foster V, Mourato S. 2002. Testing for consistency in contingent ranking experiments. *J. Environ. Econ. Manag.* 44:309–28
35. Dickie M, Fisher A, Gerking S. 1987. Market transactions and hypothetical demand data: a comparative study. *J. Am. Stat. Assoc.* 82:69–75
36. Murphy JJ, Stevens TH, Weatherhead D. 2005. Is cheap talk effective at eliminating hypothetical bias in a provision point mechanism? *Environ. Res. Econ.* 30:327–43
37. List JA, Gallet C. 2001. What experimental protocol influence disparities between actual and hypothetical stated values? *Environ. Resour. Econ.* 20:241–54
38. Champ PA, Bishop RC, Brown TC, McCollum DW. 1997. Using donation mechanisms to value nonuse benefits from public goods. *J. Environ. Econ. Manag.* 33:151–62
39. Murphy JJ, Allen PG, Stevens TH, Weatherhead D. 2005. A meta-analysis of hypothetical bias in stated preference valuation. *Environ. Resour. Econ.* 30:313–25
40. Cummings RG, Taylor LO. 1999. Unbiased value estimates for environmental goods: a cheap talk design for the contingent valuation method. *Am. Econ. Rev.* 89:649–65
41. List JA. 2001. Do explicit warnings eliminate the hypothetical bias in elicitation procedures? Evidence from field auctions for sportscards. *Am. Econ. Rev.* 91:1498–507
42. Poe GL, Clark JE, Rondeau D, Schulze WD. 2002. Provision point mechanisms and field validity tests of contingent valuation. *Environ. Resour. Econ.* 23:105–31
43. Lusk JL. 2003. Effects of cheap talk on consumer willingness-to-pay for golden rice. *Am. J. Agric. Econ.* 85:840–56
44. Atkinson G, Mourato S, Morse-Jones S, Provins A. 2008. *When to take "no" for an answer? Further evidence on using entreaties to reduce protests in stated preference studies*. Presented at the Eur. Assoc. Environ. Resour. Econ. Annu. Conf., Gothenburg
45. Sagoff M. 1998. Aggregation and deliberation in valuing environmental public goods: a look beyond contingent valuation. *Ecol. Econ.* 24:213–30

46. Prato T. 1999. Multiple attribute decision analysis for ecosystem management. *Ecol. Econ.* 30:207–22
47. Macmillan D, Philip L, Hanley N, Alvarez-Farizo B. 2002. Valuing the nonmarket benefits of wild goose conservation: a comparison of interview and group-based approaches. *Ecol. Econ.* 43:49–59
48. Alvarez-Farizo B, Hanley N, Barberan R, Lazaro A. 2007. Choice modeling at the "Market Stall": individual versus collective interest in environmental valuation. *Ecol. Econ.* 60:743–51
49. Bolt K, Ruta G, Sarraf M. 2005. *Estimating the Cost of Environmental Degradation: A Training Manual in English, French and Arabic*. Washington, DC: World Bank
50. Krupnick A. 2004. Valuing health outcomes: policy choices and technical issues. *Rep. Resourc. Future*, Washington, DC
51. Robinson L. 2007. How US government agencies value mortality risk reductions. *Rev. Environ. Econ. Policy* 1:283–99
52. Krupnick A. 2007. Mortality-risk valuation and age: stated preference evidence. *Rev. Environ. Econ. Policy* 1:261–82
53. Organ. Econ. Co-op. Dev. 2006. *Economic Valuation of Environmental Health Risks to Children*. Paris: OECD
54. Daily GC. 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Washington, DC: Island
55. Millenn. Ecosyst. Assess. 2005. *Ecosystems and Human Wellbeing*. Washington, DC: MEA/World Resourc. Inst.
56. Pearce DW, Pearce C. 2001. *The Value of Forest Ecosystems*. Montreal: Secret. UN Conv. Biol. Divers.
57. Daily GC, Söderqvist T, Aniyar S, Arrow K, Dasgupta P, et al. 2000. The value of nature and the nature of value. *Science* 289:395–96
58. Costanza R, d'Arge R, de Groot R, Farber SC, Grasso M, et al. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387:253–60
59. Bockstael NE, Freeman AM, Kopp RJ, Portney P, Smith VK. 2000. On measuring economic values for nature. *Environ. Sci. Technol.* 32:1384–89
60. Arrow K, Daily GC, Dasgupta P, Levin S, Maler K-G, et al. 2000. Managing ecosystem resources. *Environ. Sci. Technol.* 34:1401–6
61. Bishop RC. 2003. Where to from here? See Ref. 11, pp. 537–66
62. Haab TC, McConnell KE. 2002. *Valuing Environmental and Natural Resources: The Econometrics of Non-Market Valuation*. Cheltenham: Edward Elgar
63. Parsons GR. 2003. The Travel Cost Model. See Ref. 11, pp. 269–330
64. Day B. 2002. Valuing visits to game parks in South Africa. In *Valuing the Environment in Developing Countries: Case Studies*, ed. DW Pearce, C Pearce, C Palmer, pp. 236–73. Cheltenham: Edward Elgar
65. Taylor LO. 2003. The Hedonic Method. See Ref. 11, pp. 331–94
66. Bateman IJ, Lovett AA, Brainard JS. 2005. *Applied Environmental Economics: A GIS Approach to Cost-Benefit Analysis*. Cambridge: Cambridge Univ. Press
67. Day BH, Bateman IJ, Lake IR. 2007. Beyond implicit prices: recovering theoretically consistent and transferable values for noise avoidance from a hedonic property price model. *Environ. Resour. Econ.* 37:211–32
68. Cameron TA. 1992. Combining contingent valuation and travel cost data for the valuation of nonmarket goods. *Land Econ.* 68:302–17
69. Ben-Akiva M, Morikawa T. 1991. Estimation of switching models from revealed preferences and stated intentions. *Transp. Res. A* 24:485–95
70. Swait J, Louviere J. 1993. The role of the scale parameter in the estimation and comparison of multinomial logit models. *J. Mark. Res.* 30:305–14
71. Adamowicz WJ, Louviere J, Williams M. 1994. Combining revealed and stated preference methods for valuing environmental amenities. *J. Environ. Econ. Manag.* 26:271–92
72. Kling C. 1997. The gains from combining travel cost and contingent valuation data to value nonmarket goods. *Land Econ.* 73:428–39
73. Boyle KJ, Bergstrom JC. 1992. Benefits transfer studies: myths, pragmatism and idealism. *Water Resour. Res.* 28:657–63
74. Desvousges W, Johnson FR, Banzaf HS. 1998. *Environmental Policy Analysis with Limited Information: Principles and Applications of the Transfer Method*. Cheltenham: Edward Elgar

75. Navrud S, Ready R, eds. 2007. *Environmental Value Transfer: Issues and Methods*. Dordrecht, Neth.: Springer
76. Ready R, Navrud S, Day B, Dubourg WR, Machado F, et al. 2004. Benefit transfer in Europe: How reliable are transfers between countries? *Environ. Resour. Econ.* 29:67–82
77. Brouwer R. 2000. Environmental value transfer: state of the art and future prospects. *Ecol. Econ.* 32:137–52
78. Rosenberger, Loomis J. 2003. Benefit transfer. See Ref. 11, pp. 445–82
79. Brouwer R, Bateman IJ. 2005. The temporal stability and transferability of models of willingness to pay for flood control and wetland conservation. *Water Resour. Res.* 41:W03017
80. Barton DN, Mourato S. 2003. Transferring the benefits of avoided health effects from water pollution between Portugal and Costa Rica. *Environ. Dev. Econ.* 8:351–71
81. Alberini A, Cropper M, Fu T-T, Krupnick A, Liu J-T, et al. 1997. Valuing health effects of air pollution in developing countries: the case of Taiwan. *J. Environ. Econ. Manag.* 34:107–26
82. Lovett AA, Brainard JS, Bateman IJ. 1997. Improving benefit transfer demand functions: a GIS approach. *J. Environ. Manag.* 51:373–89
83. Brookshire DS. 1992. *Issues Regarding Benefits Transfer*. Presented at Assoc. Environ. Resour. Econ. Workshop, Utah
84. Kristofersson D, Navrud S. 2005. Validity tests of benefit transfer—Are we performing the wrong tests? *Environ. Resour. Econ.* 30:279–86
85. Persky. 2001. Retrospectives: cost-benefit analysis and the classical creed. *J. Econ. Perspect.* 15:199–208
86. Hicks JR. 1939. Foundations of welfare economics. *Econ. J.* 49:696–712
87. Hicks JR. 1943. The four consumer's surpluses. *Rev. Econ. Stud.* 11:31–41
88. Kaldor N. 1939. Welfare propositions of economics and interpersonal comparisons of utility. *Econ. J.* 49:549–52
89. Randall A. 1999. Taking benefits and costs seriously. In *The International Yearbook of Environmental and Resource Economics (1999/2000)*, ed. H Folmer, T Tietenberg, pp. 250–72. Cheltenham: Edward Elgar
90. Little I, Mirrlees J. 1974. *Project Appraisal and Planning for Developing Countries*. Oxford: Oxford Univ. Press
91. Dasgupta P, Marglin S, Sen A. 1972. *Guidelines for Project Evaluation*. Vienna: UNIDO
92. Squire L, van der Tak H. 1974. *Economic Analysis of Projects*. Baltimore: Johns Hopkins Univ. Press
93. Lind R. 1999. Analysis for intergenerational decision making. See Ref. 117, pp. 173–80
94. Hepburn C. 25 Feb. 2005. *Lords Select Committee on Economic Affairs Written Evidence Memorandum*. London: House of Lords
95. Harberger AC. 1978. On the use of distributional weights in social cost-benefit analysis. *J. Polit. Econ.* 86:S87–120
96. Kriström B. 2005. Framework for assessing the distribution of financial effects of environmental policies. In *The Distribution of Benefits and Costs of Environmental Policies*, ed. Y Serret, N Johnstone, pp. 79–137. Cheltenham: Edward Elgar
97. Drèze J, Stern N. 1990. Policy reform, shadow prices and market prices. *J. Public Econ.* 42:1–45
98. Brent R. 1996. *Applied Cost-Benefit Analysis*. Cheltenham: Edward Elgar
99. Fankhauser S, Tol RSJ, Pearce DW. 1996. The aggregation of climate change damages: a welfare theoretic approach. *Environ. Resour. Econ.* 10:249–66
100. Azar C. 1999. Weight factors in cost-benefit analysis of climate change. *Environ. Resour. Econ.* 13:249–68
101. Tol RSJ. 2001. Equitable cost-benefit analysis of climate change. *Ecol. Econ.* 36:71–85
102. Anthoff D, Hepburn C, Tol R. 2006. *Equity weighting and the marginal damage costs of climate change*. Work. Pap. Res. Unit Sustain. Glob. Change (FNU), Hamburg Univ.
103. Pearce DW, Ulph D. 1999. A social discount rate for the United Kingdom. In *Economics and Environment: Essays on Ecological Economics and Sustainable Development*, ed. DW Pearce, pp. 268–85. Cheltenham: Edward Elgar
104. Cowell F, Gardiner K. 1999. *Welfare Weights*. London: Off. Fair Trading
105. Atkinson G, Machado F, Mourato S. 2000. Balancing competing principles of environmental equity. *Environ. Plan. A* 32:1791–806
106. Scarborough H, Bennett J. 2006. *Estimating intergenerational distribution preferences using choice modelling*. Work. Pap. Ser., School Account., Econ. Financ., Deakin Univ.

107. Gramlich EM. 1990. *A Guide to Benefit-Cost Analysis*. Prospect Heights, IL: Waveland
108. Kanninen BJ, Kriström B. 1993. *Welfare benefit estimation and income distribution*. Beijer Inst. Ecol. Econ. Discuss. Pap. Ser. Stockholm
109. Hepburn C. 2007. Valuing the far-off future: discounting and its alternatives. See Ref. 180, pp. 109–24
110. Markandya A, Pearce DW. 1988. *Environmental Considerations and the Choice of the Discount Rate in Developing Countries*. Washington, DC: World Bank
111. Groom B, Hepburn C, Koundouri P, Pearce DW. 2005. Declining discount rates: the long and the short of it. *Environ. Resour. Econ.* 33:445–93
112. Koopmans TC. 1965. On the concept of optimal economic growth. *Pontif. Acad. Sci. Scripta Varia* 28:225–300
113. Olson M, Bailey M. 1981. Positive time preference. *J. Polit. Econ.* 89:1–25
114. Asheim GB, Buchholz W. 2003. The malleability of undiscounted utilitarianism as a criterion of intergenerational justice. *Economica* 70:405–22
115. Lind R, ed. 1982. *Discounting for Time and Risk in Energy Policy*. Washington, DC: RFF/Johns Hopkins
116. Krutilla J, Fisher A. 1974. *The Economics of Natural Environments*. Baltimore: Johns Hopkins Univ. Press
117. Portney PR, Weyant JP, eds. 1999. *Discounting and Intergenerational Equity*. Washington, DC: Resour. Future
118. Chichilnisky G. 1996. An axiomatic approach to sustainable development. *Soc. Choice Welf.* 13:231–57
119. Li CZ, Löfgren K-G. 2000. Renewable resources and economic sustainability: a dynamic analysis with heterogeneous time preferences. *J. Environ. Econ. Manag.* 40:236–50
120. Henderson N, Bateman I. 1995. Empirical and public choice evidence for hyperbolic social discount rates and the implications for intergenerational discounting. *Environ. Resour. Econ.* 5:413–23
121. Frederick S, Loewenstein G, O'Donoghue T. 2002. Time discounting and time preference: a critical review. *J. Econ. Lit.* XL:351–401
122. Weitzman M. 1999. "Just keep on discounting, but. . . ." See Ref. 117, pp. 23–30
123. Weitzman M. 1998. Why the far-distant future should be discounted at its lowest possible rate. *J. Environ. Econ. Manag.* 36:201–8
124. Gollier C. 2002. Discounting an uncertain future. *J. Public Econ.* 85:149–66
125. Newell RG, Pizer WA. 2003. Discounting the distant future: How much do uncertain rates increase valuations? *J. Environ. Econ. Manag.* 46:52–71
126. Hepburn C, Koundouri P, Panopoulou E, Pantelidis T. 2008. Social discounting under uncertainty: a cross country comparison. *J. Environ. Econ. Manag.* In press
127. Strotz R. 1956. Myopia and inconsistency in dynamic utility maximisation. *Rev. Econ. Stud.* 23:165–80
128. Heal G. 1998. *Valuing the Future: Economic Theory and Sustainability*. New York: Columbia Univ. Press
129. Hepburn C. 2003. Resource collapse and hyperbolic discounting. Dep. Econ. Discuss. Pap. Oxford Univ., Oxford
130. Stern N. 2007. *The Economics of Climate Change—The Stern Review*. Cambridge: Cambridge Univ. Press
131. Nordhaus WD. 2007. A review of the Stern Review on the economics of climate change. *J. Econ. Lit.* 45:686–702
132. Weitzman M. 2007. A review of the Stern Review on the economics of climate change. *J. Econ. Lit.* 45:703–24
133. Dasgupta P. 2006. Comments on the Stern Review's economics of climate change. *Natl. Inst. Econ. Rev.* 199(Jan.):4–5
134. Maddison. 2006. *Further comments on the Stern Review*. Work. Pap. Dep. Econ., Univ. Birmingham
135. Tol RSJ, Yohe G. 2007. A review of the Stern Review. *World Econ.* 7:233–50
136. Mendelsohn RO. 2006. A critique of the Stern report. *Regulation* (Winter 2006–2007):42–46
137. Hoel M, Sterner T. 2006. *Discounting and relative prices: assessing future environmental damages*. Discuss. Pap., Resour. Future, Washington, DC
138. Dietz S, Anderson D, Stern N, Taylor C, Zenghelis D. 2007. Right for the right reasons: a final rejoinder on the Stern Review. *World Econ.* 8:229–58
139. Intergov. Panel Clim. Change. 1996. *Climate Change 1995: Economic and Social Dimensions of Climate Change*. Cambridge: Cambridge Univ. Press

140. Dietz S, Hepburn C, Stern N. 2008. Economics, ethics and climate change. In *Arguments for a Better World: Essays in Honour of Amartya Sen*, ed. K Basu, R Kanbur. Oxford: Oxford Univ. Press. In press
141. Beckerman W, Hepburn C. 2007. Ethics of the discount rate in the Stern Review on the economics of climate change. *World Econ.* 8:187–210
142. Saelon H, Atkinson G, Dietz S, Helguson J, Hepburn C. 2008. *Risk, inequality and time: public attitudes and the economics of climate change*. Work. Pap. Dep. Econ., Univ. Oxford. In press
143. Kolstad CD. 1996. Learning and stock effects in environmental regulation: the case of greenhouse gas emissions. *J. Environ. Econ. Manag.* 18:1–18
144. Dixit A, Pindyck R. 1994. *Investment Under Uncertainty*. Princeton: Princeton Univ. Press
145. Freeman AMI. 2003. *The Measurement of Environmental and Resource Values*. Washington, DC: Resour. Future
146. Pindyck R. 2007. Uncertainty in environmental economics. *Rev. Environ. Econ. Policy* 1:45–65
147. Zhao J, Kling C. 2001. A new explanation for the WTP/WTA disparity. *Econ. Lett.* 73:293–300
148. Yap R. 2004. Option valuation of Philippine forest plantation leases. *Environ. Dev. Econ.* 9:315–33
149. Bulte E, van Soest D, van Kooten C, Schipper R. 2002. Forest conservation in Costa Rica when nonuse benefits are uncertain and rising. *Am. J. Agric. Econ.* 84:150–60
150. Albers H, Fisher A, Hanemann W. 1996. Valuation and management of tropical forests: implications of uncertainty and irreversibility. *Environ. Resour. Econ.* 8:39–61
151. Ciriacy-Wantrup CV. 1968. *Resource Conservation: Economics and Policies*. Berkeley: Univ. California
152. Bishop R. 1978. Endangered species and uncertainty: the economics of a safe minimum standard. *Am. J. Agric. Econ.* 60:10–18
153. Farmer MC, Randall A. 1998. The rationality of a safe minimum standard. *Land Econ.* 74:287–302
154. Randall A. 2007. Benefit cost analysis and a safe minimum standard. See Ref. 180, pp. 91–105
155. Berrens RP. 2001. The safe minimum standard of conservation and endangered species: a review. *Environ. Conserv.* 28:104–16
156. Pearce DW. 2004. Does European Union environmental policy pass a cost-benefit test? *World Econ.* 5:115–38
157. Barbier E, Markandya A, Pearce DW. 1990. Environmental sustainability and cost-benefit analysis. *Environ. Plan. A* 22:1259–66
158. Roach B, Wade WW. 2006. Policy evaluation of natural resource injuries using habitat equivalency analysis. *Ecol. Econ.* 58:421–33
159. Kopp R, Smith VK. 1993. Introduction. In *Valuing Natural Assets: The Economics of Natural Resource Damage Assessment*, ed. R Kopp, VK Smith, pp. 1–9. Washington, DC: Resour. Future
160. Organ. Econ. Co-op. Dev. 1997. *Regulatory Impact Analysis: Best Practice in OECD Countries*. Paris: OECD
161. Organ. Econ. Co-op. Dev. 2004. *Regulatory Impact Analysis (RIA) Inventory*. Paris: OECD
162. UK HM Treas. 2003. *The Green Book: Appraisal and Evaluation in Central Government*. London: HM Treas.
163. US Environ. Prot. Agency. 2000. *Guidelines for Preparing Economic Analyses*. Washington, DC: US EPA
164. Eur. Comm. 2008. *Guide to Cost-Benefit Analysis of Investment Projects*. Brussels: Dir. Gen. Reg. Policy
165. Belli P, Anderson J, Barnum H, Dixon J, Tan J-P. 1998. *Handbook on Economic Analysis of Investment Operations*. Washington, DC: World Bank
166. Pearce DW. 2007. Do we really care about biodiversity? *Environ. Resour. Econ.* 37:313–33
167. Silva P, Pagiola S. 2003. *A review of the valuation of environmental costs and benefits in World Bank projects*. Environ. Econ. Ser. Pap. 94, Environ. Dep. World Bank, Washington, DC
168. Hahn RW, Dudley PM. 2007. How well does the U.S. government do benefit-cost analysis? *Rev. Environ. Econ. Policy* 1:192–211
169. Devarajan S, Squire L, Suthiwart-Narueput S. 1995. *Reviving project appraisal at the World Bank*. Washington, DC: World Bank
170. Deininger K, Squire L, Basu S. 1998. Does economic analysis improve the quality of foreign assistance? *World Bank Econ. Rev.* 12(3):385–418
171. Gowdy J. 2004. The revolution in welfare economics and its implications for environmental valuation and policy. *Land Econ.* 80:239–57

172. Sagoff M. 1988. *The Economy of the Earth*. Cambridge: Cambridge Univ. Press
173. Sagoff M. 2004. *Price, Principle and the Environment*. Cambridge: Cambridge Univ. Press
174. Turner RK. 2007. Limits to CBA in UK and European environmental policy: retrospects and future prospects. *Environ. Resour. Econ.* 37:253–69
175. List J. 2005. Scientific numerology, preference anomalies, and environmental policymaking. *Environ. Resour. Econ.* 32:35–53
176. Hanley N, Shogren J. 2005. Is cost-benefit analysis anomaly-proof? *Environ. Resour. Econ.* 32:13–34
177. Sugden R. 2005. Coping with preference anomalies in cost-benefit analysis: a market-simulation approach. *Environ. Resour. Econ.* 32:129–60
178. Grossman G, Helpman E. 1994. Protection for sale. *Am. Econ. Rev.* 84:833–50
179. Aidt T. 1998. Political internalisation of economic externalities and environmental policy. *J. Public Econ.* 69:1–16
180. Atkinson G, Dietz S, Neumayer E, eds. 2007. *Handbook of Sustainable Development*. Cheltenham: Edward Elgar
181. Neil H, Cummings R, Ganderton P, Harrison G, McGuckin T. 1994. Hypothetical surveys and real economic outcomes. *Land Econ.* 70:145–70

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