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VALUING THE ENVIRONMENT

Euston Quah and Tsiat Siong Tan

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Euston Quah is an Albert Winsemius Professor at Nanyang Technological University and Visiting Hengyi Professor at Zhejiang University. Tsiat Siong Tan is a PhD researcher at Making Energy Demand More Sustainable and Resilient at Future Resilient Systems in Singapore.

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Please contact the authors for information about this paper.

Email: ecsquahe@e.ntu.edu.sg, tstan1@e.ntu.edu.sg

Asian Development Bank Institute
Kasumigaseki Building, 8th Floor
3-2-5 Kasumigaseki, Chiyoda-ku
Tokyo 100-6008, Japan

Tel: +81-3-3593-5500
Fax: +81-3-3593-5571
URL: www.adbi.org
E-mail: info@adbi.org

Abstract

To determine the amount of environmental goods and services to provide or preserve, it is necessary to weigh society's degree of preference for the environment vis-à-vis other goods and services. This value measure serves to inform the decision-making and policy-making process and thereby justify the allocation of limited resources between competing uses. This paper provides an overview of economic valuation methods for environmental goods, with examples of real-world applications. Valuation methods include both demand and non-demand curve approaches, including the dose-response method, contingent valuation method, and hedonic pricing. The paper further discusses the damage schedules approach and benefits transfer in cases where conventional valuation methods are less suitable.

Keywords: economic valuation, environmental goods, non-demand curve approaches, demand curve approaches, pairwise comparison approach, benefits transfer

JEL Classification: Q51, D61, H43

Contents

1. THE NEED TO VALUE THE ENVIRONMENT

Unlike most goods and services, environmental goods and services are not traded in conventional markets due to their public goods characteristics (non-excludable and/or non-rivalry), as well as the fact that they often exist in the form of externalities, beyond the purview of producers and consumers. As such, the environment, which cannot be directly priced by the market, is regarded as non-pecuniary and intangible. Moreover, given that in the absence of intervention no price tag is attached to the environment, market failure ensues, and the environment is inevitably over-consumed and exploited.

While not explicitly priced, intuition points to the fact that people do value the environment. We do prefer to breathe cleaner air, have access to unpolluted water, live in a predictable global climate, or simply enjoy a scenic view of natural landscapes. Yet, it is also apparent that we are not willing to sacrifice everything else for a better natural environment. Accordingly, it is established that the environment has *some* positive value, but it is not of infinite worth, or "priceless," as some proponents of the environment might suggest.

To hence determine the amount of environmental goods and services to provide or preserve, the question lies in weighing society's degree of preference for the environment vis-à-vis other goods and services. When constrained by scarce resources, trade-offs must be made. Should we allocate additional resources to improve the environment, or invest in economic growth? The underlying objective when making this decision is to ensure that everyone is as satisfied as possible, or to make as many people better off as possible. In other words, the choices made should ideally maximize society's well-being. This notion of welfare maximization raises its own set of ethical and philosophical questions, warranting an extensive debate outside the scope of this paper.

Achieving socially optimal levels of environment and economic efficiency first requires the measurement of preferences for the environment, before these can be compared with preferences for alternative goods and services, and whether an informed trade-off can be made may be ascertained thereafter. In this endeavor, money may be used to measure unpriced preferences, as unappealing or even unacceptable as this might sound to some. Valuing the environment in monetary terms allows us to conveniently make a quantifiable comparison between environmental goods and market goods. For example, if a preservation project yields a million dollars in environmental and spillover benefits, at a cost of less than a million that could otherwise be spent on other market goods, the only logical conclusion would be that this project is desirable for society at large. A distinction can be drawn between the monetary values *of* and the monetary values generated *from* environmental goods and services, as will become clearer in subsequent parts of this paper.

The most obvious alternative to monetary valuation of the environment would be to conduct a referendum whereby each individual makes one vote in accordance with his or her unobserved preferences. The main shortcoming of such a democratic majority vote, relative to using money as a measuring rod, is that the intensity of preferences is not captured. In the one-man-one-vote referendum, an individual with a high degree of concern for an environmental objective is no different from another individual who cares for the environmental objective but only to a small extent. Monetary values are, however, able to encompass this vital piece of information.

The need for a value measure is primarily to inform the decision-making and policy-making process, thereby justifying the allocation of limited resources between competing uses, whether residential, commercial, social, environmental or otherwise. Putting a value on the otherwise intangible environment at the very least advocates responsibility and precaution when striving for economic development and growth. This importance has been amplified as we increasingly acknowledge the severe impacts of environmental degradation. Quantifying the stock and flow of natural capital, in addition to physical and human capital, allows us to carefully chart our path toward sustainable development. It empowers us to better address environmental problems through policy and management measures (e.g., green taxes), conduct green national accounting, and design compensation schemes in cases of pollution incidents, thus correcting the market failure. Environmental valuation also plays a traditional role in policy impact assessments, and has conventionally been incorporated into cost-benefit analysis, cost-effectiveness analysis, environmental impact assessment, risk-benefit analysis and so forth. Conducted ex-ante, valuation is useful in stipulating and warning of potential losses and required compensation before the damage has occurred. Conversely, ex-post valuation advises future policy making.

2. TOTAL ECONOMIC VALUE

Total economic value broadly constitutes use value and non-use value. Use value can itself be subdivided into direct use value, indirect use value, and option value, while non-use value comprises existence value and bequest value. When valuing the environment, one must therefore first clearly discern if the total economic value is elicited, or that only one or some components of the total economic value is/are captured. Note that in addition to the typology of value illustrated below, there are alternative definitions and classifications of total economic value, such as drawing a distinction between anthropocentric and non-anthropocentric values, intrinsic and instrumental values (Hargrove 1992), which will not be discussed here.

Direct use value has the most straightforward interpretation: the benefits derived from the actual use of environmental goods and services. Examples include increased tourism revenue as a result of the preservation of natural parks, and revenue gains from fishing by improving water quality in fishing grounds. Indirect use value refers to the benefits from ecosystem functions, such as the regulation of the climate, or the environment's role as a waste sink. Lastly, option value (Bishop 1982; Smith 1987) is the potential benefit through options for use in some future time period. Individuals might prefer to preserve a specific plant species that has no current use, as he or she believes that it could potentially provide remedies for ailments in the future, subject to some probability of scientific discoveries.

Non-use values are conceptually more abstract, and they are also more difficult to measure empirically. Existence value (Walsh, Loomis, and Gillman 1984) is the satisfaction derived from knowing that a certain environmental good or service exists. An individual might never have or intend to ever meet a Bornean Orangutan. Nevertheless, he/she still hopes to ensure the species' existence, given the information that they are critically endangered. The individual could be driven by some moral satisfaction of knowing that the animal does not become extinct, or feeling that he/she has fulfilled an innate responsibility to protect wildlife and the natural environment, and as such the society is ameliorated in some way. Bequest value, on the other hand, refers to the utility derived from passing down environmental benefits to future generations, thus ensuring that they will enjoy access to environmental goods and services. This can be construed as a form of intergenerational altruism. In both existence value and bequest value, the individual attributing his/her value does not use the environmental good in question. In fact, bequest value is a form of existence value.

3. METHODS OF VALUATION

The environment can be valued mainly using non-demand curve approaches (Section 3.1) or demand curve approaches (Section 3.2). With non-demand curve approaches, there is no true welfare measure; the willingness-to-pay for (or willingness-to-accept) an environment good or service cannot be determined without obtaining a demand curve and the area beneath it. While not ideal, non-demand curve approaches nevertheless provide useful information for policy making, and will be briefly discussed in the following section. Non-demand curve approaches include replacement cost, mitigation behavior, opportunity cost, and dose-response methods. In contrast, demand curve approaches, as the name suggests, provide welfare measures when environmental improvement or degradation occurs. Demand curve approaches can be further classified into stated (or expressed) preference methods, and revealed preference methods.

Stated preference methods measure welfare using income compensated Hicksian demand curves. Through the use of survey questionnaires, respondents explicitly place values on environmental assets by stating their willingness-to-pay (WTP) or willingness-to-accept (WTA). The most commonly stated preference method is the contingent valuation method (CVM). Revealed preference methods elicit consumer surplus welfare measures through uncompensated Marshallian demand curves. Revealed preference methods, unlike stated preference methods, rely on market-based transactions. Environmental preferences are revealed through individuals' purchase of market goods that are related to the consumption of environmental goods. The value of unpriced environmental attributes can then be measured using methods such as the travel cost method or the hedonic pricing method.

Assigning monetary values (also referred to as shadow prices) to environmental goods and services fundamentally involves measuring preferences and utilities through real or hypothetical exchange transactions. This paper will also discuss two other approaches (Section 3.3), namely the Pairwise comparison approach and benefits transfers, which differ from the usual demand and non-demand curve approaches, and might prove particularly useful in developing countries. In the following sections, the conduct and applications of each valuation method will be detailed. It will become apparent that each method has its advantages and limitations, and choosing the one best suited for use is a judgment based on the circumstances and policy objectives at hand. Adopting the most appropriate method of estimation is crucial, with the goal that the estimated value should be as close to the true values as possible.

3.1 Non-Demand Curve Approaches

3.1.1 Replacement Cost, Mitigation Behavior, and Opportunity Cost Method

In deciding whether a project that results in the loss of environmental assets is justified, a blunt tool could be a measure of the cost of replacing, restoring, or maintaining these environmental assets by some other means. This is also known as the replacement cost method, which uses replacement costs as a proxy for the benefit of recovering environmental assets. For example, instead of measuring the widespread impacts of a potential oil spill, this method looks at the cost of clean-up as an indicator of the cost of the oil spill. While replacement costs are theoretically distinct from the benefits of replacement, the method can be useful as a benchmark when environmental assets are deemed too difficult or costly to directly value. This method is also particularly relevant when circumstances dictate that the quality of environment has to be maintained, or environmental preservation must occur for some reason. Some case studies incorporate the replacement cost method to estimate the value of aquatic species and sites of considerable importance to indigenous people in Australia (Jackson, Finn, and Scheepers 2014), and to assess the cost of soil erosion in the upper Mabaweli watershed of Sri Lanka and the Nyaung Shwe Township in Myanmar (Gunatilake and Vieth 2000; San and Rapera 2010).

The mitigation behavior method is similar to the replacement cost method, except that it uses expenditures incurred to avert the effects of lower environmental quality as a yardstick for policy decision making. For example, in the case of air pollution, one might deem expenditures on air purifiers and face masks as mitigation behaviors. In a case where the building of a dam hinders the migration of salmon to their hatchery sites, the cost of building artificial fish ladders to mitigate these impacts and allow the fish to swim upstream to breed could be used.

The opportunity cost method also provides no direct valuation of environmental goods and services, and uses the opportunity cost of a proposed project that degrades the environment. The opportunity costs of a project or policy that is environmentally detrimental are then compared with the necessary benefits to render it efficient. Concomitantly, the social opportunity cost of a given project represents the foregone social gains derived from the next best alternative use.

3.1.2 Dose Response Method and Value of Statistical Life

The dose response method establishes the relationship between a human (or more precisely any living being) *response* and increased *exposure* to environmental stress, such as an additional *dose* of pollution. The method has been widely applied to study the impacts of air pollution, with Quah and Tay (2003) providing a good illustration.

Quah and Tay (2003) applied the dose response method to study the economic cost of particulate air pollution on health in Singapore. The study used PM10 (particulate matter with an aerodynamic diameter of 10 micrometers or less) as an indicator of the health risk of air pollution. This pollutant category has previously been identified as responsible for most health problems arising from air pollution, including respiratory symptoms and diseases, carcinogenesis and premature death. Data on the ambient concentration of PM10 were first collected. Air pollution epidemiological literature and further empirical studies were then used to draw connections between pollutant emissions (using PM10 as a proxy) and human health effects, with a dose response function formulated as a result. The dose response function can be represented using the following equation (Ostro 1994; Rowe, Chestnut, and Lang 1995; Quah and Tay 2003):

$$dH_{ij} = a_{ij} \times dA_j \times POP_i$$

where dH_{ij}: Change in risk of health impact *i* due to pollutant *j*

aij: Slope of the dose response function for health impact *i* due to pollutant *j*

POP_i: Population at risk of health impact *i*

dA_j: Change in ambient concentration of air pollutant *j*

The coefficient in the dose response function will likely vary across existing medical studies, and if the policy maker were to draw from the literature, the use of a set of coefficients is recommended, i.e., a lower, central, and higher coefficient. The range of coefficients to be applied could be set at one estimated standard deviation apart.

Next, the impacts on morbidity and premature mortality can be expressed as below:

$$\Delta Morbidity = b_i \times \Delta PM10 \times POP_i$$

$$\Delta Mortality = c_i \times \Delta PM10 \times POP_i \times 0.01 \times \text{crude mortality rate}$$

where b_i: Morbidity coefficient from the dose response function for health impact *i*

c_i: Mortality coefficient from the dose response function for health impact *i*

POP: Population exposed to risk of health impact *i*

With the estimated changes in morbidity and premature mortality, the final step requires the assignment of monetary values to these changes. This in turn requires a value to be placed on human lives, a rather contentious issue, especially from an ethical or philosophical perspective. Nevertheless, the objective here is simply to list and explain some economic methods to value human lives, which is useful not only to assess environmental impacts and health risks, but also in related applications, such as determining tort compensations.

Economists assign monetary values to statistical lives instead of the specific lives of individuals. The value of statistical life (VoSL) involves aggregating a small change in fatal or non-fatal risk across a population. For instance, a 1% reduction in the risk of death that affects 100 individuals is equivalent to one statistical case. This one statistical life is not identified to a particular individual: it is not the intrinsic value of life, nor the benefit of saving a specific life with certainty, but the amount an average individual is willing to give up to reduce his or her morbidity or mortality risks. This value could for example be used to inform policy makers who are considering a project that on average saves one statistical life within the population.

One method to estimate the *value of statistical life* is to directly survey respondents and elicit their WTP for a reduced health risk, or WTA for the opposite, subject to their budget constraints in reality. If an average respondent is willing to pay $10,000 for a 1% reduction in his or her risk of death within the current time period, the VoSL can be computed as being $10,000/1% = $1 million. The same logic applies to non-fatal risks, which use the *value of statistical life year* (VSLY), or the value society places on reducing the risk of premature mortality or prolonging life. If the average respondent is willing to pay $10,000 when the expected gain in longevity from reduced pollution is one month, then it can be concluded that the WTP for one full year of life would be $10,000*(12/1) = $120,000. The conduct of such surveys is an application of the contingent valuation method, which will be elaborated in Section 3.2.2.

Another way to measure the inherent trade-off one makes between risk of death (or illness) with the consumption of other goods and services, is to look at the exchange one makes between income earnings and the risk of job-related death (or illness). To illustrate, a job with a 0.2% risk of death might warrant a wage premium of $1,000 over another job with a 0.1% risk of death, ceteris paribus. This implies that the employee has to be compensated $1,000 for taking on the additional 0.1% fatality risk. If this is representative of the average employee, and if labor markets are perfect, then the value of statistical life can be calculated as being $1,000/0.1% = $1 million. Viscusi and Aldy (2003) have provided a comprehensive review of this approach.

Alternatively, one could choose to use medical expenditures (direct costs) and illness-associated lost productivity (indirect costs) as a proxy for the value of life. This is also known as the cost of illness approach, which directly accounts for the human capital value of health, that is, the productivity returns. Furthermore, the money spent on healthcare could otherwise be spent on other goods and services, costs that are averted in good health. Dividing the cost of illness by mortality rate then gives the cost of life lost. The main shortcoming of this approach, as opposed to the two methods presented before, is that it yields an underestimation, as we omit the intrinsic value of life. In reality, we expect people to value life per se, as living life generates happiness and meaningful experiences. We also expect that patients would not completely return to their initial healthy state with medical treatments, and would be willing to pay even more to avoid pain and sufferings from illness, as well as to remain in a healthy state rather than be cured in an unhealthy state. Note that healthcare subsidies, which distort prices, or health insurance coverage, which leads to moral hazard problems, further complicate the measurement.

Other similar approaches to valuing a statistical life exist, such as observing revealed behaviors and WTP in insurance markets. One may also look at preventive expenditures instead of medical or insurance expenditures, such as price premiums for residential housing with lower rates of pollution, or WTP for helmets to reduce motorcyclists' injury risks. Discounted lifetime wages could also be indicative of a life's worth, at least in terms of the life's contribution to society from the provision of labor and human capital. Note that instead of using WTP to measure the trade-off between health-risk reductions and the consumption of other goods and services, one may instead measure the trade-off between different health states of varying durations (Hammitt 2002). This is the underlying principle of the non-monetary measures of disability-adjusted life years (DALY) and quality-adjusted life years (QALY), whereby weights are assigned to life years of discrepant health quality. For example, a year lived in illness might be worth half of one in good health. By extension, using the various valuation methods, DALY and QALY can be monetized as well (Lvovsky et al. 2000).

3.2 Demand Curve Approaches

3.2.1 Revealed Preference Approaches

3.2.1.1 Travel Cost Method

Revealed preference approaches rely on observable market-based transactions of goods that are related to the consumption of intangible environmental goods. The travel cost method uses travel costs as a proxy for price, which can be directly observed in monetary terms, to gauge the demand and hence the value of recreational sites, including natural parks and reserves. This method assumes that if an individual chooses to incur the cost of visiting a site, then this cost is at least the value the individual attributes to the site, including the environmental goods and services it provides. A relationship can thus be derived from the observed variations in travel costs and frequency of visits. As with the law of demand, a higher travel cost (price) will correspond with a lower number of visits (quantity demanded) (Clawson and Knetsch 1966; Duffield 1984; Randall 1993; Adamwowicz, Louviere, and Williams 1994).

Deriving the demand curve of a natural park then allows us to calculate the value visitors place on, say, the natural park by calculating the area under the demand curve. Note that travel costs primarily comprise fuel cost and travel time cost, the latter of which may be estimated using the average wage rate. However, if the journey in itself is an enjoyable activity to the individual, then travel time is not really an incurred cost, but an exchange of time for the joy of traveling. Including travel time cost in this case could lead to an overestimation of the site's value.

The researcher conducts on-site questionnaire surveys to collect information on the round-trip travel cost and number of visits to the amenity. Concentric circles are first drawn to define the different zones around the amenity used by people living at varying distances from the site. Each concentric zone corresponds to a different round-trip travel cost; outer zones have higher travel costs and vice versa. The number of visitors from each zone is then divided by the population in each zone to give the frequency of visits by each zone. The relationship between travel cost and frequency of visits can subsequently be estimated with an econometric model, using frequency of visits as the dependent variable and travel cost as the explanatory variable. The negative coefficient estimated tells us by how much the frequency of visits falls when travel cost rises. Thereafter, a hypothetical range of admission fees to the amenity is imposed. The hypothetical admission fees serve as a proxy for price and should be in a reasonable range as compared to similar sites. At each alternative admission fee, the schedule of total trips can be easily calculated with the regression coefficient presented above by adding the admission fee to the travel cost. With that, the relationship between the price of the amenity (the hypothetical admission fee) and the quantity demanded (the total number of visits from all zones), which is essentially the demand curve, can be derived. If the data are collected for all visitors within a single day, the area under this demand curve represents the daily value of the site, which can later be augmented to give the annual or even perpetual value of the site.

The steps described above represent the basic zonal travel cost method (Clawson and Knetsch 1966). This valuation approach is relatively inexpensive to apply, has potential for a large sample size, yields results that are relatively easy to interpret and explain, and is based on actual behavior as opposed to stated preference methods. There are also further methodological improvements to be considered. Besides the fuel cost and travel time cost, travel costs should also include accommodation spending, excess meal expenditure, purchases of goods exclusively for the purpose of the trip, and/or existing admission fees to the amenity (in this case, one would be using a hypothetical increase or decrease in admission fee in the steps described above) where appropriate. Instead of the zonal travel cost method, one could also adopt the individual travel cost method or the random utility travel cost method.

For the individual travel cost method, the researcher collects data on travel costs from individual visitors instead of assuming that visitors from the same zone are identical. Therefore, survey questions would include more details, such as the specific residence location of the visitor, the person's income, socioeconomic variables, demographic variables, other locations visited on the same trip, amount of time spent at the site, the level of satisfaction derived from the trip, perceptions of the environmental quality of the amenity, the number of visits per year, the availability of substitute sites that the respondent might have alternatively visited, and so on. This circumvents some of the limitations of the zonal travel cost method, such as assuming equal opportunity cost of time for people within the same zone, or that individuals all react to an increase in travel costs in the same way (i.e., same regression coefficient). If travel costs are incurred to not only visit the amenity but also for other purposes (e.g., a tourist visits multiple attractions on the same trip), then the travel cost incurred should not be the value for one amenity but rather apportioned across multiple attractions using the time spent on each.

The individual travel cost method yields higher empirical precision with a multi-regression model, but of course requires more extensive data collection and complicated statistical analysis. This is even more so for the random utility travel cost method. Based on the random utility theory, individuals choose a site to visit out of a set of possible amenities, and make trade-offs between the site's quality and the cost of traveling to maximize one's utility. This means that data are required from more than one site, accounting for site-specific characteristics such as environmental quality. The random utility travel cost method utilizes conditional probabilistic models to characterize demand, requires more assumptions, and is more complicated to operationalize.

It is also important to note that the travel cost method only calculates the value of the site to actual visitors who use it, and hence disregards non-use values. For environmental reserves with endangered species, one should be careful not to conclude that the findings from using the travel cost method is axiomatically the total economic value (see Section 2), as non-use values in this case are likely to be very large. A low frequency of visits (user rate) could also be an indication that the inclusion of non-use values might affect the results. People who live in close proximity to a park and thus incur almost zero travel costs thanks to a conscious house purchase decision or otherwise might also value the park more than it seems. Lastly, when differences in travel costs among visitors are not sufficiently large, this method would also prove inapplicable.

3.2.1.2 Hedonic Pricing Method

The second form of revealed preference approach is the hedonic pricing method, which again uses existing market information to determine the value of environmental goods and services. The basis of this method is that market prices (such as for real estate) are directly affected by their associated environmental characteristics. In purchasing a residential property, the consumer seeks to maximize his/her utility by making a trade-off between the price of the property and the benefits they can derive from it. These benefits not only include factors such as the property's proximity to one's workplace and local amenities, access to the transportation network, and its size and design, but also environmental benefits such as its landscape aesthetics and scenic views. Conversely, one would pay less for houses situated near to sources of pollution or other not-in-my-backyard facilities (NIMBYs) for the same reason (Quah and Tan 2002).

Knowing that different variables – although not individually priced – are implicitly included in real estate prices allows us to control for these variables. If we were to control for all variables except environmental characteristics, then any remaining price differential between real estate must be due to underlying uncontrolled differences in environmental characteristics (Rosen 1974; Freeman 1993; Smith 1993; Clark and Nieves 1994). Similarly, when controlling for environmental characteristics, the regression coefficients give the value that consumers implicitly place on these variables. Adoption of the hedonic pricing method has become increasingly feasible with technological advancements, enabling researchers to use geographical information systems (GIS) to digitally map locations and accurately link physical locations with access to amenities and other variations in desirable and undesirable environmental traits, such as varying exposure to noise pollution from highways. Jiao and Liu (2010) have utilized a geographic field model-based spatial hedonic pricing method, finding that apartments situated close to recreational spaces of the Changjiang River and the East

Lake in Wuhan in the People's Republic of China were significantly more expensive than counterparts that did not meet these criteria.

One major advantage of the hedonic pricing method is that it measures both use and non-use values. However, a prerequisite of the method is a well-functioning property market that readily reacts to changes in demand and supply. This is often not the case, as moving house is a long-term decision hindered by a variety of barriers, many of which cannot be readily observed, and the data are not necessarily collected easily. Income constraints are likely to restrict purchase (or rental) decisions of high-cost items like housing. In many states, housing prices are also highly distorted by government taxes and subsidies. Econometrically, not only is it difficult to identify all of the relevant explanatory variables that affect real estate prices, but problems of multicollinearity further confound the analysis. For example, the degree of air pollution is expected to be highly correlated to the visibility of scenic views. Finally, the findings are restricted only to (environmental) variables that are identified as varying across properties.

3.2.2 Stated Preference Approaches

3.2.2.1 Contingent Valuation Method

In the 1989 Exxon Valdez incident, the contingent valuation method (CVM) was validated by the United States government and judicial systems to assess the environmental damage of the oil spill, and its impacts on beaches, coasts, and wildlife habitats. The CVM has since become the most common and widely adopted environmental valuation method in the literature.

The economic principle of CVM can be represented as follows (Quah and Tan 1999):

$$U(Q^0, y^0) = U(Q^-, y^+) = U(Q^+, y^-)$$
$$= U(Q^-, y^0 + WTA) = U(Q^+, y^0 - WTP)$$

Where

$U(Q^0, y^0)$ is the current utility level without the hypothesized change

Q^0 is the initial quantity/quality level of environment

y^0 is the initial income level

– represents decrements

+ represents increments

WTA is willingness to accept

WTP is willingness to pay

As depicted above, given that utility is a function of the environment and income, ceteris paribus a decrement in environment quality/quantity can be offset by an increment in income, and vice versa. Changes in levels of welfare can be made possible with monetary payments. It is therefore possible to create a hypothetical market for intangible environmental goods and services by stipulating an environmental improvement (or degradation) that an individual would be willing to pay for (or willing to accept monetary compensation for) in an attempt to maximize his/her utility.

This stated preference approach requires directly surveying respondents and asking them to state their values for non-market environmental goods and services. To do so, the researcher first provides a detailed description of hypothetical scenarios involving changes in environmental quality or quantity (alternative states of the situation that differ from the status quo), which are the projected effects of a program or policy under consideration. If the environmental good or service were to be made available, an individual would be willing to pay an amount of money up to a certain point, which is his/her maximum WTP. The individual is not willing to pay beyond this amount to trade for the environmental good, and hence this is the true value assigned by the individual to the said good. The same argument follows for a removal of an environmental good or service, except that the minimum WTA is to be elicited in this case. With individuals' WTPs or WTAs, the average WTP or WTA can then be calculated and multiplied by the affected population to give the total value of the environmental asset. Deriving the total economic value (both use and non-use value) becomes possible, but whether the result is the total economic value or not ultimately depends on the design of the survey questions. For example, van Kooten and Bulte (2000) intentionally isolated the elicitation of existence values by only asking respondents their WTP to prevent the extinction of wildlife species. Meanwhile, Quah and Tan (1999) employed CVM to determine the total economic value (including use value, and option and existence non-use values) of landscape scenery, specifically the East Coast Park in Singapore. Use and non-use values are separated using CVM questions asking about the number of visits to the park, and respondents that seldom or never visit the park are deemed to be expressing their non-use values.

The value elicited using CVM is contingent on the given hypothetical scenario, as well as how each respondent interprets and assigns it a value. Therefore, it may prove difficult to elicit a true value, and to do so, it is crucial that the presented hypothetical scenario be realistic, precise (including information on the duration and scope of impact), and fully understood by the survey respondents. It might even be useful to use photographs or audio clips to describe the scenario to the respondents, although the researcher must be cognizant that this can prove counterproductive if it distorts the scenario or restricts respondents' understanding.

Designing the survey and how it is to be conducted is paramount to the success of CVM. There are various ways by which CVM questions can be asked. The most straightforward method is to use an open-ended question, i.e., "How much would you be willing to pay for/given that …?" This allows the respondent to state any amount he/she wants, which could result in a high variance in values if respondents in general have little or no idea as to the value they would attribute to the subject matter. The researcher can derive the average WTP or directly plot a downward-sloping demand curve, and calculate the area underneath using a regression or otherwise. Furthermore, control variables such as demographic characteristics might be added into a multiple regression model, and the same applies to the other methods described below. We could, for example, find significant differences in the average WTP of men versus women.

Open-ended questions are often not advised owing to their lack of statistical robustness. This has been mentioned as one of the guidelines of the experts panel, including Nobel Laureates Kenneth Arrow and Robert Solow as Chairmen, to the National Oceanic and Atmospheric Administration (NOAA) (Arrow et al. 1993). Another method is the referendum (take-it-or-leave-it) approach, in which each respondent receives one discrete choice question with one predetermined amount of monetary value, e.g., "Would you be willing to pay $x for …? Yes/No?" A range of predetermined amounts, usually at regular intervals, is given across respondents, such that some respondents are asked in terms of \$x, \$y or \$z. The collected data provide information regarding how many people are willing to pay at least the given values, and the lower-bound WTP can be estimated using various estimators, such as a Turnbull estimator (Turnbull 1976). To illustrate simply, if 20% of the respondents are willing to pay at least \$40, and 80% are willing to pay at least \$20, the average WTP at the lower bound works out to be 0.2*\$40 + 0.8*\$20 = \$24. Moreover, the data could be analyzed econometrically using probit/logit models and/or ordered probit/logit models. The demand curve (using survival analysis) can be derived as more people would be willing to make small amounts of payment relative to large amounts. The downside is that the sample size has to be larger. See Subade and Franciso (2014) for an application of the referendum approach to elicit the non-use values of the Tubbataha Reefs in the Philippines.

The payment card method is another way of conducting CVM studies. Instead of one predetermined amount per respondent as in the referendum approach, each respondent faces the entire range of predetermined values, i.e., “Would you be willing to pay the following amounts for …? Option 1: \$x. Option 2: \$y. Option 3: \$z.” Ordered probit/logit regressions are then used for analysis. As with the referendum method, or other methods that provide some form of predetermined values, it is vital to note that given values are arbitrary and should be justified with existing studies or theoretical support, or at the very least are intuitively acceptable. Moreover, one would expect that the payment card method has a tendency to yield results with WTP clustering at the lower amounts, and WTA clustering at the higher amounts. One may refer to Arin and Kramer (2002) for an application of the payment card method to value divers’ WTP to visit marine sanctuaries in the Philippines.

The fourth approach is in the form of auction bidding (Davis 1963; Cummings, Brookshire and Schulze 1986; Hanemann 1994). For WTP, the respondent is asked whether he/she is willing to pay \$x first. If yes, would he/she be willing to pay \$(x + ε). If yes again, would he/she be willing to pay \$(x + 2ε), and so on, until the respondent answers “no.” This switch point indicates the maximum WTP. For the minimum WTA, the same logic applies, with a downward bidding with successively lower prices instead of an upward bidding. This approach is more cumbersome to conduct and usually requires a face-to-face interview or an online survey, but the demand curve can be directly derived, and the results can be analyzed using a multiple regression model. Setting ε to be a reasonable amount also reduces survey fatigue, as a very small ε could result in a long survey process and repetitive questions. See Yu and Abler (2010) for an application of the auction bidding CVM to ascertain the WTP for unpolluted blue skies in Beijing, People’s Republic of China.

Another method proposed by Haneman, Loomis, and Barbara (1991) concerns the use of a double-bounded CVM. The respondent is asked if he/she is willing to pay \$x. If yes, would he/she be willing to pay \$2x? If no, would he/she be willing to pay \$x/2? Each respondent receives one starting price, but a range of starting prices (e.g., \$x, \$y, or \$z) is randomly assigned to different respondents. Again, the arbitrary predetermined starting values have to be properly set, but in this case, a larger range of values can be easily tested. We would then be able to collect information regarding the number of people lying within each range of value: whether bid < \$x/2, \$x/2 < bid < \$x, \$x < bid < \$2x, or bid > \$2x (and the same for \$y and \$z). Probabilistic and/or ordered probabilistic regressions are used for analysis.

For the payment card and double-bounded formats, follow-up questions directly asking for the maximum WTP or minimum WTA could always be included, as the exact WTP or WTA is not already elicited. Extreme bids and zero (protest) bids (Portney 1994; Kristrom 1997), which could be due to non-cooperative respondents or misunderstanding the survey, should be omitted from analysis. Refusing to pay for the environmental damages caused by others, or a perception that it is the government's responsibility to pay for and resolve the issue, are indicative of protest bids.

There have been extensive debates for and against the use of CVM. Arguments against CVM are largely due to either the hypothetical nature of the methodology, or respondents' innate behavioral biases. The former can be addressed or mitigated through the proper design of surveys and choice of research questions. We will therefore focus on the behavioral responses in CVM that could distort expressed preferences.

First, WTP and WTA measures are not equivalent. It has been empirically observed that WTA exceeds WTP. For example, Hanley (1989) found that the average WTA for a ban on the burning of straw is about four times as high as the average WTP against the same ban. Behavioral economists explain this discrepancy as loss aversion. People generally value losses more than gains. WTP, which measures the benefit of a welfare improvement (a gain), is therefore smaller than WTA, which measures the cost of a welfare deterioration (a loss). Hence, framing CVM questions as a loss or gain must be carefully considered. The choice of measure should depend on the current assignment of property rights. If the victims of pollution supposedly have the right from pollution, the researcher should be asking the affected general public or local residents their WTA as compensation for a lower environmental quality as a result of the proposed policy change. If instead the polluters have the right to pollute at status quo, CVM questions should ask for the victims' WTP to avoid a worsened environmental state due to the same policy change. One should also note that the public might be more familiar with and accepting of the concept of the polluter-pay principle (WTA for environmental losses) rather than the victim-pay principle (WTP to avoid environmental losses). Furthering the discussion, negative changes can also occur in the domain of gains, and therefore should not be treated as a loss, but rather as a reduction in gains. The choice of WTP and WTA thus becomes even trickier. For a detailed discussion on welfare measures concerning gain and loss domains, see Knetsch, Riyanto, and Zong (2015).

The way in which monetary payments are to be made hypothetically also matters. This is additionally known as the payment vehicle bias. For example, respondents might be willing to pay a smaller amount when it involves raising taxes, but willing to pay a larger amount when the money is contributed to a conservation fund for the same purpose. The solution to this discrepancy is to adopt the payment vehicle that is most realistic for the context. On a similar note, respondents may deceive the surveyor if they think that their responses might, for example, affect the actual tax rates they face in the future. This strategic non-revelation of preferences could be an attempt to free-ride on other people's payments, or to profit from compensation schemes. The researcher has to assess and make a judgment as to the severity of strategic bias, report any potential overestimation or underestimation, and adjust accordingly. One way to detect strategic bias is to check the distribution of elicited values using the Shapiro-Wilk W statistic (Brookshire et al. 1982; Cummings, Brookshire, and Schulze 1986; Carson and Mitchell 1993; Maddala 1997). Anchoring bias and embedding effects (Knetsch 1998; Chuenpagdee, Knetsch, and Brown 2001) also constitute a potential problem in CVM when the final WTP or WTA is centered on the starting bid provided, which respondents use as a mental reference point. Using a set of different starting bids and randomly assigning these starting bids to different respondents resolves this issue.

Finally, people might assign the same monetary value to a part of an environmental good (e.g., saving a specific wildlife species) and the entirety of the environmental good (e.g., preserving the entire forest with a variety of wildlife species, including the above-mentioned species). This irrational behavior, known as the part-whole bias, is explained by mental accounting. People tend to allocate a portion of their income or wealth for different categories of goods, such as a fixed budget for environmental goods and saving wildlife species. One way to mitigate this problem is to have respondents work out their overall budget for the environment before asking for their WTP (Turner and Adger 1995). Alternatively, one could use CVM to evaluate a basket of environmental goods rather than trying to value each environmental good individually, or conduct studies in both manners for comparison.

3.3 Other Approaches

3.3.1 Pairwise Comparison Approach

The conventional demand curve valuation methods detailed above have been highly debated as to their reliability, largely because they are centered on monetary assessments. At times when we might not be as confident of the final monetary values derived, the pairwise comparison approach or the damage schedules approach is preferred. This non-monetary method has received limited attention and is able to measure whether one environmental good is worth more than another. It values environmental assets in relative terms rather than absolute nominal terms. The approach aims to develop an interval ranking of relative importance for a set of intangible environmental issues and policies, derived from respondents' judgments of environmental degradation. Since it is only an indicator of relative social preferences, it does not face the problems of WTP and WTA non-equivalence and loss aversion (Champ and Loomis 1988; Knetsch 1990; Loomis et al. 1998).

Conducting a Pairwise Comparison

(Reproduced: Knetsch and Chuenpagdee 2002)

Step 1: Develop hypothetical, but realistic scenarios of different levels of damage to the resources of different levels of activities that can cause such damage.

Step 2: Use the paired comparison method to present these scenarios (generally as a questionnaire booklet).

Step 3: Conduct the survey, asking the respondents to complete the survey on their own.

Step 4: Analyze the data using the variance stable rank sum method.

Step 5: Test for any significant difference between the rankings of relative importance of resources obtained from various interest groups, and aggregate responses as appropriate.

Step 6: Suggest policy responses in accordance with the relative importance of the resources.

The above table describes the six main steps of conducting a pairwise comparison study. In each survey question, survey respondents are presented with pairs of environmental losses, alongside some hypothetical descriptions that invoke intrinsic feelings, and must choose the environmental loss they would prefer to suffer. The set of environmental losses are of different types and different levels of damage. The choices are to be randomized in order to control for order effects. If the choice set does not contain an excessive number of objects, n, all possible pairs can be presented to each respondent n(n-1)/2. Repeated measures for each element within the choice set yield more reliable estimates than single-point estimates as in CVM (Peterson and Brown 1998). However, one should acknowledge that the method is neither feasible nor satisfactory where there are a large number of objects (David 1998). The rationale behind using the pairwise comparison approach is that respondents are usually more comfortable in comparing objects pairwise and making a discrete choice, rather than stating a monetary value for intangible goods with which they might be unfamiliar. The method is also more intuitively appealing than an ordinal ranking of all objects, especially when differences between objects are subtle, and preferences might be conditional on the full set of alternatives that respondents see at one point in time.

Upon completing the survey, binary choices are aggregated across respondents to give an interval scale, an ordering of preferences amongst the elements within the choice set. This is achieved by giving a preference score for each item, which is the number of times the respondent prefers that item over other items. The preference score is aggregated across respondents and summarized by the variance stable rank method (Dunn-Rankin 1983), i.e., the proportion of times each item is chosen relative to the maximum number of times it is possible to be chosen. With that, we multiple this proportion by 100 and derive a collective judgment scale of the relative importance of all the items (Chuenpagdee, Knetsch, and Brown 2001) ranging from 0 to 100. Non-parametric statistical tests of significance are used to determine the degree of concordance among individual survey respondents and between different respondent groups. One example is the Kendall's coefficient of concordance, in which a statistically significant coefficient denotes consensus in ranking amongst the respondents. Correlation across subgroups can also be compared to uncover heterogeneity in preferences.

The pairwise comparison approach is primarily applied to ensure that policies correspond to public values and hence can be successfully implemented. Rutherford, Knetsch, and Brown (1998) used the pairwise comparison approach to uncover the expected damage from oil and toxic liquid spills of various types and magnitude. Quah, Choa, and Tan (2006) also used the method to compare four environmental problems perceived by Singaporeans as the most important environmental goods: degradation of the coastal and marine environment, polluted air, ozone depletion, and an unhygienic environment pertaining to food and water. In this study, two levels of environmental qualities, moderate and severe, were tested for each item.

If the findings from the pairwise comparison study show consistent choices, they indicate that decision makers are rational. Inconsistent choices are detected by circular triads, in which A is preferred to B, B is preferred to C, but C is preferred to A. This could be a result of intransitive preferences, random choice, or respondents' cognitive limitations. It is also possible that inconsistent choices are meaningful as a result of objects that are multidimensional with different characteristics or different levels (Kahneman, Ritov, and Schkade 1999). Respondents might perceive these choices as a close call and might be indifferent between the items. To avoid this problem, it is advisable to conduct a pilot survey and check the viability of the survey options. Alternatively, inconsistent choices may be repeated at the end of survey without explicit indication in order to ascertain the preference switches for inconsistent choices. If inconsistent responses are resolved during this retrial, the implication is that intransitive preferences are not the cause of the issue, but rather close calls and indifference between objects. Coefficients of concordance and correlation can be tested for their sensitivity to the inclusion or exclusion of intransitive observations.

The pairwise comparison method yields various advantages (Quah, Choa, and Tan 2006). This method is less costly in terms of time and money than other primary research methods and can deal with intractable valuation projects. The method is also flexible, as new scenarios of environment losses can be added by expanding the damage schedule through interpolation and extrapolation from the existing set of environmental goods measured. It provides an effective comparison of multiple values of environmental goods without multiple studies in a standardized manner, and the results are easy to interpret. The researcher can also choose to include monetary elements (sums of money) within a choice set to elicit monetary WTP or WTA. As long as some form of comparison in terms of the importance or severity of the objects provided can be made, a scaling can be derived (Sunstein 1994).

3.3.1 Benefits Transfer

Benefits transfer (Freeman 1984; Quah and Toh 2011) is another useful valuation approach that involves the adaptation and generalization of information from existing research to a different setting. Existing primary research and studies are referred to as study cases/sites, while the setting in which the information is adapted is termed the policy case/site. The policy site may differ from the study sites in terms of economic, biophysical, temporal and/or spatial situation (Freeman 2003; Wilson and Hoehn 2006).

Benefits transfer might be selected because it has been deemed that primary valuation is not warranted, primary valuation is too costly to conduct, there is a lack of expertise to conduct primary data collection, and/or there is immediate urgency to make a policy decision. To conduct a benefits transfer, a thorough literature review of relevant studies is crucial. Only with a sufficient number of studies would the adaptation of information be capable of yielding precise and robust estimates. Given that benefits transfer essentially draws from other valuation studies, it faces the same potential problem of measurement errors. In addition, it is subjected to transfer errors (errors when generalizing across different contexts), especially when adapting information to a setting that is notably different.

Benefits transfer can be categorized into two types: value transfer and function transfer. Value transfer involves a direct application of summary statistics from study cases to the policy case, making adjustments when necessary. The summary statistic could be WTP or WTA measures, or even demand elasticities. The adjustments to be made include a discrepancy in environmental impact between study and policy cases, a different affected population, currencies and inflation, and so on. Function transfer (Loomis 1992) involves the application of a statistical function rather than direct use of the summary statistic. Compared to the former, function transfer requires that more extensive adjustments be made through the statistical function to reflect the characteristics of the policy case, but yields more precise and robust estimates, as the differences in site characteristics are more effectively considered (Brouwer 2000).

Value transfer can be further separated into three types: transfer of point estimates, transfer of measures of central tendency, or transfer of administratively approved estimates. A transfer of point estimate typically uses a range of point values from various existing study cases. The shortcoming of this method is that the study sites and policy site should ideally be similar in terms of characteristics, including the geographical location, the baseline state of the environment, the degree of environmental change, the composition of the population, as well as other market, institutional and cultural characteristics. These assumptions are often not satisfied. Transfer of measures of central tendency uses the mean or medium of the estimates in, or the confidence interval of study cases. The decision to use the median over the mean is especially apt if study cases have outlier estimates that might skew the latter. Lastly, transfer of administratively approved estimates is the simplest approach in value transfer. However, these study cases' estimates have often (if not always) undergone the government's evaluation and approval, and so the process by which these estimates are endorsed and published might not be entirely objective. Below is a list of the steps to conduct a transfer of point estimates. The logic of conducting a transfer of measure of central tendency and transfer of administratively approved estimates is the same (refer to Rosenberger and Loomis 2003 for details).

Conducting a Point Estimate Transfer

(Source: Largely reproduced and adapted from Rosenberger and Loomis 2003; Smith 2014)

Step 1: Define the policy context.

This definition should include various characteristics of the policy site, what information is needed, and in what units. Policy site characteristics include the location, the type of environmental good, the availability of substitutes, the affected population (which may include both users and non-users and their socioeconomic status), and so forth. The degree, direction and timing of change in environmental assets must also be quantified, which first requires determination of the baseline state of the environment.

Step 2: Locate and gather original research outcomes. Conduct a thorough literature review, and obtain copies of potentially relevant publications.

The researcher should conduct a keyword search by country or region, type of environmental asset, valuation technique, year, etc.

Step 3: Screen the original research studies for relevance. How well does the original research context correspond to the policy context? Are the point estimates in the right units, or can they be adjusted to the right units? What is the quality of the original research?

The key is to maximize scientific soundness (the methodology and assumptions), relevance (the similarity in context), and richness in detail (the data description and information collected) (Desvousges, Johnson, and Banzhaf 1998).

Note that for intra-national benefits transfer, adjustments for inflation should be made. For international benefits transfers, adjustments for currency and income elasticity of WTP or WTA should be adjusted. If there is incomplete information on income elasticity, a sensitivity analysis of income elasticity between 0.5 and 2.0 could be considered. Otherwise, income elasticity could be assumed to be equal to one.

Step 4: Aggregation over environmental goods and services, affected population, and duration of impact.

This aggregation therefore gives us the valuation estimate in the policy case.

Function transfer can also be further classified into benefit function transfer and meta-analysis function transfer. Benefit function transfer is straightforward. It applies the regression coefficients from an existing benefit function (from a single study case) to the summary statistic of the policy case. The explanatory variables are the characteristics that affect the value estimate in both the study and policy sites, and this is a judgment to be made by the researcher. Using the same regression coefficient also implicitly assumes that both populations react in the same way toward the value of the environmental asset, which might not be empirically true (VandenBerg, Poe, and Powell 2001). On the other hand, the meta-analysis function transfer uses regression coefficients from multiple study sites. There is a clear advantage of not being restricted to one study site. The meta-analysis function transfer approach explicitly includes methodological explanatory variables such as the method of valuation specific to each study into the regression model, which allows for control of a large number of possible confounding variables. The number of studies to be included is however a trade-off between relevance and amount of information. Nelson and Kennedy (2009) provide a comprehensive review of over 140 meta-analyses involving the economic valuation of the environment.

4. CONCLUSION

Various tools and methods are used to value the environment, and these are continually being improved and developed. This paper has highlighted these valuation methods, but nonetheless it is not exhaustive. The choice of valuation method ultimately depends on the researcher's judgment: certain methods are more feasible and appropriate for certain study conditions, and no method is without its flaws. It is important to think through the motivation and design of the study, and to report the findings and their limitations transparently.

The importance of valuing the environment is compelling, as people become increasingly aware of the environmental challenges that we face today. Valuing the environment is even more important in developing countries that possess a large concentration of the Earth's natural resources and assets. For economic growth to converge with the developed world, developing countries must sustainably tackle the twin goals of rapid development and environmental preservation. Simultaneously faced with greater budgetary constraints than their developed world counterparts, the relative lack of financial resources at governments' disposal necessitates difficult and prudent trade-offs. Environmental valuation, along with cost-benefit analysis, aids developing countries in making these choices in an informed manner.

The differing circumstances in labor, goods, and financial markets under which developed and developing economies operate have no bearing on the fundamental principles underlying cost-benefit analysis. However, in applying the principles, certain valuation techniques commonly used in developed countries are not appropriate for developing countries (see Quah 2013 for a detailed discussion). Indeed, most revealed preference approaches, including hedonic pricing and the travel cost method, require strong assumptions of rationality, perfect information, and perfect mobility in order to be valid (Quah and Ong 2009), while stated preference approaches, including the contingent valuation method, are susceptible to a large number of behavioral effects (Kahneman and Knetsch 1992; Carson, Flores, and Meade 2001) and methodological biases. In the context of a developing nation, such flaws may be magnified. In the example of the national park, if fuel were distributed through a rationing system in a developing country, then the private cost of traveling would be very difficult to determine, and the demand curve obtained through typical travel cost techniques would be inaccurate. For stated preference approaches, behavioral effects may be more pronounced in developing economies owing to people's relative lack of experience of participating in survey research. List (2003) has shown that behavioral effects are, at least in part, brought about by a lack of experience with decision-making circumstances. Therefore, the magnitude of behavioral biases in stated preference approaches is likely to be much more significant in developing nations. Methodological biases in stated preference approaches also tend to be larger in developing nations because of the general lack of trained interviewers (Hanley and Barbier 2009). One common problem is the inability of both interviewers and interviewees to differentiate between willingness to pay and ability to pay. Such misunderstandings are further exacerbated by cultural and linguistic differences. In addition, surveys typically carry significant costs that cash-strapped governments will be hard-pressed to cover. Thus, particularly for developing nations, these two valuation techniques have obvious pitfalls that may render results dubious.

If a primary study is required, the paired comparison approach may prove to be the best solution for developing countries, as it avoids the obvious flaws of the other two methodological classes (Quah, Choa, and Tan 2006). Given that a paired comparison uses surveys, like stated preference methods, it avoids the need for strong assumptions as required by revealed preference methods. It also overcomes the key behavioral effect that plagues contingent valuation methods, which is known as the endowment effect. Paired comparison also offers a third reference point: that of the selector. As no real or perceived loss occurs in this case, behavioral effects like loss aversion, which can affect the results of a willingness-to-accept survey, are avoided (Kahneman, Ritov, and Schkade 1999). In a case in which there is no need for a primary study, benefits transfer can prove to be a low-cost approach in terms of money as well as time.

When conducting environmental valuation and cost-benefit analysis in developing countries, it is also important to not entirely forego intragenerational and intergenerational equity in the pursuit of efficiency. For intragenerational equity, one should note that developing countries generally lack governmental channels, such as progressive taxation and estate taxes, to redistribute wealth and prevent the income gap from widening too much or too quickly. In fact, prevalent corruption, a chronic problem for most developing nations, specifically prevents the formation of such channels, because it is often in politicians' interests to line the pockets of their business sector donors. Furthermore, income inequality is generally a greater problem for developing nations than for developed nations. When ranked by their Gini coefficients, the ten countries with the highest income inequalities are all developing nations, while the majority of the ten countries with the lowest income inequalities are developed nations (UNDP 2016). One commonly proposed strategy is to apply weights to costs and benefits in order to reflect the relative importance of monetary values to different social classes. Benefits or costs accruing to low-income groups may be multiplied and thus magnified, and projects in their favor will thus have greater likelihood of being approved. For intergeneration equity, there is a tendency for current generations to bias environmental decisions against future generations. Like labor and goods markets, financial markets in developing economies are also weaker than those in developed economies. Private banks in developing countries usually wield considerable monopolistic power, which they may exploit by charging interest rates above what a free market would produce (Yildirim and Philippatos 2007). This implicates the issue of temporal discounting when dealing with future benefits and costs, as the social discount rate should ideally take into account both the opportunity cost of capital and a society's time preference. Intertemporal discounting has to be done properly to avoid downplaying the future society's welfare, and to accurately quantify the transfer of resources across generations. We need to consider whether future gains are to be sacrificed for immediate losses, or if current costs are to be incurred for future benefits.

Finally, when using environmental values to guide public policy and to devote and divert funds amongst competing needs, one should always bear in mind that the conversion or destruction of environmental assets is often an irreversible process. Furthermore, the value of the ecosystem in its entirety, along with its life-support functions, is not simply the sum of all environmental goods. The dimensions of environmental goods are profound and their relationships with the ecosystem may not be immediately apparent. Nevertheless, despite its limitations, economic valuation of the environment provides useful information, such that the public policy choices made are commensurate with society’s welfare, and it is the first and critical step to guide us. Valuing the environment is never perfect, but some valuation is generally better than none.

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