Cost−benefit analysis (CBA)

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1 Introduction

A cost−benefit analysis (CBA) can be defined as an economic technique applied to public decision−making that attempts to quantify the advantages (benefits) and disadvantages (costs) associated with a particular project or policy. This technique has been used to analyze policies affecting transportation, urban regeneration, agriculture, public health, criminal justice, defence, education, and the environment. The appeal of CBA is that by monetizing the benefits of the policy, it is possible to compare and/or aggregate many different categories of benefits with one another, and with the costs of the policy.

A program or project would be deemed acceptable if its benefits outweigh the costs. In this sense, a CBA informs decision−makers of both the direction and the strength of social preferences, and thereby also of the social desirability of a project or policy. Out of a number of alternative programs being examined, CBA would recommend choosing the one with the largest net benefits, where net benefits are defined as the benefits minus the costs. Cost−benefit analysis—or more specifically, the estimation of costs and benefits that is required to perform a CBA—also allows one to determine the socially optimal size of the program or project, i.e., the one that maximizes net benefits. At the socially optimum program, the marginal benefits of the program will be equal to its marginal costs.

It should be noted that in order to determine the net effect of a proposed policy, we must first identify those persons who stand to gain and those who stand to lose from the implementation of the policy, and then estimate their respective gains or losses. For all practical purposes, CBA adopts the principle of a potential Pareto improvement, where winners can potentially compensate the losers. In total the benefits and the costs examined in a CBA are the aggregate gains and losses experienced by the individuals who comprise society. Thus, if no individual is made better off by the public program, there are no benefits associated with it. If no one is made worse off by the program, there are no costs.

We also wish to emphasize that benefits and costs, even though they are expressed in monetary terms in CBA, go well beyond changes in individuals' incomes. If someone's well−being is improved because of cleaner air—through a reduction in the physical discomfort or symptoms associated with pollution exposures, for example—this person experiences a benefit even though his or her income has not changed.

As mentioned, CBA focuses on the aggregate costs and benefits of a policy or programme. One limitation of CBA is that it ignores distributional issues, including whether those who care about the benefits of the policy can actually afford to pay for it. Another problem with CBA is that it must monetize categories of costs and benefits that are experienced at different times. Economists recommend discounting costs and benefits incurred in the future to compute their present value, but doing so requires choosing a discount rate. Present values are very sensitive to the choice of the discount rate and to any assumption about whether the discount rate does or does not stay constant over time. We discuss the issue of discounting at length in section 2.3.3 below, while other limitations and pitfalls of CBA are discussed in section 2.3.2.

CBA typically makes use of monetary valuation methods. The benefits of projects or policies will often take the form of improvements in individuals' welfare that are not traded in markets, and for which there is no market price. We discuss below non−market valuation techniques that can be used to circumvent this problem. As we discuss below, which specific non−market valuation technique is appropriate for one's benefit estimation exercise depends on the context, on the category of benefits being considered, and on whether the market or individuals' behaviours are consistent with the assumptions of the method being used. Based on these considerations, we conclude that the estimation of the benefits of a policy is generally a difficult task, much more so than the estimation of the costs of a policy. Despite these limitations, we believe that cost−benefit analysis is a useful tool for policymakers, and that it provides important information that should be taken into account in the policymaking process.

2 Methodology

The first order of business of a CBA is to identify possible costs and benefits of the proposed policy or project, along with the parties who incur such costs or experience such benefits. Next, the analyst needs to place a monetary value on the various categories of costs and benefits, and aggregate the various categories of monetized costs and benefits into cost and benefit totals. This task is complicated by the fact that many policies entails costs and benefits that are incurred in different time periods—which requires discounting them to compute their present values—and that many categories of benefits are non−market goods, and as such are not bought and sold in regular marketplaces.

3 Process

In the next sections we discuss categories of costs and benefits, and possible approaches for estimating the latter in the absence of documented transactions in regular marketplaces.

3.1 Categories of Costs

When economists estimate the costs of a policy or project for the purpose of conducting a CBA, they refer to the *social* costs of the policy, which may well be different from the *private* costs of the policy. Briefly, the total social costs are the sum of all opportunity costs incurred by society because of the new policy. The opportunity costs are the value of pf the goods and services lost by society resulting from the use of resources to comply with and implement the policy, and from reductions in output (if any). For simplicity, in what follows we place the costs of the policy into five broad categories:

- real−resource costs, including compliance costs,
- government regulatory costs
- social welfare losses
- transitional costs
- indirect costs.

By real−resource cost, we mean all of the resources that are used up to implement the program or policy. By real−resource *compliance* costs, we mean all of the resources that must be expended for complying with the regulatory aspects of the program. Suppose, for example, that we were considering an environmental policy. In this case, the real−resource compliance costs would be costs associated with purchasing and installing new pollution control equipment, changing the production process by using different inputs or different mixtures of inputs, and capturing the waste products, reusing or re−selling them, or disposing of them in approved landfills or incinerators.

These costs would, therefore, be borne by firms or plants, as well as private citizens. For example, if a mobile source policy required car owners to subject their vehicles to emissions testing, (i) the fee paid to take the test, (ii) the value of the time spent by the car owner to go to and wait at the testing facility, and (iii) the cost of any repairs to the emissions control equipment would be examples of possible real−resource compliance costs incurred by these individuals.

The second category of costs—government regulatory costs—includes the monitoring, administrative, and enforcement costs associated with the policy, especially when the latter has a regulatory aspect. The cost of setting up new markets (e.g., an emissions trading program) would be included in this category of costs. However, any incentives paid out by the government to private firms or individuals to induce them to undertake certain actions (or refrain from undertaking them) are generally *not* counted among the costs of the policy.[1] Presumably, these are transfers from one group of agents (e.g., taxpayers) to another, and so they would be costs the former and benefits to the latter. To avoid double−counting them, they are thus generally excluded from the cost−benefit calculus.

By social welfare losses, one generally means the consumer and producer losses associated with possible rises in prices or decreases in output that occur as a result of the policy.

Presumably, if the cost of a product were to rise as a result of the policy (consider, for example, a policy that regulates the use of pesticides in agriculture), consumers would have to buy less or switch to a substitute, which would lead to a fall in consumer surplus. The extent of the fall in consumer surplus depends on the market structure, and on the elasticities of demand and supply for this product. Once these are known, it is possible to obtain an estimate of the change in consumer surplus. The task of estimating this category of costs is complicated by the difficulty of estimating the demand and supply for the product, and any structural changes in the market for the good whose price or supply is affected by the policy.

When the US Environmental Protection Agency was called to cancel or allow continued registration of pesticides for use on crops under the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA), the law explicitly instructed the agency to compare the costs and benefits of allowing continued use of the pesticides in questions. Producer and consumer losses due to higher produce prices were calculated and are documented in the Record of Decision that accompany the final outcome for each of the pesticides considered by the Agency (Cropper et al., 1992).

Transitional social costs include the value of all the resources that are displaced by the policy, and the private costs of reallocating these resources. These include, among others, unemployment and firm closings, shifts of resources to other markets (for example, capital might be attracted to other markets if the returns on the presently considered market decline), transaction costs, and any disruptions in productions that occur as a result of the policy.

It should be kept in mind that any job losses associated with the policy might be offset by the additional hires that must be made to ensure compliance with the policy. In addition, when computing the costs due to job losses, it is important to examine the duration of the spell of unemployment, the size of the group of workers affected, and the cost of re−training programs and unemployment benefits.

The remaining category is comprised of indirect costs, such as adverse effects that the policy may have on product quality, factor productivity, innovation, discouraged investment, and changes in markets indirectly affected by the policy. In some cases, some of these effects will already be captured by the losses in producer and consumer welfare. It is, therefore, important to avoid double−counting of these costs.

3.2 Categories of Benefits

In theory, the benefits of a policy are correctly captured by the beneficiaries' willingness to pay for the policy. The total benefits of the policy are the sum of the individual beneficiaries' WTP.

This implies that the first order of business for the analyst who wishes to estimate the benefits of this policy is to identify the beneficiaries of the policy. The next step involves quantifying the physical effects of the policy. This may require the estimation of epidemiological dose−response or concentration−response functions[2] (if the effects of the policy are on human health), hydrological models (if the policy regulates water), transportation models (if the policy regulates congestion or alters roads), etc., depending on the context. Once the physical effects are established, a monetary value must be attached to them.

To illustrate, suppose one was considering a policy that seeks to reduce emissions of pollutants into the air. Presumably, effects on human health would be an important consequence of the policy. One would need a concentration−response function to predict the reduction in the cases of illness associated with pollution exposure, and would need then to multiply this number by the value of (avoiding) a case of illness.

Developing an exhaustive list of the possible benefits of the policies covered by the Sustainability−A−Test project is beyond the scope of this document. For illustrative purposes, we report a list of possible benefits associated with an *environmental* policy in table 1, along with methods commonly used for placing a value on them. We wish to emphasize that the possible benefits of a policy depend on the policy and on the context, and that assistance from a variety of disciplines (epidemiology, engineering, etc.) may be needed to form a comprehensive list of the possible benefits of the policy under consideration.

Table 1. Example of categories of benefits of environmental policies

Benefit category	Example of service flows affected by the policy	Possible monetary valuation methods
Human health benefits: morbidity and mortality risks	Reduced risk of cancer, reduced risk of respiratory symptoms	–– averting behaviour –– contingent valuation –– hedonic pricing methods –– cost of illness
Amenities	Visibility affected by air quality	–– averting behaviour –– contingent valuation –– hedonic pricing methods
Ecological benefits: market products	Provision of food, fuel, timber, fibre, fur	Market approaches
Ecological benefits: recreation and aesthetics	Viewing, fishing, boating, swimming, hiking, etc.	–– production function –– contingent valuation –– hedonic pricing methods –– travel cost method
Ecological benefits: ecosystem services	Flood moderation, climate moderation, water filtration, sediment trapping, groundwater recharge, soil fertilization, pest control	–– production function –– averting behaviours –– hedonic pricing methods
Ecological benefits: existence and bequest values	No associated services (passive use values)	Contingent valuation
Materials damage	––	–– Averting behaviour –– market approaches

We conclude this section by pointing out that sometimes scientifically sound information is not available for estimating some of the possible benefits of a policy. If so, analysts may be forced to restrict attention to other benefits. Moore (1995) comments on the categories of benefits examined by the US Environmental Protection Agency when it considered changes to an existing air quality standard for sulphur dioxide. Of the 48 categories of possible benefits, the US EPA had data to do a partial assessment for 8 categories. An additional 36 categories were believed to have possible benefits, but no assessment was possible.

In some cases, the analyst may choose to omit certain categories of possible benefits of the policy because they are deemed to be negligible. When the analyst chooses or is forced to omit a possible category of benefits from the CBA, it is important that the reason(s) for this decision be fully disclosed and discussed, as this decision may form the basis for subsequent sensitivity analyses.

3.3 Estimation of the Benefits of a Policy

The theoretically correct way to measure the benefits of a proposed policy is to ask the beneficiaries of the policy what is their willingness to pay (WTP) for it. Willingness to pay is the amount that must be subtracted from the person's income to leave this individual just as well off following the implementation of the policy as before it.[3]

A. Market Approaches

In some cases, it is possible to determine the WTP for the proposed policy by using information available in regular markets. For example, if an environmental or agricultural policy results in increased agricultural output, the benefits of the policy should be equal to the change in crop, multiplied by the market price of the crop. Conversely, if the policy prevents a decline in agricultural output, as is the case with soil conservation policies, the benefits of the policy are equal to the loss in yield avoided by implementing the policy. When valuing ecological benefits of a proposed policy, it might be possible to identify service flows that can be bought and sold competitively as factors of production or final consumption goods. Increased productivity of farmland, rangeland, and forestland, for example, may provide significant market benefits, as may commercial fish species and timber.

An extension of this approach considers the ecosystem services that natural biomes provide. This concept is based on the idea that ecosystems provide services to humans not only through their direct and on−site uses (such as recreational uses, agriculture, production of timber and biomass). More importantly, the benefits of an ecosystem are also transmitted through the interaction with other ecosystems.[4] This includes the role of soils or forests in sequestering carbon dioxide and contributing to a stable macro− and micro−climate, the role of many ecosystems in maintaining biodiversity or the filtering of rainwater that percolates through the soil into the groundwater. While the value of such services is more difficult to assess than the value of direct human uses, research indicates that these values can indeed be significant (Balmford et al. 2002, Costanza et al. 1997).

In other cases, it is possible to measure the defensive or damage averting behavior and costs incurred by individuals in response to pollution. It should be kept in mind, however, that averting expenditures sustained are merely a lower bound of the benefits of the proposed policy.

B. Non−market valuation techniques: Stated Preference Approaches

In many cases, WTP cannot be measured easily using market prices, because the impacts of the policy are not traded in regular markets. Consider, for example, an urban regeneration policies that, among other things, alters traffic patterns and reduces emissions from mobile sources. The improvement in visibility and avoided illnesses due to the reduction in air pollution are examples of goods or benefits that are not exchanged in regular markets. Economists have devised a number of techniques—known as non−market valuation methods—to circumvent the lack of markets for these goods or benefits, and to arrive at a value for them. It should be kept in mind that certain methods may be suited for only certain types of non−market benefits.

Contingent valuation is an example of one such method. Contingent valuation is a survey−based approach that asks individuals to report their WTP for a public program that would deliver a public good or a specified improvement in environmental quality (Mitchell and Carson, 1989; Bateman et al., 2002). The public good or the environmental quality improvement is hypothetical and is described to the respondent in the course of the survey. No actual payment takes place, making contingent valuation a stated−preference approach, in that it relies on what respondents *say* they would do under hypothetical circumstances. Contingent valuation is the only method capable of capturing non−use values, i.e., the fact that someone may be willing to pay to protect a natural resource even if he or she does not use it, and remains the only technique capable of placing a value on environmental improvements that are outside of the range of available data.

Carson et al. (2000) document over 5000 papers and articles employing or studying the contingent valuation method. Stated−preference studies have been conducted to value changes in visibility (Chestnut and Rowe, 1990; Tolley et al., 1985), changes in drinking water quality (Carson and Mitchell, 2000), groundwater protection (Bergstrom, Boyle and Poe, 2001; Görlach and Interwies, 2003), recreational services (Hanley et al., 2003) and changes in health effects attributable to pollution (Alberini et al., 2004).

In recent high−quality applications of contingent valuation, respondents have been asked to consider a referendum on a ballot about the proposed public program, and have been asked whether they would vote in favor or against the proposed program if its implementation would imply a cost of €X for their household. The proposed program would be implemented only if there was a majority of "yes" votes, and would be abandoned (implying no additional costs to the household) if the majority of the votes were against the program. In addition to being an intuitive approach that reduces the cognitive burden placed on respondents when providing information about their WTP for the program, this approach has been endorsed by the NOAA Panel on Contingent Valuation (Arrow et al., 1993). Another advantage of the dichotomous choice approach is that it mimics behavior in regular markets, where people usually purchase, or decline to purchase, a good at the posted prices. It also closely resembles people's experience with political markets and propositions on a ballot.

Economic theory shows that the dichotomous−choice referendum approach is incentive−compatible (Hoehn and Randall, 1987), which means that it is in the respondent's best interest to answer truthfully to the vote question.[5] It may be argued that in the US most people are familiar with local referenda and are comfortable answering a referendum CV payment question. Recently, researchers based in the Netherlands (Polomé, Geurts and van der Veen, 2004) have expressed concern over the appropriateness of such an argument at other locales where individuals do not frequently participate in referenda, as is the case in some European countries.

It is important to note that the dichotomous choice approach does not observe WTP directly: At best, we can infer that the respondent's WTP amount was greater than the bid value (if the respondent is in favor of the program) or less than the bid amount (if the respondent votes against the plan), and form broad intervals around the respondent's WTP amount. Mean WTP is estimated after fitting binary data models of the responses.

To improve the precision of the WTP estimates, in recent years researchers have introduced follow−up questions to the dichotomous choice payment question (e.g., Hanemann, Loomis, and Kanninen, 1991). To illustrate, consider a respondent who states he is not willing to pay $10 for the proposed plan. The follow−up question might ask him if he would pay $5. If the respondent answers "no" to both questions, it is assumed that his WTP amount falls between 0 and $5. If the respondent answers "no" to the initial question, and "yes" to the follow−up questions, it is assumed that his WTP amount falls between $5 and $10. The bid level offered in the follow−up question will be greater than that offered in the initial payment question if the answer to the initial payment question is "yes."

Recent studies (see for instance Alberini et al., 1997b) have examined WTP for government programs, finding that mean WTP estimated after the follow−up questions can be lower than that implied by the responses to the initial payment question. A possible explanation for this finding is that some respondent may treat the suggested cost of the project as a signal for the quality of the program and/or might erroneously believe that the program to be valued in the follow−up is different from the initial one.[6]

Herriges and Shogren (1996) raise doubts about whether the follow−up payment question elicits information about the original willingness to pay amount. They propose that respondents update their original WTP amount with information about the cost of the program, as revealed by the initial bid. The amount underlying the response to the follow−up bid, therefore, would be a weighted average of the original willingness to pay and the initial bid amount.

In many CV studies, researchers have found that the majority of the responses to the initial and follow−up payment questions are of the "yes"−"yes" and "no"−"no" variety, with fewer respondents offering one positive and one negative answer. DeShazo (2002) spells out the predicted probabilities of the various response categories under four distinct theories (the framing model, which derives from prospect theory; the cost expectation model; the strategic behaviour model; yea−saying; and anchoring, or starting point bias) and uses data from three contingent valuation studies to empirically discriminate between these competing hypotheses.

Contingent valuation has also been criticized on the grounds that it "asks a hypothetical question" and thus obtains "a hypothetical answer." Accordingly, recent research has examined whether CV produces results that are consistent with those obtained using revealed−preference approaches, such as the travel cost method, the hedonic pricing method or other consumer behavior approaches. Carson et al. (1996) gather 83 studies that result in 616 comparisons between WTP from contingent valuation with the corresponding estimates from revealed preference studies. The commodities being valued include beach recreation, hunting and fishing, boating, workplace risk reductions, housing inside and outside earthquake zones, air pollution, and public pest control programs, among others.[7] The revealed preference methods covered in this meta−analysis include the travel cost, hedonic pricing and averting behaviour methods. Carson et al. find that the estimates of WTP from contingent valuation questions are highly correlated to the estimate from the companion revealed preference studies. They also find that the CV estimates of WTP are slightly lower than those from the companion revealed preference studies. The mean WTP CV/RP ratio is 0.89 for the complete dataset, 0.77 for a 5% trimmed dataset, and 0.92 for a weighted dataset.

Because contingent valuation is entirely dependent on the hypothetical scenario presented to the respondent, and on implicit or explicit assumptions about the availability of substitutes for the proposed program (see Arrow et al., 1993), the approach has been criticized on the grounds of the possible biases created by the way in which the scenario is presented to the respondent.

In 1992, in the wake of the Exxon Valdez oil spill, the National Oceanic and Atmospheric Administration assembled a panel of well−known economists, asking them to examine the appropriateness of contingent valuation for the estimation of damages to natural resources. The standards set out by the so−called NOAA Panel for contingent valuation studies, although originally intended for natural resource damage assessment, have been adopted by many practitioners. We summarize the NOAA Panel guidelines, and our assessment of the research that has sought to investigate these recommendations in table 2 below.

Table 2. Contingent valuation survey guidelines of the NOAA Panel on Contingent Valuation.

Recommendation by the NOAA Panel	**Our comments**
Face to face interviews (in which case one must pre−test for interviewer effects), with possible allowance for telephone interviews when appropriate; probability sampling; minimize non−response rates	The NOAA panel explicitly allows the use of quota or even convenience samples for preliminary testing of specific experimental variations. Face to face interviews are generally expensive to conduct, but may be necessary when the topic of the survey and the complexity of the good to be valued by the respondents require explanations and the use of visual aids, and when the population to be surveyed is difficult to reach otherwise. If telephone or mail surveys are conducted, we would recommend that the researcher keeps a track of those persons who were contacted but declined to participate in the survey, compare

	the characteristics of these persons with those of the people who did participate in the survey, and attempt to correct, if possible, for self−selection into the sample. “Drop−out” cards or follow−up phone calls can be used to obtain information about those persons who chose not to participate in the survey. Alternatively, one may use census information about the residents of a given area.
Conservative design (when aspects of the survey design and analysis of the responses are ambiguous, opt for interpretations that tend to underestimate WTP)	This suggestion seems appropriate for natural resource damage assessment, but less appropriate for other policy applications, especially when the WTP are interpreted within a “precautionary principle” context.
Elicit WTP rather than WTA	This recommendation is motivated by the fact that WTA tends to be much larger than WTP, and that WTA questions can encourage protest behaviour. We recommend eliciting WTP, keeping in mind that is possible to ask people to report their WTP to *avoid* a situation that would make them worse off.
Dichotomous choice referendum format	The dichotomous choice approach has been shown to be incentive compatible when the payment question is phrased as a referendum on a ballot. The dichotomous choice approach can, of course, be used to value a private good, such as reduction ni the respondent’s risk of dying or contracting an illness), in which case the respondent is usually asked whether or not he would buy the good at a given price. Unfortunately, in the latter context the dichotomous choice approach can no longer be proven to be incentive compatible. Polomé et al. (2003) have recently raised the question whether it is appropriate to use the referendum format at locales where people are not accustomed to referenda, such as the Netherlands.
No−answer or would not vote	a.Carson et al. (1998) investigate the effect of including this option, finding that recoding would−not−votes responses to votes against the program produces the same percentage of votes in favour and against as with the standard dichotomous choice approach; b. Champ et al. (2004) report results of a study that does not Carson et al.’s finding, and conclude that whether or not the no−answer option is comparable to a “would vote against” or “no” is likely to depend on the study. c. Mattson and Li (1995), Ready et al. (1995), Alberini et al. (2003), and others experiment

	with response formats allowing uncertainty or multiple answers. Alberini et al. (2003) point out that the interpretation of the uncertain responses is not unambiguous, and that the estimates of WTP for the good or policy can differ dramatically, depending on the statistical model fit to the responses.
Incorporate follow−up questions investigating the specific reasons why the respondent answered “yes” or “no” to the payment questions	This practice is followed by many CV survey practitioners.
Remind the respondent of (undamaged) substitute commodities	
Remind the respondent of budget constraint	This practice is widely followed by many CV survey practitioners. Studies based on split samples (Loomis et al., 1994b; Loomis et al., 1996) have found no discernible effect of such budget reminders on WTP. When budget constraint reminders have been folded into an explicit warning about hypothetical bias, they were found in some cases to exacerbate the possible positive hypothetical bias (Aadland and Caplan, 2003a, 2003b).
Test the effect of photographs	
Allow adequate time lapse from the accident that injured the natural resource	The NOAA Panel report does not specify exactly how long this time should be. Moreover, it is not clear how this requirement would be applied when the CV survey is for policy purposes, and not for natural resource damage assessment purposes.
Cross tabulate WTP with variables such as income, prior knowledge of the site, prior interest in the site (visitation rates), attitudes toward the environment, attitudes towards big business, distance to the site, understanding of the site, belief of the scenarios, ability or willingness to perform the valuation task.	These cross tabulations can be substituted with regression analyses and are part of normal checks of the internal validity of the responses.

Contingent valuation studies have been conducted in many European countries, but only in a few cases have the estimates of WTP from these studies constituted the grounds for policy. For example, Bishop and Romano (1998) survey a total of 12 CV studies conducted in Italy from 1983 to 1998. Most of these studies are concerned with hunting and fishing, and with the valuation of amenities and natural resources. The samples are generally small, and studies are conducted primarily as exercises. Romano (personal communication, 2001) explains that in Italy contingent valuation is not legally recognized by the law or by public agencies. This is in sharp contrast with the US, where, for example, the US Department of the Interior regards estimates of WTP from contingent valuation studies as a “rebuttable presumption” in its natural resource damage assessment cases.

In the UK, contingent valuation surveys have been conducted to place a value on a variety of public goods, environmental quality, and transportation policies and safety programs. For example, estimates of the Value of a Statistical Life from contingent valuation surveys (e.g., Jones−Lee, 1989) have formed the basis for transportation safety policy and are currently being examined by DEFRA for environmental policy analyses.

Contingent valuation has also been applied in accession countries. For example, several contingent valuation studies have been conducted in the Czech Republic since the mid−1990s eliciting WTP for improved air quality, water quality, household waste, flood prevention, morbidity, mortality risks, and landscape. In Poland, CV was used to value programs that would reduce eutrification in the Baltic Sea region. Our assessment of these studies is that the valuation exercise was—with very few exceptions—well accepted by the respondents, despite the relative lack of familiarity with markets and market behaviors. The use of contingent valuation is not required nor recognized by government agencies in Poland or Hungary, although we are aware of one CV study conducted in Hungary that was recognized by one government agency. The agency, however, ceased to exist in 2002. In the Czech Republic, contingent valuation studies have been conducted for research purposes and sometimes been funded by certain ministries, and the WTP figures have been used by certain organizations to make a case for or against specific policies. A study on the cement factor in Tman, for example, non−governmental organizations have used the public's opposition to the new plant—which was documented through the CV study—in its campaign against the plant.

C. Revealed−preference Methods

Revealed−preference studies use information about actual individual behavior to infer the value of a non−market good. The recreational benefits of a resource, for example, or the benefits associated with visiting a cultural heritage site, can be sometimes measured using recreational demand models. In the single−site **travel cost** model, individuals are assumed to derive utility from consumption X and from visits r to a recreational site, such as a fishing or hunting site. Visits depend on quality of the recreational site, in the sense that they increase with the quality of the site, and when the quality is the absolutely worst possible, the number of visits will be zero. This is commonly referred to as the weak complementarity assumption. Individuals maximize utility by the choice of consumption, the number of trips, and—assuming that their work hours are flexible—leisure time, subject to a budget constraint. Assuming that the travel time and the time at the site are given and exogenous, it is straightforward to derive demand functions for consumption, leisure, and for the number of trips, which depend on non−work income (y), the individual's wage rate (w), the out−of−pocket cost of a trip (p_r), and the quality of the recreational site.[8]

The demand for visits to the recreational site can also be reformulated to depend on the full price of the trip, which includes out−of−pocket costs, plus the opportunity cost of time. The opportunity cost of time has usually been imputed to be a fraction (usually about one−third) of the individual wage rate, but recent studies (Feather and Shaw, 1999) have pointed out that the value of time may well be higher than the wage rate for those individuals who work more hours than they would like to, and is lower than the wage rate for those individuals who work fewer hours than they would like to.

A change in quality at the site is assumed to shift out the demand function, in the sense that at any given trip prices more trips to the site will be undertaken, so that the corresponding change in surplus can be calculated. The easiest way to estimate the demand for trips is to collect data about trips to the site from a sample of individuals who live at different distance from the site and/or have different opportunity costs of time. Since their price per trip varies, it is now possible to regress trips taken on price per trip and other individual characteristics thought to affect visitation frequencies. It is important that the price of visiting a substitute site be included in the right−hand side of the regression model to avoid obtaining a biased coefficient of price per trip. The benefit of the amenities in its current condition and at the current trip price is thus the consumer surplus, as shown in Figure 1.A.

Figure 1.A

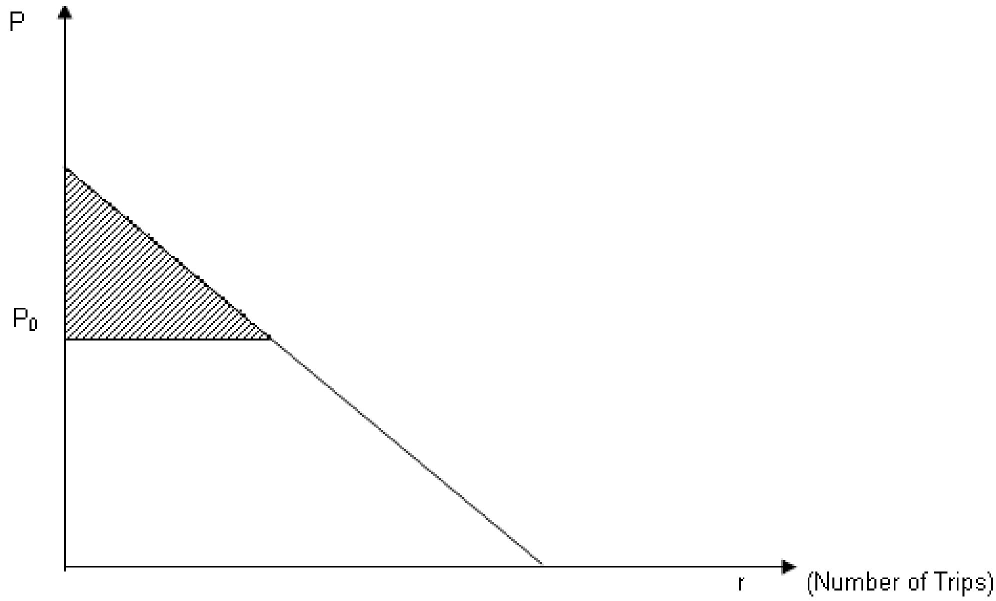
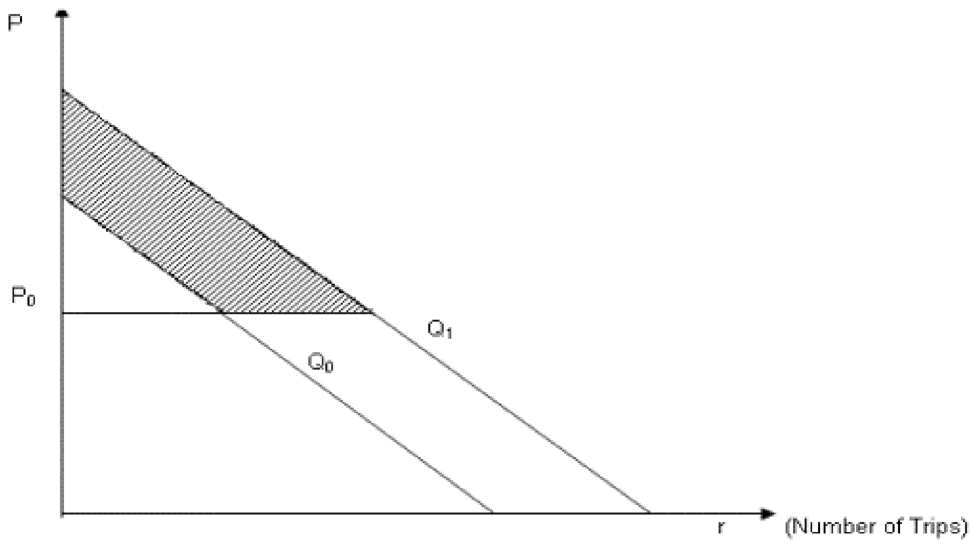


Figure 1.B



To estimate the effect of a change of quality, two approaches are possible: The first is to observe visitation rates at different points in time, before and after the quality change. The second, which is applied whenever the former is not possible, is to ask people how many times they would visit the site at the improved environmental quality conditions. Presumably, at the improved quality level the demand function would shift out, as shown in Figure 2.B, and the value of the improvement is the change in consumer surplus, shown by the shaded area.

Travel cost models work well for estimating changes in the number of trips over a period of time. They are limited in the their ability to model the recreationist’s choice among competing sites, in which case it is better to resort to **models of discrete choice** among alternatives. The latter use conditional logit or nested logit models to explain the decision to recreate at a specific site as compared to alternative substitutes. The model considers travel cost and environmental quality variables associated with all competing sites.

There are many difficulties associated with the use of models of discrete choice among recreational sites. The first is that estimation of these models depends crucially on how the choice set is defined. The choice set is usually constructed by the researcher, which makes omissions or inclusion of redundant sites possible. Nested logit models are useful because they relax partially the assumption of independence of irrelevant alternatives (IIA), but the estimates of welfare change depend crucially on how the nesting structure has been set up.

Finally, we wish to remind the reader than both travel cost and models of choice among alternative sites are limited in the sense that can capture only the use values of a resource, but fail to capture non−use values.

Another serious limitation of the travel cost method is that is best suited to value localised environmental goods that have some amenity value. It is not suited for situations where environmental impacts are widespread (such as climate change), or where the environmental impact has little impact on the amenity (such as the loss of a particular species).

Hedonic property value models assert that individuals perceive housing units as bundles of attributes and derive different levels of utility from different combinations of these attributes. When transactions are made, individuals make tradeoffs between money and attributes that reveal that marginal values of these attributes. To estimate these marginal values, one gathers data about property values by using individual property sales in real estate markets. The price of a parcel or a house is then the dependent variable in a regression on the structural characteristics of the house, neighbourhood characteristics, and environmental quality. Formally, the regression equation is:

(1) [illegible] ,

where *p* signifies the price of the house or parcel, **x** is a vector of structural characteristics of the house, **N** is a vector of neighbourhood characteristics, and **E** is a vector of environmental quality or other policy variables.

The coefficients on the attributes allow the analysts the recover the marginal values of those attributes. The hedonic housing price method has been used to place a value to environmental quality, climate change (Maddison and Bigano, 2003), and urban policies, such as traffic calming and pedestrian zone schemes (Curto and Simonotti, 1994).

The approach rests on two key assumptions. First, individuals are assumed to be perfectly aware of the level of environmental quality, climate, etc. at the locale where the property is located. In practice, however, there may be differences in how the environmental quality attributes are perceived by scientists and individuals, and it is *individuals’* perceptions that matter in determining prices. Second, the hedonic pricing approach assumes functioning housing markets, including a sufficient number of transactions as well as sufficient demand and supply for the market to clear,[9] and no barriers to entry.

Even if these assumptions hold, hedonic pricing studies are fraught with econometric difficulties. For example, it may be difficult to disentangle the bundle of factors that influence house prices, since many of these will often occur in conjunction. For example, in the vicinity of urban brownfield sites, there is not only the threat of contaminated soils, but also a higher crime rate, visual impacts on the cityscape etc. (This problem can be mitigated by performing a repeat−sale study: if documentation is available about multiple transactions of the same house or parcel, both before and after a certain policy or even that is believed to have affected price, in a repeated−sale study the dependent variable is the change in price from one transaction to the next. Assuming that all else about the parcel and the neighbourhood is the same, the change in price (if any) is ascribed to the policy or event of interest, or to a change in environmental quality.)

Hedonic price models are also very sensitive to the choice of functional form and to the definition of the extent of the market. Even more important, hedonic pricing studies estimate the market equilibrium locus, which reflects both demand and supply for the attribute (e.g., environmental quality) being studied. But to estimate willingness to pay—and not just the implicit marginal price of each attribute—it is necessary to identify the entire demand schedule, which can be accomplished if data from multiple markets are available or if the researcher is willing to make assumption about the functional form of the individuals’ utility function.

Compensating wage studies apply the hedonic pricing method within the context of labour markets. In a typical compensating wage study, a worker’s wage rate or income is regressed on education, work experience, tenure with the current employer, occupation and industry, gender, union status, etc. *and* other factors of interest that are believed to be captured into the wages, such as climate or the risk of dying on the job or climate. The coefficient on the latter variable is used to estimate the Value of a Statistical Life (VSL), i.e., the rate at which individuals are prepared to trade off income for risk reductions. The VSL is a key input into the estimation of the mortality benefits of policies that save lives, such as transportation policies or certain environmental and workplace safety policies.

Viscusi (1993) recommends running the regression model

(2) [illegible]

Where *w* is the wage rate (or a transformation of it), **x** is a vector of individual characteristics or job attributes thought to influence the wage rate, p is the risk of dying on the job, q is the risk of a non−fatal injury, and WC is the worker’s compensation in the event of an accident (which means that q×WC is the *expected* worker compensation). The coefficient d allows one to recover an estimate of the VSL.

In order to estimate the VSL, however, one must rely on a number of stringent assumptions, including that the workers’ risk is measured correctly, and that perceived risk is equal to objective risk. Moreover, the majority of the compensating wage studies surveyed in Viscusi (1993) and Viscusi and Aldy (2003) ignore self−selection of workers into riskier (or safer jobs), and the fact that, once again, compensating wage studies identify market equilibrium, and not workers’ preferences based on their entire demand for safety schedule (see Alberini et al., 2004, for a discussion).

Finally, we wish to mention two methods that are frequently used to value the human health effects of certain policies, namely the **averting expenditure** and the **cost−of−illness** methods. The averting expenditure method simply measures the costs incurred by individuals to protect themselves from human health risks. For example, if drinking water were contaminated, individuals would turn to drinking bottled water. The cost of bottled water is thus used to measure the benefits of eliminating pollution in the drinking water. Similarly, if individuals installed soundproofing materials in their homes to offset the noise from nearby traffic, the cost of purchasing and installing these materials would be interpreted as a measure of the benefits of a noise−reducing policy or of re−routing traffic. It should be kept in mind, however, that these measures *underestimate* the true benefits of the policy, because the averting expenditure method does not capture the value of the disutility (discomfort) associated with any sickness caused by drinking polluted water or by the noise.

With the cost−of−illness method, the analysts simply totals the medical expenses associated with mitigating symptoms, plus the loss of income due to missing work (or school days missed, in the case of health effects on children). Again, this approach, which is widely used in policy analyses, misses the value of the disutility due to the illness, and should thus be regarded as a lower bound on the benefits associated with removing the causes of the illnesses.

D. Which Method Should One Choose?

The choice between the various methods available depend on a number of factors, including the type of benefit being valued, data availability, whether housing or labor markets satisfy the assumptions needed for hedonics, the need to focus on certain categories of beneficiaries and benefits, and the amount of funding available to perform the analysis. These considerations are reflected in table 1 and made explicit in table 3.

Table 3. Non−market valuation methods.

Method	*Suitable for…*	*Type of values*	*Conditions*
Stated−preference approaches			
Contingent valuation	Virtually any public policy or program; extremely flexible	Use values, Non−use values	Design and administration of the questionnaire are difficult, a number of biases are possible that can limited through careful construction and pretesting of the survey instrument.
Revealed−preference approaches			
Travel cost methods	Only for amenities, natural resources (e.g., beaches, bodies of water, national parks or wildlife reserves) or cultural sites (monuments) that people actively visit.	Use values	Travel cost can be subject to measurement error, especially if the researcher wishes to include the opportunity cost of time. It may be difficult to identify substitute sites. Questions about trips taken under hypothetical conditions may be necessary to trace out the demand function at post−policy conditions.
Hedonic pricing methods	Only for changes in environmental or urban quality that can be captured into housing markets; only for job risks that are captured into compensating wage differentials.	In theory, both use and non−use	Individuals are assumed to be perfectly aware of the environmental, urban quality, job risks. Market must clear. Sufficient transactions must be observed to estimate the hedonic regression, and sufficient variability in environmental or urban quality or job risks must exist to

			identify their effect. Difficult to separate the effect of these variables from other factors that can influence housing prices or wages.
Averting expenditures	Human health effects or other effects (e.g., materials damage) from which people can protect themselves	n/a	Possible when individuals can document actions and expenditures incurred to reduce risks. In some cases, it is possible to engage in actions that reduce risks (e.g., staying indoors in days with high air pollution) but it is not easy to place a monetary value on these actions. Fails to capture the value of the discomfort of being sick.
Cost of illness	Human health effects	n/a	Relatively easy to perform, but fails to capture the value of the discomfort of being sick.

Based on our discussion and as shown in table 3, one could use contingent valuation or the travel cost method to value, for example, the benefits of a policy that improves the quality of a body of water, and hence improves the safety of swimming, boating, and fishing in it, or enhances its properties as wildlife habitat. While contingent valuation can reach anyone, including people that do not currently visit the body of water or will after the policy, the travel cost method can only target users, and will thus miss non−use values. By contrast, if one wishes to value the human health effects of a transportation or emissions policy in an urban setting, one may resort to contingent valuation, to the cost−of−illness or averting expenditures methods, or to the hedonic pricing method, but would not use the travel cost method.

Different valuation methods need not be mutually exclusive. In some cases, different methods can be used to address the same question. This allows a comparison and validation of the results obtained with different methods.[10] In other cases, different methods may be complementary, which implies that results obtained with different methods can be combined. For example, non−use values, which can only be measured through contingent valuation studies, could be combined with use values for a resource or amenity, which can be estimated using the travel cost method.

Even when using revealed preference methods, it is sometimes possible to increase the amount of information available to the researcher by using well−crafted hypothetical questions. For example, the researcher's ability to trace out recreational demand may be improved by asking a hypothetical question about how many trips the recreationist would take if the price was X, or whether or not the recreationist would continue to go or stop going to the site altogether if the price per trip was Y, where Y is higher than the current price.

We feel that it is important to add a word of caution if non−market valuation methods are to be used for ecosystem valuation. Non−market methods are suitable for valuing incremental changes in the ecosystem (e.g., increasing the populations of one or more endangered species) but not for valuing the ecosystem is its entirety.[11]

Finally, since some of the valuation methods can be costly and time−consuming to implement, the value of the information obtained should be weighed against the cost of carrying out a valuation study. In this context, if sufficient data exist from other studies, the use of benefit transfer may represent an option of arriving at (tentative) results at a lower cost (see below).

3.4 Benefit Transfer

Conducting a valuation study is generally expensive and time−consuming. In some cases, it is possible to answer policy questions without necessarily conducting an original valuation study. The benefit transfer approach relies on existing information from previous study, and seeks to apply it to a new context or locale.

The benefit transfer approach has the potential to save analysts the time and resources needed to develop benefit estimates of the proposed policy. Given the demands of the regulatory process, these considerations may be extremely important. Moreover, the analysts engaging in a benefit transfer are able to gauge the quality of existing studies prior to conducting the benefit exercise.

However, benefit transfer is not without drawbacks. First, estimates derived using benefit transfer techniques are unlikely to be as accurate as primary research tailored specifically to the new policy case. Of concern to the analyst is whether more accurate benefit information makes a difference in the decision−making process. There are many situations in which a benefit transfer may provide adequate information. For example, if the entire range of benefits estimates falls well above or below the costs of the policy being considered, more accurate benefit estimates will probably not answer the efficiency conclusions. Other factors to consider to consider when deciding whether to conduct a benefit transfer include the availability of relevant, high−quality existing studies and the degree to which additional primary research would reduce the uncertainty of the current benefit estimates.

In practice, the following steps are usually required when conducting benefit transfer:

a. describe the policy case, identifying possible impacts and the population of beneficiaries;
b. identify existing studies that are relevant to the benefit transfer by conducting a literature search. The literature surveyed may include articles published in peer−reviewed journals, unpublished research and government reports;[12]
c. review available studies for quality and applicability. In other words, the analyst must determine whether the basic commodity being valued is essentially equivalent, the baseline and the extent of the change should be similar, and the affected population is similar. Differences and possible limitations implicit in the benefit transfer should also be identified.
d. Transfer the estimates, and address uncertainty.

In practice, there are three major ways to conduct a benefit transfer. The first is simply to transfer the benefits, without any adjustment, from locale A to locale B.

The second approach is to adjust WTP for income. For example, if mean WTP by a household for a specified change in environmental quality or a public good in country A is €50, and country A’s household income is €20,000, while country B’s income is €30,000, one could multiply €50 by 1.5. This relatively naïve approach presumes that the income elasticity of WTP is 1. Alternatively, one could rely on an estimate of the elasticity of WTP with respect to income (say, 0.7), and could adjust the transferred WTP accordingly. In this example, country B’s WTP would be $50´(30{,}000/20{,}000)^{0.7}$, or about €66. Finally, one could transfer the WTP function. To implement the latter approach, one would run a regression based on the data collected in country A relating WTP to individual characteristics and to characteristics of the good being valued, and construct a predicted WTP for country B based on the individual characteristics and characteristics of the good being valued appropriate for the latter.[13]

International comparison within Europe (Ready et al., 2004, Alberini et al., 2004) and the US and Taiwan (Alberini et al., 1997) have been performed in the area of WTP to avoid morbidity effects and reduce mortality risks. Results were mixed, in the sense that in some studies transfers based on the WTP function perform reasonably, while in others they performed poorly. In the latter, country−specific effects—due perhaps to differences in sampling, or differences in hard−to−account for factors—weigh heavily into WTP.

Another example of a benefit transfer occurs when values for one outcome or context are applied to another. For example, both in the US and the UK, government agencies have used estimates of the Value of a Statistical Life (VSL) estimated from compensating wages studies, and have applied them to arrive at the mortality benefits of policies that save lives (environmental or traffic safety programs). This has been done even though environmental mortality risks are very different than the risk of dying in a workplace accident over the next year, and the beneficiaries of environmental policies—the elderly and people with certain health conditions—are very different than workers in their prime age.

4 Review

4.1 Evaluation results

Not described yet.

4.2 Experiences

CBAs can be conducted ex ante—before the policy is implemented—or ex post. The US Environmental Protection Agency recently (1998, 1999) conducted retrospective CBAs of the 1977 Clean Air Act and of the 1990 Clean Air Act Amendments. These analyses concluded that the benefits due to the premature or cancer−related deaths avoided by this environmental statute accounted for over 80% of the total benefits.

We are aware of several pieces of legislation that require the agency in charge to balance the cost and the benefits in setting standards or issuing regulations. In the US, the Federal Insecticide, Fungicide and Rodenticide Act, first passed in 1947 and extensively amended in 1972, when responsibility for its implementation was transferred from the US Department to Agriculture to the US Environmental Protection Agency, requires that pesticides be registered with the EPA before they can manufactured for commercial use. Distribution of any pesticide that is not registered or that is improperly labeled is prohibited. The burden of proof about the safety of a new pesticide is placed on the manufacturer, while the EPA is responsible for gathering evidence from toxicological studies if it wants to reconsider the registration of an existing pesticide.

In determining whether to register a pesticide, the Congress required EPA to consider economic, social, and environmental costs and benefits of use of the pesticide. Information about the benefits and risks of pesticide use are collected and documented within the so−called Records of Decision.[14]

By contrast, in the Clean Air Act the EPA is instructed by the law to set national ambient air quality standards that ensure the protection of human health, with a wide margin of safety and without considering the costs of attaining such protection.

Within the EU, the Water Framework Directive contains provisions that may be interpreted to imply that the benefits of the policy be balanced with its costs. Specifically, the Directive requires that all bodies of water be brought to ‘good ecological status’ but allows for derogations to the rule if the costs are disproportionately high.

CBA have been carried out for a number of proposed or actual EU policies. For example, DG−Environmental has conducted a CBA of lowering the sulfur contents of gasoline and diesel fuels to less than 10ppm.[15] The benefits of the policy include, among others, reduced consumption of fuel for vehicle of more recent vintages, and reduced mortality and morbidity effects associated with air pollution exposures.[16] Under the “main scenario 2005," the present value (at a discount rate of 4%) of the net benefits is equal to over €2,673.5 million.

4.3 Combinations

Not described yet.

4.4 Strengths and weaknesses of CBA

The appeal of CBA is that by monetizing the benefits of the policy, it is possible to compare and/or aggregate many different categories of benefits with one another, and with the costs of the policy.

One major criticism that has been brought to CBA is that it assumes that those persons who stand to gain from the policy change compensate the losers. This ignores distributional issues, and the fact that, for example, a wealthy person would be able, and therefore willing, to pay more than a poor person for the same improvement in environmental quality, even though both cared about it with equal intensity. Equity considerations are not sufficiently reflected in CBAs. The argument that projects or policies with the best benefit−cost−ratio are socially desirable rests on the assumptions that the gainers can – in principle – compensate the losers of a project/policy and still be better off. Whether such compensation actually takes place is not part of the information provided by a CBA.

Regarding the fact the individuals with lower incomes may be limited in their ability to pay for the implementation of the program, economists would suggest that the benefit−cost calculus be amended to alter the weights given to the benefits of certain categories of beneficiaries or the burden imposed on certain categories of individuals.

CBA can be expensive and time−consuming, and results are likely to be sensitive to the many assumptions often required to complete the estimation of the benefits and the costs of the proposed policy and program (Moore, 1995).

Discounting

Another difficulty of CBA—and a key factor when performing a CBA within the broader goal of assessing sustainability policies—lies in the fact that many public programs produce streams of costs and benefits over time, rather than in one−shot increments. This requires discounting future benefits and costs to present values. There is considerable disagreement among economists about the interest rate (or rates) at which these future costs and benefits should be discounted, and about whether the choice of interest rates should reflect uncertainty about the costs and benefits of the program. In its impact assessments for planned legislation, the European Commission routinely applies a discount rate of 4%.[17]

Briefly, the present value of a sum, X, to be incurred in the future—T years from now—is:

(3) PV= $$X \cdot \left[\frac{1}{1+\delta}\right]^{T} \approx X \cdot e^{-\delta T}$$.

The quantity d is equation (1) is the *discount rate*, whereas [illegible] is the *discount factor*. If the sum X (which might be the cost or the benefit of a policy or project) is incurred every year for the next T years, the present value of the stream of payments or benefits is:

(4) PV= [illegible] .

Equation (2) can be generalized to the situation where the payments or benefits are different every year:

(5) PV= [illegible] .

The problem with the choice of the discount rate d is further complicated by the fact that present values are very sensitive to the choice of the interest rate. For example, €1 million in benefits incurred fifty years from now has a present value of only €8,500 when an interest rate of 10 percent is used, €87000 when the interest rate is 5 percent, and €371,500 when the interest rate is 2 percent. This is of concern when CBA is applied to the evaluation of policies with significant intergenerational effects, such as those pertaining to the prevention of global climate change or the disposal of high−level radioactive wastes (which will be lethal for hundreds of thousands of years).

It is generally recognized that the social discount rate need not be equal to the private discount rate. Economists, however, argue that the discount rate should be positive, even in the context of intergenerational decisionmaking and investments. Treating one euro today as if it were worth as one euro several years from now is counterintuitive, and would result in an excessive sacrifice in current wellbeing, to the point that each successive generation would be impoverished in order to further the wellbeing of the next (OECD, 2004).

The notion of social discount rate is usually presented in the form of the following equation, known as the Ramsey equation:

(6) [illegible],

where d is the social discount rate, r is the ‘pure’ rate of time preference, which reflects people’s impatience,[18] g is the growth rate of future per capita consumption, and m is the elasticity of the marginal utility of consumption. The quantity m is therefore the percentage change in welfare derived from a percentage change in consumption (or income), and its intuitive interpretation is that it expresses individuals’ aversion to fluctuations in income levels. Because people expect to be richer, which means that the marginal utility of a euro in the future is lower than the marginal utility of a euro now, [illegible] measures the portion of the discount rate due to the fact that future people will be richer than people today.

What are, then, reasonable values for the social discount rate d? Recent literature reviews suggest that m should be equal to one (Cowell and Gardiner, 1999), and that the pure rate of time preference should be at most 0.5% (Pearce and Ulph, 1999). This implies that for an economy growing at, for example, 2% per annum, the social discount rate should about 2.5%. Indeed, social discount rates developed using equation (4) generally range between 0.5% and 3% (US EPA, 2000).

Is the social discount rate constant over time? Gollier (2002) examines how equation (4) should be amended when there is uncertainty about future income. Precautionary saving—the *prudence effect*—lowers the social discount rate. By contrast, the social discount rate increases with m. The net effect on the social discount rate, therefore, depends on which of these two factors is stronger. Expectations about the rate of growth of the economy over time are an important determinant as to whether the social discount rate stays the same or changes over time. Even if people expect the economy to continue growing at the same rate across time periods, the social discount rate may be declining over time if people exhibit decreasing relative risk aversion to risk as wealth increases. Weitzman (1998, 1999) discusses another rationale for time−declining discount rates, which are driven by uncertainty about future discount rates. As time goes by, the discount rate converges to the lowest possible discount rate.

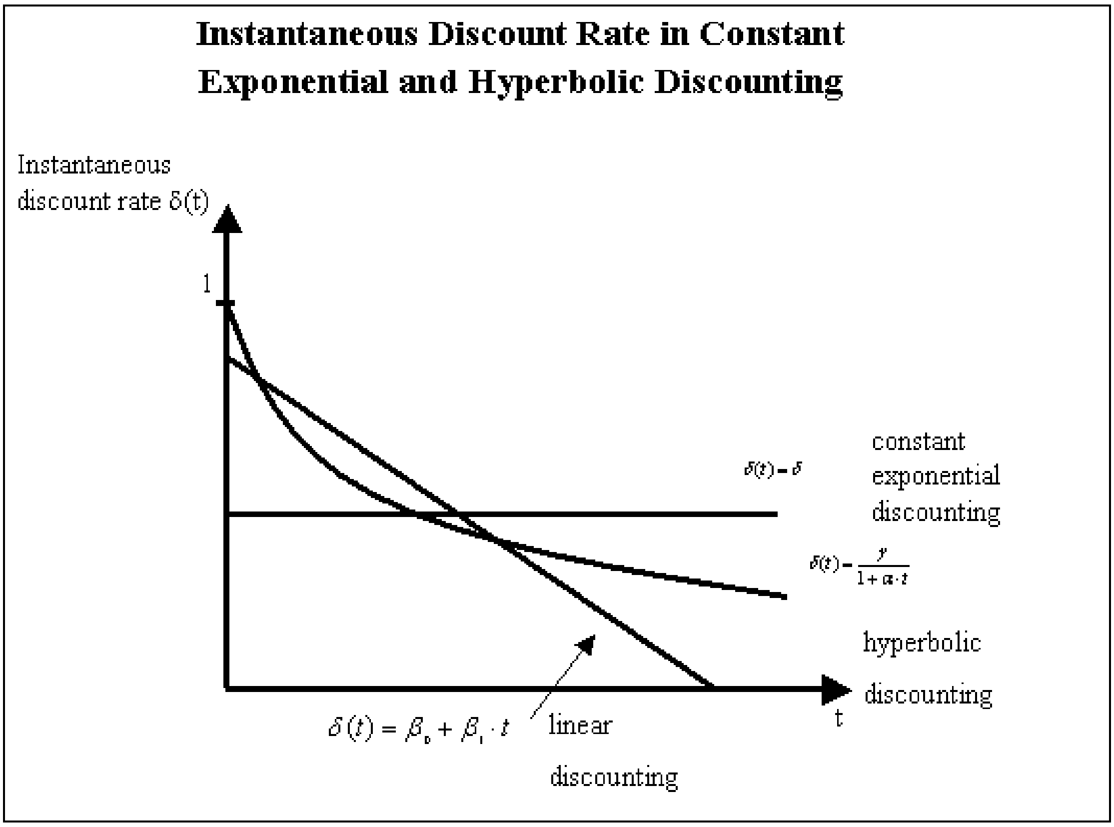
Chichilnisky (1997) considers current−day decisionmakers whose objective is to maximize the discounted net benefits under a sustainability requirement (effectively, future generations’ well−being). A declining discount rate over time is consistent with the requirement that current generations must always take into account the wellbeing of future generations.

Li and Lofgren (2000) obtain social discount rates that decline over time if society is comprised of two individuals, a conservationist and a utilitarian, where the former has a lower discount rate than the latter. Depending on the weights attached to the conservationist and the utilitarian individuals in final decisionmaking, the discount rate may well decline over time, tending to lowest possible discount rate (that of the conservationist).

Weitzman (2001) surveys 2600 professional economists, asking them “what real interest rate […] should be used to discount over time the (expected) benefits and (expected) costs of projects being proposed to mitigate the possible effects of global climate change.” The survey data indicate that over the next 1−5 years, the average discount rate is 4 percent; for the 6−25 year horizon, 3 percent; for 26−75 years, 2 percent; for 76 to 300 years, 1 percent, and for more than 300 years, 0 percent.

Empirical evidence based on individual behaviours and data does support the notion that people’s private discount rates may be declining, rather than staying constant, over time. It is sometimes suggested that individuals apply hyperbolic—rather than constant exponential—discounting. Under hyperbolic discounting, the discount factor is higher than that implied by constant exponential discounting for sums of money that are incurred in the near future, but lower when the sum X is incurred at a later time. Figure 2 below summarizes this concept.

Figure 2.



Assuming exponential discounting with discount rates that are constant over time, researchers have inferred individual discount rates from behaviours or hypothetical choice questions. Interestingly, the discount rates exhibited by individuals vary considerably across contexts, e.g., when making money−versus−money tradeoffs, risk−versus−money tradeoffs, and employment decisions. For example, by observing the acceptance or rejection of retirement packages in the military, Warner and Pleeter (2001) estimate that discount rates range between 35 and 54% among enlisted personnel, and 10 to 19% among officers. Harrison et al. (2002) estimate the discount rates implicit in the choice to accept money now or later to be 28% for a sample of Danes participating in a laboratory experiment. Moore and Viscusi (1990) estimate the discount rates of workers facing mortality risks associated with workplace exposures, while Horowitz (1990) administers a questionnaire involving hypothetical tradeoffs between immediate and future risks, and Alberini et al. (2004) ask respondents to report their willingness to pay for immediate and future mortality risk reductions. In the money v. mortality risk tradeoff context, the discount rates range between 2 and 14%.

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