VALUING ECOSYSTEMS

A METHODOLOGICAL APPLYING APPROACH[1]

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Abstract:

In this paper the ecosystem's valuation framework is described and discussed at a conceptual and formal level. Following utilitarianism and the capital asset analogy, one defines the concept of ecosystem value and how to quantify it by using individual preference based techniques rooted in welfare economics, namely stated individual preference techniques like Contingent Valuation. Several controversial questions arise when one tries to compute ecosystem's value by using utilitarianism and the capital asset analogy due to the particular ecosystem's natural specifics and the limitations of the economic theoretical framework. These controversial questions are enumerated, analysed, and the most commonly practitioner practices used to overcome the theoretical and technical difficulties of the appliance are assessed.

JEL: Q2; Q3; M4.

Key Words: Ecosystem; Valuation; Total Economic Value; Utilitarian Discounting; Contingent Valuation Method.

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1. Introduction

In a very general and simplified way of saying, ecosystem[1] valuation means, from the economic point of view, the act of measuring the usefulness that the ecosystem has to society by using some type of *numeraire*, normally money.

Ecosystem valuation within the context of environmental decisions is an incentive measure and a support for decision – making and has three main types of use:

i) Cost-Benefit Analysis (CBA) of investment projects, policies, and decisions that more or less may somehow affect or destroy that minimum level necessary for ecosystem to sustain[2] and whose destruction can lead to an irreversible loss of the natural, basic functions[3], for both current and future generations;

ii) Environmental accounting at the national, local and firm level (green and community green accounts, and environmental reporting);

iii) Incentives to enforce local communities to comply with conservation, i. e. compensation payments that are given to locals and stakeholders for setting aside to conservation extensions of private owned land, containing ecosystems; or to compensate them for losses related with any eventual natural resource injury originated from other individual or institutional decisions (i. e. Natural Resource Damage Assessment); or even to reward locals and stakeholders for having taken their economic decisions in compliance with conservation rules, and forgone economic gains.

Initially, the importance of environmental valuation has been tiny related to CBA specially in the US, where this economic technique has been used extensively as an input in decision making ever since President Reagan issued Executive Order 12292 in 1981. This Executive Order imposed the formal analysis of costs and benefits to federal environmental regulations with significant costs or economic impacts.

In Europe, CBA has a long tradition only in the evaluation of some investment projects, like the evaluation of transportation investment projects in many countries, although it seems to be

[1] An ecosystem is a natural system with animal and vegetation's populations and the assemblage of the particular physical conditions within which those populations live in (Gilpin 1992). In some highly populated regions like Europe, ecosystem encompasses large areas of semi-natural vegetation interspersed with grazing areas, hedgerows, farmland, and small villages and towns.

[2] To be sustained, ecosystems need to maintain their resilience. Resilience means the ecosystem's capacity of life-support to absorb internal and external levels of stress or shock, without flipping the current ecosystem's state to another regime of behaviour, i. e. another stability domain.

[3] Ecosystems have several functions like to maintain ecosystem's functional robustness and ecosystem resilience (these are the primary functions of an ecosystem); to support gene and species diversity (biodiversity) including human communities, and to produce natural goods and services as well (these are the secondary functions of ecosystems).

no legal basis for CBA in any European country. Only UK requires a comparison of costs and benefits – UK Environment Act. However, there are countries with some administrative guidelines for project and policy evaluation and in some cases a section on environmental valuation techniques is included (see Bonnieux et Rainelli 1999 and Navrud and Pruckner 1997).

During the last decade, however, there has been an increasing interest in the use of environmental valuation not only for CBA but also forenvironmental accounting and costing and as an economic incentive framework measure. This is the case not only at the country level, but also at the international organisations level as well like OECD and World Bank, and regional organisations like EU and Asian Development Bank (see Navrud 2000).

There have been numerous environmental valuation studies of biodiversity and ecosystem's functions, but very few in what total economic value of ecosystems concerns (see Nunes and van der Bergh 2001, and Navrud 1999 for overviews of some valuation studies in the US and in Europe). There are several reasons to explain this shortage of empirical total economic ecosystem valuation' studies. Among others we highlighted the following reasons for such state of the art: i) the economist's failure in recognising and defining the complexity of the ecosystem as a system; ii) some misunderstanding surrounding the concept of economic value, which arises because economists do sometimes a poor job with the explanation of the concept and the methodology used to determine it; iii) economic valuation methods are difficult and costly to apply; they are based on econometrically sophisticated methods that are not easily understood by non-economists.

It has been the conjunction of several items closely related with the environmental economic valuation framework that justifies the aim of this paper. The misunderstanding surrounding the economic and methodological valuation framework coming from both economists and non-economists, in combination with the increasing interest and importance that valuation is been awaken as an institutional tool to improve environmental management policies, are the main reasons that justify this paper. The main issue is though to enumerate, characterise, and discuss, at a conceptual level, the basic methodological steps practitioners must go through to monetise the value of an ecosystem for environmental accounting and incentive measures building. The main focus of debate within the actual scientific literature is referred for each methodological step, as well as the answers that practitioners has being assigned to resolve the several obstacles that characterised the economic valuation framework. The issue of the paper is by no means innovator. What one had in mind was to contribute to a greater comprehensiveness of such an important environmental management tool within a backdrop of lack of interest and knowledge towards economic valuation, particularly felt among Portuguese scientific and decider communities.

The paper is composed of five sections following this Introduction. In section 2 we describe the concept of total economic value of an ecosystem using the natural asset analogy. In section 3 and section 4 we formalise the concept, and describe and discuss conceptually the main steps practitioners have to follow in order to get an estimate for the total economic value of an ecosystem in the absence of markets. In section 5 the main problems arising from the introduction of the temporal dimension in the methodological valuation framework are assessed and finally in section 6 we present the conclusions and some final remarks.

2. The Value of Ecosystems: What Does this Mean to Economics?[1]

In common usage, *value* means importance or desirability. To an economist, the value of an ecosystem is related to the contribution it makes to human well-being[2]. We are dealing with a very clear anthropocentric, utilitarian viewpoint according to which ecosystems are valuable insofar as they serve humans or to the extent they confer satisfaction on humans (Goulder and Kennedy 1997). We would like to underline that utilitarianism is not necessarily synonymous with exploitation or depletion of nature. On the contrary, it can be consistent with nature conservation where protection is perceived as a source of satisfaction or well-being.

The utilitarian approach allows value to arise in a number of ways depending on how individuals use ecosystems. Ecosystems are natural capital, generally defined as the stock of environmentally provided assets (e. g. soil, atmosphere, forests, water, wetlands, minerals) that provide flows of natural goods and services that are appropriate directly and indirectly by economic sector and society in general at free cost (Serageldin 1996).

Accordingly to the type of functions provided by ecosystems, the ecosystem value can be classified in p*rior value* and s*econdary value*. The *prior value,* also called *primary value*, consists of the system characteristics upon which all ecological functions depend (resilience capacity, individual resource stability, biodiversity retention) (Turner 1999). Their value arises in the sense that the ecosystem's characteristics produce other functions with value –

[1] See for instance Smith 2000a) to read more about the current state of non-market valuation.
[2] Human Well-being depends on the basic requirements for a good quality of life including freedom of choice, health, good social relations and security. Well-being may be broadly understood as happiness. Well-being as experienced and perceived by humans is situation-dependent, as it reflects the local geography, culture and environmental circumstances.

the *secondary functions*. The variety and importance of these *secondary functions* and associated values depend on the maintenance, health, existence and the operationally of the ecosystem as a whole. The *primary value* is related to the fact that the ecosystem *holds secondary functions and values* and, as such, in principle, has economic value.

Accordingly to the type of use society makes of ecosystems, economists have settled for taxonomy of total ecosystem value interpreted as a *Total Economic Value* (TEV) that distinguishes between *Direct Use Values* and *Passive*[1] *(Non-use) Values*. TEV and its components has been the subject of huge debate among environmental economists, ecologists, psychologists, and others, about the viability, the usefulness, or the ethics of monetising it, especially passive uses[2]. Nevertheless there is actually a growing trend towards using the TEV measure concept, instead of the direct use value and/or the passive value separately, on the grounds that theoretically there is no need to adopt a dichotomy that involves the adoption of arbitrary assumptions[3]. Advances in ecological economic models and theory also seem to stress the value of the overall system[4] as opposed to individual system components. This clearly points to the value of the system itself (the *primary value*) when exhibiting resilience capacity defined as the ability of the ecosystem to maintain its properties of self-organisation and stability while enduring stress and shock (Turner 1999).

Use Values include[5]:

- *Direct Use Values*: these derive from the benefits appropriated by the society that arise from the actual use of natural resources for agricultural, fishing, forestry, industrial, and commercial purposes, land use, or self-consumption purposes (e.g. harvesting timber, fishing, collecting herbs and minerals); tourism and recreation[6]; education and research[7]; aesthetic, spiritual and cultural ends;

[1] Passive Use is now used interchangeably with Non-use or Existence Value. Other terms that have been used include preservation value, stewardship value, bequest value, inherent value, intrinsic value, vicarious consumption and intangibles (Carson et al. 1999).
[2] For a more comprehensive understanding of this debate see for instance Turner 1999.
[3] See Randall 1991 and Smith 2000b) for a more comprehensive understanding of total value and non-use value discussion.
[4] Environmental resources are increasingly recognised as assets providing services that are no longer readily available. Increasing demand to measure their value and incorporate them into legal, political, and economic decisions is a clear sign of what we would expect as their scarcity grows (Smith 2000b).
[5] See OECD 1999 for a more detailed definition of the different type of uses. See also Daily 1997.
[6] For instance, see Mendes 1997 for an empirical estimation of the value of one day of leisure and recreation in a National Park using the Travel Cost Valuation Approach, and Mendes 2003 for the estimation of a recreation use price.
[7] One of the more recognised sources of value is that associated to knowledge. According to a report for the U.S. National Academy of Sciences, the basic source of over $60 billion in current market value was obtained from biodiversity (plants and insects). Furthermore, several of the most productive and robust grain species were genetically derived from wild specimens. Currently, most drug and biotechnological companies are well aware of the strategic, commercial and scientific value of ecosystems as vital depositories of biodiversity. See for instance Chichilnisky and Heal 1998 for more details of these deals. See also Metrick and Weitzman 2000 for their interesting parabolic perspective of ecosystem and biodiversity in terms of Noah's Arc.

- *Indirect Uses*: related with the benefits arising from the use society makes of ecosystem functions like watershed values (e.g. erosion control, local flood reduction or regulation of stream-flows) or ecological processes (e.g. fixing and cycling nutrients, soil formation, cleaning air and water). It further includes *Vicarious Use Value* addressing the possibility that an individual may gain satisfaction from pictures, books, or broadcasts of natural ecosystems even when not able to visit such places;
- *Option Values*: related with individual willingness to pay a premium to ensure future ecosystem availability and usage;
- *Quasi-Option Value*: refers to individual willingness to pay a premium to ensure more accurate scientific information;

Passive (Non-use Values) include:

- *Existence Value*: reflects the moral or altruistic satisfaction felt by an individual from knowing that the ecosystem survives, unrelated to current or future use;
- *Bequest Value*: considers individual willingness to pay a premium to ensure that their heirs will be able to decide about the better ecosystem's use in the future.

The economic value of an ecosystem thus relates to the TEV.

However, TEV is not an absolute value because economics provides valuations only in comparative terms. When they say they are valuing an ecosystem, economists are really defining a trade-off between two situations involving a change: e.g. maintenance or non-maintenance of the ecosystem. Following the Hicks 1939's and Kaldor 1939's generic economic definition of value, the economic value of the ecosystem will then be the amount an individual would pay or be paid to be as well off with the ecosystem or without it.

Thus, economic value is an answer, mostly expressed in monetary terms (but not necessarily), to a carefully defined question in which two alternatives are being compared. The answer (the value) is very dependent on the elements incorporated in the choice, which are basically two: the object of choice and the circumstances of choice (Kopp and Smith 1997). Economics defines *objects of choice* as any tangible or non-tangible object, process or activities that allow for a choice. The objects of choice are defined by a set of characteristics and attributes that are perceived by individuals but not necessarily by all individuals. In our case, the object of choice will be an ecosystem whose specificity is defined by a set of environmental and ecological attributes to a greater or lesser extent perceived by individual users and passive users. The *circumstances of choice* describe the context in which that choice is made (e. g. to accept the political option to conserve the ecosystem or, alternatively, to accept the political option of non-conserving the ecosystem). It is important to describe to the individual the consequences of his/her choice, specifically in terms of: i) what is foregone by the choice and what is gained; ii) specifying the rights of assignment; iii) defining the mechanism of choice, i. e. the manner through which the individual will exercise choice: by voting, or through private market transactions or other unspecified behaviours.

The *object* and the *circumstances* surrounding one choice define the *context* of the choice. In the case of ecosystems, value will depend on the ecosystem's location and the level of human presence in it, the actual or threatened level of degradation as well as the degree to which natural services provided may be or may not be substituted by other substitute ecosystems. The substitutability is a highly important concept within the economic valuation's framework, as objects with significant numbers of close substitutes are not rated as valuable as others with few or even no substitutes. In the case of ecosystems, the degree of substitutability is relative and dependent on factors like the scale and level of aggregation and the time-scale involved. If the ecosystem is classified as a Protected Area, substitutability can be contested, as Protected Areas are defined when unique ecosystems producing rare, non-substitutable amenities face very serious depletion and extinction risk.

For specifying rights of assignment, there are two possible choice situations. Either the individual gives something up to receive the object of choice that will affect his/her utility or well-being or the individual receives something to give up the object of choice that could affect his/her utility or well-being. The former situation corresponds to *Willingness to Pay* (WTP) and the latter to *Willingness to Accept* (WTA) and these are the fundamental monetary measures of value in economics.

These welfare measures applied to non-market transacted objects of choice as is the case of ecosystems were first proposed by Mäler (1971; 1974) as an extension of the standard theory of welfare measurement related to market price changes formulated by Hicks (1943). The analysis of this type of problems that involve changes in either the quantities or the qualities of non-market environmental goods and services rather than changes in prices or income is often referred to as the *theory of choice and welfare under quantity* (Johansson 1987, Lankford 1988).

Mäler stated that it was possible to build four measures of individual welfare change associated to choices involving non-market goods. If the object of choice generates an *improvement* in individual well-being (a rising utility), two situations become possible. Either the individual is WTP an amount to secure that change, termed *Compensated Willingness to Pay* (WTP^C) or he/she is willing to accept a minimum of compensation to forgo it, the *Equivalent Willingness to Accept* measure (WTA^E). If the object of choice generates a *deterioration* in well-being (a decreasing utility), again two situations are possible. Either the individual is WTP to avoid this situation, termed the *Equivalent Willingness to Pay* measure (WTP^E) or he/she is WTA compensation to tolerate the damages suffered, the *Compensated Willingness to Accept* measure (WTA^C). When economists talk about the value of an ecosystem they than are referring to an individual TEV measured by one of these four welfare measures: WTP^C/WTA^E if the individual faces an improvement of well-being; or WTP^E/WTA^C where the individual faces deterioration in well-being.

Following Mäller's basic model of individual utility one can define welfare measures related with the ecosystem preservation. If ecosystems are objects of choice, then a change of the quality of their environmental amenities matters to the individual as well as the ecosystem existence or non-existence. Then the changes must be shown up either in the individual preference function or in a constraint.

Let U = U (**x, q**) be the utility function of an individual with preferences for various conventional market commodities and where consumption is denoted by the vector **x** (**x** = $x_1, \ldots, x_i, \ldots, x_n$), and for non-market environmental amenities denoted **q** (**q** = $q_1, \ldots, q_j, \ldots, q_m$). **q** may be a scalar where related to a single amenity or is a vector where related to several amenities as is the case of **q** representing the ecosystem one wishes to value. The individual takes **q** as given which means **q** is a public good. It is also assumed that preferences represented by the utility function are continuous, non-decreasing and strictly quasi-concave in **x**[1]. The individual faces a budget constraint based on their disposable income m, and the prices of market commodities, **p**. We assume there are no positive prices for the **q** elements. The individual maximisation utility problem of decision is then formalised as:

$$\max_{x^*} \; U(x,q) \qquad \text{subject to} \; \sum_i p_i \, x_i = m \qquad (1)$$

The solution of this problem yields a set of ordinary or Marshallian demand functions for **x** denoted

$$x^*_i = g_i(p,q,m) \qquad (2)$$

for i = 1, …, n market commodities, and an indirect utility function as well denoted

$$U(x,q) = \varphi(p,q,m) = U\left[g_i(p,q,m);q\right] \qquad (3).$$

The dual is an expenditure minimisation model defined by:

$$\begin{aligned} &\min_{x_c} \ \sum_i p_i x_i \\ &\text{subject to} \quad U(x,q) = U \end{aligned} \qquad (4)$$

The solution of the dual yields a set of compensated or Hicksian demand functions for **x** denoted

$$x_{ic} = h_i(p,q,U) \qquad (5)$$

for the i =1,…, n market commodities, and an expenditure function as well

$$m = e(p,q,U) = \sum_i p_i h_i(p,q,U) \qquad (6).$$

Let us assume that a representative individual has preferences for various conventional market commodities **x** and for non-market environmental amenities provided by one ecosystem, and that ecosystem is menace by destruction. If the ecosystem is going to be destructed the individual will face that loss and **q** is going to change *ceteris paribus* to reflect that loss. The individual i will have then to choose between two states. The state q^0 (the initial state characterised by the preservation of an healthy ecosystem that produces the amenities **q**); and the state q^1 (the final state characterised by the ecosystem destruction and the loss of the natural amenities produced by it) where $q^1 < q^0$. If he or she chooses q^0 the level of utility is given by $U^0 = U^0(p,q^0,m)$; and if he or she chooses q^1 the utility is given by $U^1 = U^1(p,q^1,m)$, so that $U^1 < U^0$. The welfare change associated to this utility level change can be measured using Mäler's Compensation Surplus (CS) or Equivalent Surplus (ES) measures, defined respectively by:

[1] The specific form of the utility function will affect the shape of the indifference curves. The shape of the indifference curves indicates the preferences the individual has for **x** and **q**. In this case, to say the utility function is quasi-concave is merely for the sake of analytical convenience. To say this is realistic or not is considered by utilitarian theory as an empirical question (Hanemann 1999).

$$\varphi\,(p, q^1, m + CS) = \varphi\,(p, q^0, m)$$

And (7)

$$\varphi\,(p, q^1, m\,) = \varphi\,(p, q^0, m - ES)$$

The choice between CS and ES depends on the same consideration applying to the choice between CV and EV measures for price change.

The sign of CS and ES depends on the change in **q** being an improvement or a loss. If **Δq** is an *improvement*, then $\Delta U = U^1 - U^0 > 0$, CS measures the individual's maximum willingness to pay something (WTP^C) to secure the change, and ES measures his or her minimum willingness to accept something (WTA^E) to forgo it. Conversely if **Δq** is a *loss* as is the case of our ecosystem, then $\Delta U = U^1 - U^0 < 0$, - ES measures the individual's WTP to avoid the change (WTP^E), while – CS measures his or her WTA to tolerate it (WTA^C)[1].

Given the duality between the indirect utility function and the expenditure function, equivalent definition of CS and ES can be written in terms of the expenditure function as:

$$WTP^C / WTA^C = CS = e(p,q^1,U^0) - e\left(p,q^0,U^0\right) = \int_{q^0}^{q^1} \frac{\partial e\left(p,q,U^0\right)}{\partial q} dq \qquad (8)$$

And

$$WTP^E / WTA^E = ES = e\left(p,q^1,U^1\right) - e\left(p,q^0,U^1\right) = \int_{q^0}^{q^1} \frac{\partial e\left(p,q,U^1\right)}{\partial q} dq \qquad (9)$$

In equations (8) and (9) $\frac{\delta e(p,m,U^t)}{\delta q}$ [2] is the derivative of the expenditure function with respect to **q**, where t = 0 refers to the initial level of utility and t =1 the final level of utility after the change in **q**. Such derivative represents the marginal value of a small change in q and is equal to the income variation that is just sufficient to maintain utility at its initial level (in the case of CS money measure, t = 0) or final level (in the case of ES money measure, t =1). In geometrical terms, the absolute value of the derivative of the expenditure function with respect to **q** is equal to the slope of the indifference curve through the point at which the

[1] By convention, CS and ES can be positive or negative and the sign depends on the decreasing or increasing level of utility, associated with the change in **q** but WTP and WTA are always defined so as to be non-negative (Hanemann 1999).

[2] One can prove that $\frac{\delta e(p,m,U^t)}{\delta q} = p\ \delta x_C / \delta q = -\mu\ \delta\, U(x_C, q) / \delta q$, where $x_C = x_C(p, q, U)$ is the compensated demand function for private goods **x** and μ is the marginal valuation of the ecosystem (Johansson P-O 1987).

welfare change is being evaluated. Within the same equations, the integral is the value of a non-marginal change in **q** for the relevant range.

The geometrical representation of CS and ES money measures associated to the ecosystem destruction is shown in Figure 1, where $U^1 < U^0$, x is the *numeraire* with a price of one (p = 1) and the budget line is horizontal i. e. x = m. This means individual will spend all his or her available income in x for every level of natural amenity **q**.

It is assumed that at given prices and income, the individual will always choose more of q if given the option.

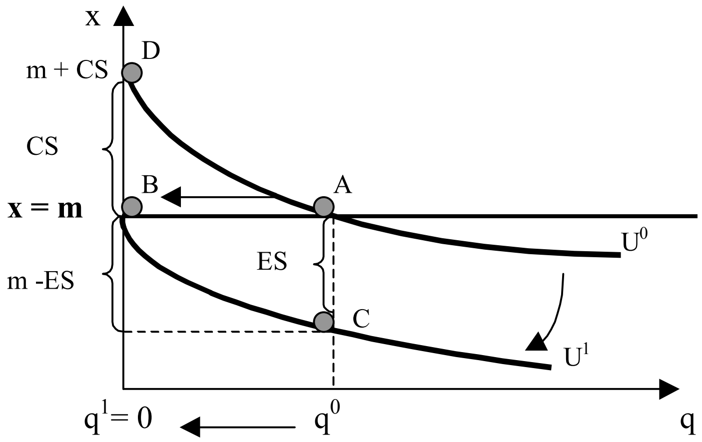

Figure 1 Compensating and Equivalent Surpluses for a loss in q from q^0 to q^1, where $q^1 = 0$

In Figure 1 the point A is the initial consumption level of the individual, where he or she consumes q^0 and spend all his or her available income in x, and achieves the utility U^0. The decrease in q from q^0 to $q^1 = 0$ (because of the destruction of the ecosystem) enables the individual to reach U^1at point B. At B if income is increased by CS the individual is pushed back to U^0 but at point D compensating him or her monetarily for the welfare decreasing associated with the destruction of the ecosystem (WTA^C). Similarly, at A, if income is reduced by ES, the individual is pushed to U^1 at point C: the individual prefers to pay to avoid the loss of welfare associated to the destruction of the ecosystem (WTP^E).

Generally, WTA is different from WTP. Actually there is a substantial body of evidence from both stated preference studies and laboratory experiments in evaluation that these differences may be quiet large. Such discrepancies[1] stem from the different welfare measure definitions

[1] See Hanemann 1991 for more comprehensive analysis about the WTP and WTA discrepancy when applied to the valuation of environmental services.

and contexts of choice one has to deal with. The WTP economic value of an object of choice is constrained by individual wealth and by the existence or non-existence of substitutes. These constraints are not present in the WTA money measure. WTP and WTA are approximately equal only when income changes, while (p, q) remain unchanged. When m^0 changes to $m^1 = m^0 + \Delta m$ the resulting CS and ES are equivalent and equal to Δm. But for (p, q) changes, WTP and WTA are different.

Randall and Stoll (1980) extended Willig's analysis (Willig 1976) of welfare measures for changes in q, to bound the differences between WTP and WTA. They found that, with some alterations, Willig's formulas could be used, by using information on the value of the price flexibility of income over the interval $[q^0, q^1]$. Later on, Hanemann (1991, 1999) showed that the price flexibility of income is the ratio between the income elasticity of demand for **q** and the aggregated Allen-Uzawa elasticity of substitution between **q** and **x**. If the elasticity of substitution is low, CS may be very different from ES.

Over the past fifteen years, considerable evidence has been accumulated regarding the existence of some pattern of individual behaviour in judgement and choice involving phenomena commonly known as *loss aversion*, *endowment effect* or *status quo bias* (Horowitz and McConnel 2002). In short, the essence of these phenomena is that individuals weight losses more heavily than comparable gains. As Kanemann and Tversky (1979) stated, there is some evidence that individuals experience "loss aversion", that is individuals may be expected to value a unit of loss more highly than a unit of gain where he or she believes that some right to the current amount of environmental asset exists.

WTP is only be equal to WTA where there is no income effect, where there are perfect substitutes of the object under valuation and where the individual is neutral to losses and gains.

WTP and WTA are though the fundamental monetary measures of value in economics for market and non-market commodities and when economists set about measuring them they do it by observing the preferences that individual have towards market and non-market commodities. Standard theory assumes people have well-defined preferences over alternative bundles of consumption goods, including non-market goods, and that people know their preferences. Also it is theoretically assumed that preferences are non-lexicographic, in other words, that preferences have the property of substitutability.

Preferences for private commodities are revealed directly by using actual, observed, market – based information, at the very moment the individual purchases them on the market. But as for the ecosystems because they are not market tradable, individual preferences have to be elicit directly by simply asking people what is his or her WTP/WTA for a certain object of choice – the preservation or the destruction of the ecosystem. Because people feel satisfaction with ecosystems and the amenities they provide as reproducible natural asset, the ecosystem's economic value will be necessarily linked to the value of flows of services and passive use it generates, at the moment the valuation question is being made. However, the ecosystem and its components provide flows of services and of passive use over a time path and not only at the time of the questionnaire. So, the value of the ecosystem accordingly to the asset analogy will be equal, not to the sum of the individual stated WTP/WTA at the moment of the questionnaire, but to *discounted sum of the stated WTP/WTA over individuals for those services and passive use benefit flows instead*.

On using the natural asset analogy one must be aware that ecosystem's changes depend on changes in the quality/quantity of service flows and their passive use values; where there are substitutes for ecosystem services, the TEV will respond to changes in that substitute; and that TEV will depend on individual income constraint and methodological factors as well.

To estimate TEV one though has to go through the following steps: i) to estimate WTP/WTA for an ecosystem that produce amenities by assessing the value individuals give to the existence of that ecosystem after real responses stated by individual; ii) to estimate TEV, using the aggregate values of stated WTP/WTA discounted for some rate, following the natural asset analogy.

3. Estimating the Ecosystem's TEV at Time t by Using Individual Stated Preferences

The Contingent Valuation Method (CVM)

Following the previous theoretical approach, let now q^0 be a state characterised by an existing ecosystem and q^1 the one characterised by the loss of that ecosystem so that $\Delta U = U^1 - U^0 < 0$. Accordingly to Mäller's welfare measures for quantity changes, WTP^E is the maximum individual's equivalent WTP to avoid that loss while WTA^C is the minimum compensation he or her is WTA as a compensation to tolerate the disutility of the loss of an ecosystem.

Contingent Valuation Method (CVM) [1] provides the means to estimate WTP^E and WTA^C after real responses.

Accordingly to CVM, by drawing a sample of people from a population, one simply has to ask them about their maximum WTP^E or minimum WTA^C. WTP^E is the more applied measure because it overcomes the individual's "risk aversion" related problems. The type of population one has to consider depends on the ecosystem's characteristics in terms of size and ecological importance. These direct valuation questions permit to derive a set of welfare measures w_i (i = 1, …, n individuals) for the n respondents of the sample and for the moment t, the moment the questionnaire is displayed where $w_i = WTP_i^E$.

An estimate of TEV for the relevant population at the moment t can be obtained so that

$$TEV^t = w^t \times N \quad (10)$$

Where N is the population and w_t is some mean estimated from the w_i answers.

Three methods can be used (Freeman III 2003) to get the mean w^t. It may be simply equal to the sample mean w*. Or, alternatively, w^t can be obtained as a bid function for the given change from q^0 to q^1 by regressing the responses on income and other socio-economic characteristics to obtain $w^t = w^t$ (m, SE, P) where SE is a vector of socio-economic characteristics of the relevant population; P is a vector of variables that measure the degree of individual's ecosystems perception that condition his or her attitudes towards ecosystem conservation. A third approach is to obtain w^t as a bid of a function as well, but by including the **q** variation across the sample as part of the survey design, obtaining $w^t = w^t$ (Δq, m, SE). This bid function can be estimated and then used to calculate the values associated with different alternative scenarios regarding ecosystem changes[2].

The Validity and Reliability of the CVM's Estimates

The use of CVM to estimate the theoretical economic measures to quantify the TEV of natural amenities, and specifically the WTP^E to avoid the ecosystem destruction, has been one of the most fiercely debated issues within environmental economic valuation literature over the last twenty years. One of the main debated issues has been the validity and reliability's problematic of CVM' WTP^E estimates as the key to any evaluation method is that it must be assessed in terms of how closely it represents an accurate measurement of the real value. The

[1] For a detailed description of the CVM see Mitchell and Carson 1989. For a more theoretical detailed description of economic valuation methods see Freeman III 2003, or Braden and Kolstad 1991. See also Hufschimdt et al 1983. For examples of valuation methods applied to value ecosystem's services see Goulder and Kennedy 1997. See also Carson et al 2003 for an elaborate and state-of-the-art valuation study.
[2] See Mitchell and Carson 1989 for details about this and about all the other issues concerning CVM application to value environment.

closer the real values are to the estimated, the more accurate the valuation method is. If WTP/WTA were observable there would be no problem. But given they are not, it is then necessary to use other complex criteria and "rules of evidence" to assess accuracy. In measurement, accuracy means the *reliability* and *validity* of data analysis used for the valuation framework[1].

From the economics perspective, *reliability* is related with the accuracy of aggregate WTP over appropriately defined aggregates of individuals. Economists tolerate certain amounts of unreliability in the estimated WTP, if random errors in measurement remain within tolerable boundaries. Thus, the valuation technique reliability depends on the degree of data "noise". The bias between the CVM estimated WTP/WTA and the theoretical WTP/WTA grows where the former tends to systematically diverge from the latter.

The concept *validity* relates to the CVM application process. It involves numerous issues that must be resolved mainly based on individual judgement of the CVM implementing entity. Because WTP/WTA are not observed, inferences as to validity have to be based on indirect evidence related both with the *content validity* of the CVM study's design and execution, and with the *construct validity* dealing with the degree to which the estimated money measure relates to other theoretical measures. A number of guidelines have been developed to assume CVM credibility, validity, and reliability (Portney 1994; Arrow et all 1993). The most important are related to the presentation of adequate information over the object of choice (i. e. the ecosystem[2]) and the context of choice[3], the choice of a credible (hypothetical) payment mechanism[4] and the use of a referendum format[5].

Detractors argue that respondents provide answers inconsistent with the basic assumptions of utilitarian rational choice because of the *embedding effect*. *The embedding effect* refers to several interrelated regularities in contingent valuation surveys like insensitivity to scale and scope, sequential effect and sub-additive effect. Firstly, WTP is sometimes much less dependent of the quantity of the public good provided that it should be theoretically (*insensitivity to scale and scope*). Secondly when the several public good are valued in the same survey, the WTP for a particular one often depends on its position in the sequence of

[1] See Mitchell and Carson 1989 for a comprehensive description of these methodological CVM problems and their potential effect upon estimates and Freeman III as well. See also Jakobsson and Dragun 1996 for a comprehensive survey of literature on such issues.
[2] The attributes of the ecosystem, the level of the provision of those environmental attributes "with or without intervention" and if there are undamaged substitute commodities. The researcher must previously determine which attributes (services) affect the value an individual places on a good (Green and Tunstall 1999; Fischoff and Furby 1988).
[3] To explain the extent of the market by informing respondents of how and when the environmental change will occur, as well as the decision rules in the use for such provision e.g. majority vote, individual payment.
[4] Taxes, property taxes, sales taxes, entrance fees, changes in the market prices of goods and services or donations to special funds.
[5] This is the only elicitation format that is, under certain circumstances, incentive compatible.

public goods (*sequential effect*). Finally, the sum of WTP for individual changes often exceeds the WTP for a composite change in a group of public goods (*sub-additive effect*). Some CVM's critics see the *embedding effect* as evidence for non-existent individual preferences for the public good but an individual *worm glow* effect instead, created by the survey process itself. Defenders acknowledge that early applications suffered from many of the problems critics have noted (see Mitchell and Carson 1989); however, recognition is required of how more recent and more comprehensive studies have dealt and continue to deal with those objections (see Carson et all 2001).

More recently there has been a trend to include expertise from other disciplines such as marketing research, survey research, and psychology (both cognitive and social) to improve the CVM methodology both from the theoretical and empirical point of views. The importance of these contributions to survey research is almost intuitive because CVM is broadly survey valuation method based. Due to the hypothetical nature of the WTP question asked together with the unfamiliarity of the task, one cannot exclude the possibility of respondents to fail to consider the effect of their budget constraints and substitutes. Though, symbolic valuation in the form of attitude expression and superficial answers are to be expected (Kahneman et all 1999). Psychologists criticise the utilitarian approach[1] because they consider individuals do not chose in isolation but are affected instead by the characteristics of their particular social–economic group; and they do not choose based on only one restriction such as income. Preferences are not static and they are not equal across all individuals. Psychologists criticise utilitarian economics for the assumption that all values are commensurable and ultimately reducible to a single metric be it money or another type. Psychologists defend CVM responses are sensitive to these methodological factors that in standard economic theory are deemed irrelevant[2]. Underlying much of this discussion are implicit assumptions of what Fischoff (Fischoff and Furby 1988) called the *philosophy of basic values*.

Another capital issue of debate concerning the valuation framework approach of non-market natural amenities has been the doubt with the inclusion or non-inclusion of passive users as a

[1] See Green and Tunstall 1999 for a comprehensive introduction to the psychological perspective on economic valuation.

[2] This is the case of the so-called protest responses: i. e., protest bids (since zero to infinite bids), non-response, or *unreasonable* sacrifice. Within the context of neo-classical definition, *unreasonable* may include WTP values far beyond the individual's ability to pay, or sacrifices that involve excessive opportunity costs such as loss of human life. In the face of these responses, which are very common in some CVM studies, practitioners conclude the individual has lexicographic preferences which means he or she bases his or her responses from a hierarchy of values whose structure is dependent upon the strength of individual attitudes, beliefs or dispositions he or she holds towards the valuation context. If there are an excessive number of protest responses the scope and the quality of the information for economic assessment of values may be limited because lexicographic preferences violate the assumptions of continuously defined, differentiable and convex preferences in standard utilitarian theory (Rosenberger et al 2003). But economists tolerate certain amounts of protest responses if random errors in measurement remain within statistical tolerable bounds, as we already discussed.

TEV component[1]. Many detractors of this kind of valuation framework generally argue that passive uses cannot be seen, and though cannot be economically measured, as a simple component of the economic total value. They claim that by the fact of being strictly associated with ethical, religious principles, passive uses cannot be measured through WTP/WTA. However, as far as the ecosystem's valuation concerns, passive use will always implicitly manifest itself if individual are willing to pay some amount of money to preserve the ecosystem from typical economic activities, even though that individual is not an actual nor intend to be a future user of the ecosystem's services and would not suffered a utility decreasing derived from the destruction. Passive uses do not complicate the valuation task at a conceptual level and CVM seems to be capable of measuring TEV including *passive use*. There is however some problems with passive uses if they exist for many other natural resources at the same time and if they are widespread among the population. Being it so, the valuation task will be more complicated, because a substantial surveying effort will be necessary to get an accurate accounting for environmental amenities.

4. Estimating the Economic Value of Ecosystem as an Asset Using Individual Stated Preferences

The TEV estimated following the equation (10) is the aggregated welfare money measure of the value the relevant population puts on the existence of an ecosystem, at the moment t. But, as we already told, an ecosystem is a natural, reproducible asset that yields flows of goods and services over time. And the measure given by (10) only reflects the value of the ecosystem's goods and services flow generated at the single moment t. One though has to consider the temporal dimension of these flows.

The default criteria used by economists to include time is provided by the *discounted utilitarianism framework* which until now has dominated more for lack of convincing alternatives than because of the conviction it inspires (Heal 1998). It has proven particularly controversial not only within non-economists concerned with environmental valuations but also with some economists. We shall discuss this later on in this paper.

Following discounted utilitarianism approach (see Freeman III 2003 and Perman et al 2003) let us assume a simplified intertemporal social welfare function of the type

$$W = W\left[\phi^1 U^1 (x^1, q^1), \ldots, \phi^t U^t (x^t, q^t), \ldots, \phi^T U^T (x^T, q^T)\right] \quad (11)$$

[1] See Bateman and Willis 1999 and Jakobsson and Dragun 1996.

Where the arguments are assumed additively separable, the single-period utility function denoted $\phi^t U^t (x^t, q^t)$ is the same for all the periods, and the ϕ^{t}' s are weights used in summing utility over generations to obtain a measure of intertemporal social welfare. The utilitarianism approach to intertemporal questions typically assumes that:

$$\phi : [\phi^1 = \frac{1}{1+\rho} .....; \ \phi^t = \frac{1}{1+\rho^t} ; \dots ; \phi^T = \frac{1}{1+\rho^T}] \qquad (12)$$

where ρ is a subjective rate of time preference. If $\rho > 0$ we can write:

$$W = \sum_{t=1}^{T} \frac{1}{(1+\rho)T} U^T \left(x^T, q^T\right) \qquad (13)$$

Whose continuous time version is:

$$W = \int_{t=0}^{T=\infty} U^T \left(x^T, q^T\right) e^{-\rho t} dt \qquad (14)$$

Let us now assume a marginal decrease in q^t so that individual would be willing to pay a quantity of the numeraire $\mathbf{x}^t$ to avoid the loss in that period (WTP^{tE}) or be willing to accept a greater quantity of numeraire in that period to held utility constant (WTA^{tC}). The equivalent compensated money measures of welfare change are equal to the marginal rate of substitution between $\mathbf{q}^t$ and $\mathbf{x}^t$ so that:

$$W_{q^t}^{t} = MRS_{x^t, q^t} = \frac{\dfrac{\partial U\left(x^t, q^t\right)}{\partial q^t}}{\dfrac{\partial U\left(x^t, q^t\right)}{\partial x^t}} \qquad (15)$$

According to the first-order conditions to maximise the intertemporal social welfare function given by the equation (13) (or (14)) subject to the wealth constraint[1], where the individual's intertemporal marginal rate of substitution must be equal to one plus the interest rate, one can say that, with intertemporal equilibrium, the individual would be indifferent between paying an amount of money equal to $W_{q^t}^{t}$ in period t to avoid the loss and paying $W_{q^t}^{0} = \frac{W_{q^t}^{t}}{(1+\rho)^t}$ in

[1] $\sum_{t=0}^{T} x^t \frac{1}{(1+\rho)^t} = W^* = W^0 + \sum_{t=1}^{T} m^T \frac{1}{(1+\rho)^t}$, where W* is lifetime wealth and W^0 the initial health. The individual can borrow or lend m_t in a perfect capital market at an interest rate equal to ρ (see Freeman III 2003).

period 0, i. e. now. Or the individual would be indifferent between accepting the quantity of money $W^{t}_{q^{t}}$ now, to tolerate the loss of the ecosystem.

If the stream of future ecosystem's service flow (let us say **Q** = $q^0, \ldots, q^t, \ldots, q^T$) is expected to decrease, the marginal willingness to pay now to avoid the decreasing over the time path ($W^{0}_{Q^{0}}$), will be equal to the sum of the willingness to pay to avoid each of the components of the decreasing; or the marginal willingness to accept a compensation to tolerate the negative effects over the utility of that loss will be equal to the sum of the willingness to accept a compensation to tolerate each of the components of the decreasing so that:

$$W^{0}_{Q^{0}} = \sum_{t=0}^{T} \frac{w^{t}_{q^{t}}}{(1+\rho)^{t}} = \int_{t=0}^{T} w^{t}_{q^{t}} e^{-\rho t} dt \qquad (16)$$

where $w^{t}_{q^{t}}$ is one of the single-period welfare measures WTP ^{E} or WTA ^{C}.

Applying this intertemporal utilitarian approach to equation (10) we obtain the TEV of a natural asset generating a flow of amenities over a relevant period of time T by simply summing up the present value of the single-period welfare measures by using some discount rate ρ[1] so that:

$$TEV = \sum_{t=0}^{T} \frac{TEV^{t}}{(1+\rho)^{t}} \qquad (17)$$

whose continuous version is:

$$TEV = \int_{t=0}^{T} TEV^{t}\, e^{-\rho t} dt \quad (18)$$

By considering these results, the typical practice in evaluating the TEV of some ecosystem as an asset, that yields a flow of amenities over many years, is to estimate a single-period welfare measure by using the CVM and to compute the present-value of this single-period benefit using some discount rate. At a first glance, this seems to be a quite easy task. However, although discounted utilitarianism approach has dominated, it has proven particularly controversial when applied to environmental valuations. At origin of such

[1] Nevertheless we can not say that compensate/equivalent welfare measures are *equal* to compensate/equivalent discounted single-period welfare measures for some discount rate ρ. This procedure will under estimate the true lifetime welfare benefit of the stream of improvements in Q, and overstate the loss of decrease in Q. However, the difference will be significant only if high values for the elasticity of intertemporal substitution combine with large values of compensate welfare measures over time. See Freeman III 2003

controversy are: the practice of the discounting, itself; the number used to measure the rate of discount, ρ; and the irreversibility, risk, and uncertainty, surrounding the future outcomes.

5. Problems that Arise From the Consideration of Temporal Dimension

The time period and the positive rate of discount

A first target of criticism concerning discounting utilitarianism is the legitimacy it self of applying discounting to the ecosystem valuations due to the very large period of time involved. A positive utility discount rate forces an asymmetry between the evaluations of current and future generations, particularly those very far in the future. The consideration of an existing positive rate of discount is a way by which the market penalises investments with long-term payoffs. This is precisely what happens when one has to value ecosystems by discounting the flow of amenities they provide over time. At any positive discount rate, the present value of any ecosystem is almost irrelevant due to geometric discounting and so it is irrational to be concerned with extinction and conservation benefits. Economists face some difficulties with valuing natural assets, because typical economic time horizons differ by an order of magnitude from those that are typical for ecosystems. To economist 30 years is a very long time. But to nature is a very short time.

It seems we have to deal here with some kind of a paradox. From one side societies are really concerned with the social costs that will occur one hundred years or more ahead if ecosystem depletion is to occur. From another side, the only convincing existing method to evaluate intertemporal projects seems not to capture a very real part of society's concern with the future. The appliance of a discount rate to a future sum can make it look very small in present-value terms when we are dealing with periods of 50, 100, 200 years or more[1].

Some notable economists like Ramsey (Ramsey 1928) argued that applying a positive rate of time preference to discount values across generations is "ethically indefensible". They are convinced that the only ethically defensible position in comparing utilities over successive generations is to treat utilities equally, which means to make ρ equal to zero[1]. However, Heal 1985, 1998 demonstrated that discounting the future utilities is in some sense logically necessary because without it a variety of non-setting inter temporal paradoxes can be founded. He showed that Ramsey's approach only works under some severe constraints, which in general are not true. It can be proved that if t tends to infinity, as is the case of preserving

[1] For instance if one discounts present world GNP over 200 years at 5% annum it would worth only a few hundred thousand euros today. Discounted at 10% it would worth an amount almost equivalent to a used car.

ecosystems, the Ramsey's approach provides incomplete rankings of inter temporal utilities. Besides, a growing body of empirical evidence suggests that the discount rate that people apply to future projects is *positive*, not zero, and depends on the futurity of the projects, and on the magnitude of the income involved. Over short 5 years periods, individuals use discount rates that are higher even then many commercial rate, around 15% and sometimes more. For projects over 30-50 years the implied discount rate drops from about 5%, and down to about 2% for projects over 100 years[2]. Some defenders argue that the use of a positive discount rate to discount the ecosystem conservation's benefits is necessarily implied by the proper logic of discounting utilitarianism and intergenerational equity[3].

Discounting is controversial and so it is the arithmetic of discounting. The utilitarian geometrical discounting based on a positive constant rate is discriminating against the future generations, by giving their utility levels less weight. But empirical evidence seems to deny the constancy of the discount rate. This put some economists[4] to think over classical utilitarian approach alternatives, particularly over those who place more weight on the future then do the conventional one.

The logarithmic discounting (or hyperbolic discounting)[5] is one of these alternatives. It is grounded in the empirical individual behaviours, which suggest that a given change in futurity leads to decreasing weighting of the future utilities, the further the event goes into the future[1]. E. g. postponement by one year from next year to the year after is much more different from postponement by one year from 50 to 51 years hence. The result is that the discount rate is inversely proportional to distance in the future, which means one has to measure t in a manner different from the one used in the utilitarian approach, by equalising *proportional* increments in time rather then by equalising *absolute* increments. In this case, (17) and (18) will be estimated by:

$$TEV = \sum_{t=0}^{T} \frac{TEV^t}{(1+\rho)^{\log t}} = \int_{t=0}^{T} TEV^t \, e^{-\rho \log t} dt \qquad (19)$$

Where $e^{-\rho \log t}$ is a logarithmic factor of discounting. When $t \to \infty$, the discount rate and the discount factor goes to zero in the limit. Unfortunately, as Heal 1998 noted, the use of a

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[1] Harrod 1948 agrees with Ramsey's opinion in the essential case of environmental subjects. In the particular context of global worming, Chine 1992 and Broome 1992 also argue for the use of a zero discount rate.
[2] Cropper et al 1994; Lowenstein and Elster 1992; Lowenstein and Prelec 1992; Lowenstein and Thaler 1989; Thaler 1981.
[3] See also Baumol 1968.
[4] Ramsey was one of these economists.
[5] Lowenstein and Prelec 1992.

deterministic declining rate, though consistent with individual preferences, produces time-inconsistent decisions.

Alternatives to use continuously declining rates are to simply use a lower rate for long-term projects. Ramsey, as we already told, defended a zero discount rate although for certain cases the zero approach provides incomplete rankings (when $t \to \infty$) or even creates mathematical problems like non-existing solutions, as it seems to be the case with the evaluation of pure environmental depletion problems[2].

Further later, economists tried to introduce other criteria like Overtaking's, Limiting Payoffs, and Chicilnisky's criteria[3] to avoid the problems arising from the appliance of a zero discount rate and yet maintaining equal weight to present and future flows. The first two criteria are criticised because they do not rank paths completely, they are neutral with respect to timing, and neglect the present to over-emphasise the long run. The later, as demonstrated by Heal, is a mixture of the discounted utilitarianism with the other two approaches. Chichilnisky proposes to replace discounted integral of utilities by the following:

$$\alpha \int_{t=0}^{\infty} \mathrm{TEV}^{t}\ \Delta(t)\, dt + (1-\alpha) \lim_{t\to\infty} \mathrm{TEV}^{t} \qquad (20)$$

Where $\alpha \in (0, 1)$ and $\Delta(t)$ is any measure of the discount factor that, in particular, could be the conventional exponential factor $\Delta(t) = e^{-\rho t}$, or the logarithmic factor of discounting, or even the general non-constant factor discounting (like in the empirical case). This is, in effect, a mixture of the other approaches. The left side of the addition is a generalisation of the discounted utilitarian approach; and the right side represents the sustainable utility level attained by the environmental related decision (see Heal 1998 for details).

More recently Newell and Pizer 2003 described an alternative explanation of declining discount rates that fits within the standard framework of geometric discounting based on market-revealed rates. They introduced only one alteration in the geometric discounting framework, by substituting a certain, constant discount rate, by an uncertain one. By doing so, they claim there will be an increase in the expected net present-value of future payoffs.

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[1] For a comprehensive discussion over the rationales to adopt the hyperbolic discounting see Harvey 1994, Weitzman 1998 and Newell and Pizer 2003.

[2] See Heal 1998 for a clear discussion of this issue.

[3] See Heal 1998 for a comprehensive theoretical description of these criteria.

Irreversibility and Uncertainty

Beyond the controversy surrounding the discounting utilitarian framework and its arithmetic's some other important questions arise where ecosystem preservation is discussed like *irreversibility* and *uncertainty*. With economic growth and technological change it is reasonable to assume a tendency for the relative value of ecosystem services, i. e. the value of scarcity, to increase (Krutilla and Fisher 1985). A way to introduce this into equations (17) e (18) is to assume that preservation benefits grow at a rate ***a*** so that (and only for the continuous version just to simplify):

$$\mathrm{TEV} = \int_{t=0}^{T} \mathrm{TEV}^{t}\ e^{-(\rho-a)t} dt \qquad (21)$$

Generally the value assumed for ***a,*** is equal to the long-term rate of economic growth[1], because it is assumed by many as a plausible lower bound for the value **a,** should to assume.

This approach is similar to hyperbolic discounting.

Uncertainty is also an important characteristic faced by individuals who are users or potential users of ecosystem amenities. They might be uncertain as to whether a specific ecosystem amenity flow will be available for their use in the future; or whether individuals themselves will want to use some ecosystems in the future or to give future generations the opportunity to decide between conservation and development. These uncertainties will affect the welfare measures and the calculation of TEV. One has to adopt the *expected value* of the TEV instead the TEV as we did in (17) e (18).

Following Perman et all 2003 and Freeman III 2003, the measure of the Expected Value of TEV [E(TEV)] with risk aversion (and for the continuous version only to simplify) will be given by:

$$E(\mathrm{TEV}) = \int_{t=0}^{T} E(\mathrm{TEV}^{t})\ e^{-\rho t} dt - \int_{t=0}^{T} \mathrm{OV}^{t}\ e^{-\rho t} dt \qquad (22)$$

where OV^{t} is the individual's Option Value at time t that is the maximum amount the individual would be willing to pay in each period t, for an option that would guarantee the existence and the use of an ecosystem. OV is a risk aversion premium (Cicchetti and Freeman 1971) and, in principle, can be obtained directly from suitable application of the Contingent Valuation Method, although in practice its calculation is a very difficult task.

What Rate of Discount Can be Used?

What number should be assigned to the rate of discount ρ? More recently Arrow et al 1996 advocate that there are basically two approaches to choose the discount rate, the *prescriptive approach* and the *descriptive approach*. Under the former, lower future discount rates must be used (tending to zero) in contrast with the last approach that relies fully on historical market rates of return to measure the discount rate.

In practice, policymakers have, in some cases, applied lower discount rates to long-term intergenerational projects (Bazerlon and Smetters 1999) but it seems this causes time-inconsistency problems as we already mentioned. If one adopts the *descriptive approach* instead, the discount rate is equalised to the market interest rate and (17) e (18) will be re-written as follows:

$$TEV = \sum_{t=0}^{T} \frac{TEV^t}{(1+r)^t} = \int_{t=0}^{T} TEV^t \; e^{-rt} dt \qquad (23)$$

where r is the market interest rate.

However, there is a multiplicity of market interest rates. Such multiplicity is related with the existence of transaction costs, market imperfections, differences in the tax treatment of interest paid and interest received, and different risk degrees faced by individuals. The first problem is related with the effect that taxes have on income. As a consequence, discounting for welfare evaluation should be done at after-tax interest rate because an optimizing lender will equate his or her intertemporal marginal rate of substitution to the after-tax rate of return (Freeman III 2003). A second problem arises with inflation, though there is a universal agreement among economists: they all agree in using real rates instead of nominal ones. Overall, the evidence from financial markets suggests that the individual's real after-tax of interest lies in the range of 1% - 4% (Freeman III 2003). Finally, there is a third problem which is that of individual facing different interest rates, and having different portfolios of investment. How shall one deal with all these rate differences when we have a single-period welfare measure that is aggregated across individuals facing different rates of interest? The standard assumptions say to assume the risk-free market interest rate as a proxy for the relevant market interest rate, that is, the interest rate on government bonds[2].

[1] To ecologists the efficient sustainable rate is equal to the growth rate of Net National Welfare (NNW) where NNW = GDP + Normal Market Output – External Costs – Pollution Abatement – Depreciation of Created Capital – Depreciation of Natural Capital (Goodstein 1999).

[2] See Freeman III 2003 procedure for calculating aggregated values in the second-best situation, where individuals face different interest rates at the margin.

A major issue in the evaluation of ecosystem preservation has been: how to account the fact that preserving some ecosystem is likely an impediment to make an alternative economic investment that could generate a higher rate of return than the effective interest rate governing individual's intertemporal substitutions? In an ideal, theoretical world, there would be no problem to choose the discount rate because it would be equal to the market interest rate and to the investment rate of return. But, in reality, investment rates of return are, in fact, generally higher and in some cases considerably higher than market rates of interest[1]. Which of these two must be chosen?

Economists are divided. With the market interest rate individuals would be better off; but they would have been even better off if the private project would go ahead[2]. Nevertheless, where environment is involved, most non-economists (and some economists) take the view that the lower rate is the more adequate to apply geometric discount, because it attaches more weight to the interests of future generations than the higher one.

6. Conclusions and Final Remarks

In common usage, *value* means importance or desirability. To an economist, the value of an ecosystem is related to the contribution it makes to human well-being. The utilitarian approach allows value to arise in a number of ways depending on how individuals use ecosystems. Ecosystems are natural capital, generally defined as the stock of environmentally provided assets (e. g. soil, atmosphere, forests, water, wetlands, and minerals) that provide flows of natural goods and services that are directly and indirectly appropriate by the economic sector and society in general at free cost. Hence, economists have generally settled for taxonomy of total ecosystem value interpreted as a *Total Economic Value* (TEV) that distinguishes between *Direct Use Values* and *Passive (Non-use) Values*.

The economic value of an ecosystem thus relates to the TEV and is an *answer*, mostly expressed in monetary terms, to a carefully defined question in which two alternatives are being compared. Mäler stated that it was possible to build four measures of individual welfare change associated to choices involving non-market goods and consequently involving ecosystems too. If there is an improvement in utility the measures are WTP^C and WTA^E. If there is a loss the measures will be WTP^E or WTA^C.

[1] See Freeman III 2003 to a more comprehensive explanation about the implications of this issue over the discounting of benefits related with public investments in general.
[2] However this would depend on the amount of the environmental cost if the development project went ahead.

To estimate TEV one has to go through the following steps: i) to estimate the WTP/WTA for an ecosystem that produces amenities, by assessing the value individuals give to the existence of that ecosystem after real responses stated by individual; ii) to estimate TEV, using the individual aggregated value of stated WTP/WTA discounted for some rate, following the natural asset analogy.

The use of CV methods to estimate the theoretical economic measures of TEV for environmental services has been one of the most fiercely debated issues within environmental economic valuation literature over the last twenty years. The main issues of debate are the validity and reliability of CV estimates, and the inclusion or non-inclusion of passive users as a TEV component.

The CV method allows getting an aggregated welfare money measure of the value that each individual puts on the existence of an ecosystem, at the moment t. To calculate the ecosystem's value as an asset one uses, by default, the discounted utilitarianism framework by simply summing up the present value of the single-period welfare measures. Although discounted utilitarianism approach has dominated, it has proven controversial when applied to environmental valuations, being the practice of discounting, the number used to measure the rate of discount, the irreversibility and uncertainty, the main origins of that controversy.

Although there is controversy towards theoretical, methodological, and empirical aspects of the ecosystem's TEV framework one must not agree with the resigned argument that "a number is better than no number". Because when talking about economic values economists are not referring to *any* number, but to *a* number rigorous theoretically defined and carefully applied instead in order to capture the real value people put on the object of choice under certain circumstances of choice.

Better then saying, "a number is better than no number" is to say "an *economic* number is better *as a minimum reference number*, than any number at all. What we mean is that nobody is defending that this economic tool should rule the day. Economic valuation is only one of the criteria available to evaluate environmental policies and decisions. The utilitarian approach allows value to arise in a number of ways depending on how society uses ecosystems. And society includes individuals such as consumers, scientists, educators, politicians, and stakeholders, like firms or non-governmental organisations. CVM is an individual stated preference based technique used to estimate the TEV in a way consistent with the valuation of marketed goods rooted in welfare economics. So, CVM can be used together with other economic valuation methods, like Implicit Valuation Methods, Delphi Methods or Multi-criteria Analysis. But these ones may not be considered as alternative economic valuation methods to CVM because they are based on specific stakeholder preferences rather then general individual preferences: Implicit Valuation is based on the preferences of policy makers; Delphi Methods bases on the preferences of scientific experts and finally, Multi-Criteria Analysis lies on the preferences of specific interest groups.

Although environmental accounting seems to have a bright future it is by no means a simple task, non-exempted of errors and some controversy. That is why the brightness of the future of environmental accounting can only be considered as a fait accompli within the backdrop scenario of inter-disciplinarily and co-operation.

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