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Economic Valuation as a Framework Incentive to Enforce Conservation

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1. Introduction

A Protected Area (PA) is an “area of land and/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means” [1]. In this paper biological diversity means “...the dynamic network of biological, chemical, and physical interactions that sustain a community and allow it to respond to changes in environmental conditions” (in [2], p. 41). Consequently PA means biological diversity conservation, or ecosystem conservation as a whole, with their geographical, biological and different types of resilience to different types of human activities resulting pressures. By adopting this definition we will be following the Subsidiary Body on Scientific, Technical and Technological Advice of the Conference of the Parties to the Convention on Biological Diversity that indicates the importance of looking at biodiversity conservation under an ecosystem approach rather than focusing on the individual components within the ecosystems.

PA's are the groundwork of conservation policies [3-7], and therefore, traditional biological conservation management practices have been based on them [7-8] and on the Safe Minimum Standard Principle (SMS), in line with the ecological perspective of sustainability as defined by the Daily Rule [9]. The SMS principle states that a sufficient area of ecosystem must be conserved to ensure the continued provision of ecosystem services unless the social costs of conservation are unacceptably high. The Daily Rule states “never reduce the stock of natural capital below a level that generates a sustained yield unless good substitutes are currently available to the services generated” (In [9], p. 112, citing Daly H. (1996) *Beyond Growth: The Economics of Sustainable Development*. Boston: Beacon Press. 81-82).

Nevertheless, PA's based conservation policy by itself seems not to be sufficient to achieving the required conservation levels for outstanding ecosystems, with resident populations, located near urban centres and with good means of access and especially within a low or

middle income country context [10]. Over the last century there has been a growing realisation that biodiversity is being lost at an alarming rate, in spite of the national efforts to expand protected areas to incorporate a wide variety of ecosystems put into danger by the human population expansion, increasing damaging pressure placed by human activities like deforestation, pollution, poaching, or intensification of agriculture. One reason for such insufficiency is that biodiversity in general and PAs in particular, exist neither in isolation nor are independent of human activity. Over 50% of the 20,000 official PAs established over the last 200 years are on land historically occupied by indigenous peoples [8]. Furthermore, some stress that official conservation initiatives have tended to neglect some of the subsequently protected environments as the result of their long-term interaction with humans. This is surely the case of Europe, and Portugal in particular. In these regions, landscapes encompass large areas of semi-natural vegetation interspersed with grazing areas, hedgerows, farmland, and small villages and towns. Or, as in the case of wetlands, the coastline is frequently host to important urban communities depending on fishing activities and/or other marine activities and/or tourism.

Despite the richness and historical record of such human environmental interaction, earlier conservation practices have implemented an ideology primarily based on nature and some kind of “conservation ethic” potentially devaluing those values and systems that sustained human ecological practices in their respective contexts. Currently, while most of the governmental conservation agencies broadly recognise and accept the notion of “conservation-with-use” [8] by explicitly integrating it into national conservation legislation most state-declared PA local communities have not been active participants in designating or managing their surroundings. This state-imposed development and conservation, leaving local community on the peripheries of power, has created local antagonism and suspicion [8]. PA inhabitants, be they local farmers, herders or fisherman, all complain about being marginalised. When the state sets aside PAs, residents claim it does so at the cost of their own land, resources, and livelihoods, mostly without proper compensation for loss of property or forgone revenues [11]. The act of conservation is thus interpreted as a misfortune rather than an opportunity for sustainable development, accordingly to the Triple Bottom-Line methodology. This thereby leads to increasing evasion of conservation regulations while the government delays in answering the worsening conservation problems resulting from those individual actions [12,13].

Although current economic investigation is proving that enforced resource conservation measures like PA's are efficient in the sense that they do not cause Pareto inefficiency and that Pareto optimal allocations cannot only be reached through competitive markets [14], we would additionally state conservation practices have to change sharply, and must stretch beyond the SMS principle through both engaging locals and other users with the conservation process and creating a broad consensus as to the existence and objectives of conservation initiatives.

Within this framework, Co-Managed Protected Areas (CMPA) is emerging. CMPA are official state-established PAs managed with the effective engagement of other social actors,

including indigenous and local communities [8]. They are universally accepted means of community enforcement [8] with three objectives: the conservation of local natural and cultural heritage, the participation of civil society in the management process and the equitable distribution of benefits and costs. Community empowerment has to be reinforced with the implementation of incentive measures to enforce actor compliance, reinforce capacity building and provide intelligible information about the value of nature conservation (for example, see [2, 12] for a comprehensive description about incentive measures for biodiversity; or [15] as an example of how to use a travel cost approach to estimate the value of an entrance fee for a National Park). One of the means of achieving the aforementioned community empowerment is to make people aware that a PA is a sort of capital, - the natural capital -, and is as valuable as any other sort of capital goods like houses, land, art, factories, or infrastructures being able in generating flows of monetary benefits. Following [16], natural capital is generally defined as the stock of environmentally provided assets (e.g. soil, atmosphere, forests, water, wetlands, minerals) generating flows of goods and services, that are appropriate by economic sector and society at free cost. Locals must be aware that setting land aside by government for conservation purposes, instead of being a curse, may be a way of earning money and meliorating their way of living. The key idea is to ensure PA inhabitants as the owners of the natural capital, gain a strong economic interest in protecting that capital and therefore to guarantee the sustainability of the environment they live in, either for their own profit or to maintain resale value [9]. This is by no means an easy task for policy-makers and is packed with social, scientific, and practical difficulties and ambiguities [17-18]. Furthermore, biodiversity policy has its own respective complexities, differing to classical pollution problems, including heterogeneity, irreversibility, accumulation of impacts, information gaps, mix of values and pressures [2]. Since economic decisions are market-based, it may be necessary to apply economic instruments to conservation policy, like economic incentives, prices, or information concerning the monetary values of the benefits generated by the natural capital, and the monetary value of the natural capital stock itself. This is profit-based conservation and we defend its application in conjunction with the SMS principle to improve conservation policy rather than as any simplistic alternative. As several economists and ecologists have been warning, too little nature will be conserved by market force alone [13, 19].

In this chapter, we discuss the advantages and disadvantages of economic valuation of ecosystems as an incentive measure to enforce local community co-operation in conservation decisions and management. This issue is not new to economic and ecological literature. However, the literature mostly serves to demonstrate that economic valuation is anything but exhausted as an issue for discussion. Misleading pro and contra arguments and definitions are often used, both by economists and ecologists, making any understanding of monetary evaluation of biodiversity and its respective methodology more complicated than is justifiable. By recognising such difficulties, this chapter tries to assess the main points of discussion around the economic valuation of biodiversity, including the advantages and disadvantages of applying it as an incentive tool to enforce conservation

attitudes. Analysis is only provided from the economist's point of view and does not seek to be exhaustive. The chapter is organised as follows. We proceed by clarifying what "value" means to economics. In section 3, we describe the most important steps required for ascertaining the most accurate possible monetary value. In section 4, we describe why monetary valuation is an important and reliable tool in policy and conservation management despite these difficulties and controversies. Finally, conclusions are drawn.

2. The value of ecosystems: What does this mean to economics?

In common usage, "value" means importance or desirability. To an economist, an ecosystem is a non-marketed good, and therefore its value is related to the contribution it makes to human wellbeing (see for instance [20-21] to read more about the state of non-market valuation of environmental resources). Human wellbeing depends on the basic requirements for a good quality of life including freedom of choice, health, good social relations and security and may be broadly understood as happiness. Wellbeing, as experienced and perceived by humans, is situation-dependent as it reflects the local geography, culture and environmental circumstances. 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 any sort of satisfaction on humans [21, 22].

The utilitarian approach allows value to arise in a number of ways depending on how individuals use ecosystems [21, 23]. The "prior" or "primary value" consists of the system characteristics upon which all ecological functions depend (resilience capacity, individual resource stability, biodiversity retention). The value arises in the sense that the ecosystem produces other functions with value – "secondary functions", and, as such, in principle, has economic value. These secondary functions and associated values depend on the maintenance, health, existence and the operability of the ecosystem as a whole. Hence, economists have generally settled for taxonomy of total ecosystem value interpreted as a Total Economic Value [23] (TEV) that distinguishes between Direct Use Values and Passive (Non-use) Values. 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 [24]. Figure 1 provides a diagram detailing the relationship between the TEV's taxonomy of use and non-use values and the ecosystem's service concept.

Use Values include Direct Use Values and Indirect Use Values (see [2, 25-26] for a more detailed definition of the different type of uses). Direct Use Values derive from the actual use (consumptive and non-consumptive) of natural resources for: commercial or self-consumption purposes (e.g. harvesting timber, fishing, collecting herbs and minerals); tourism and recreation; education and research; aesthetic, spiritual, and cultural ends. These are the so called primary functions of the ecosystems. One of the more recognised and important sources of ecosystem's use 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,

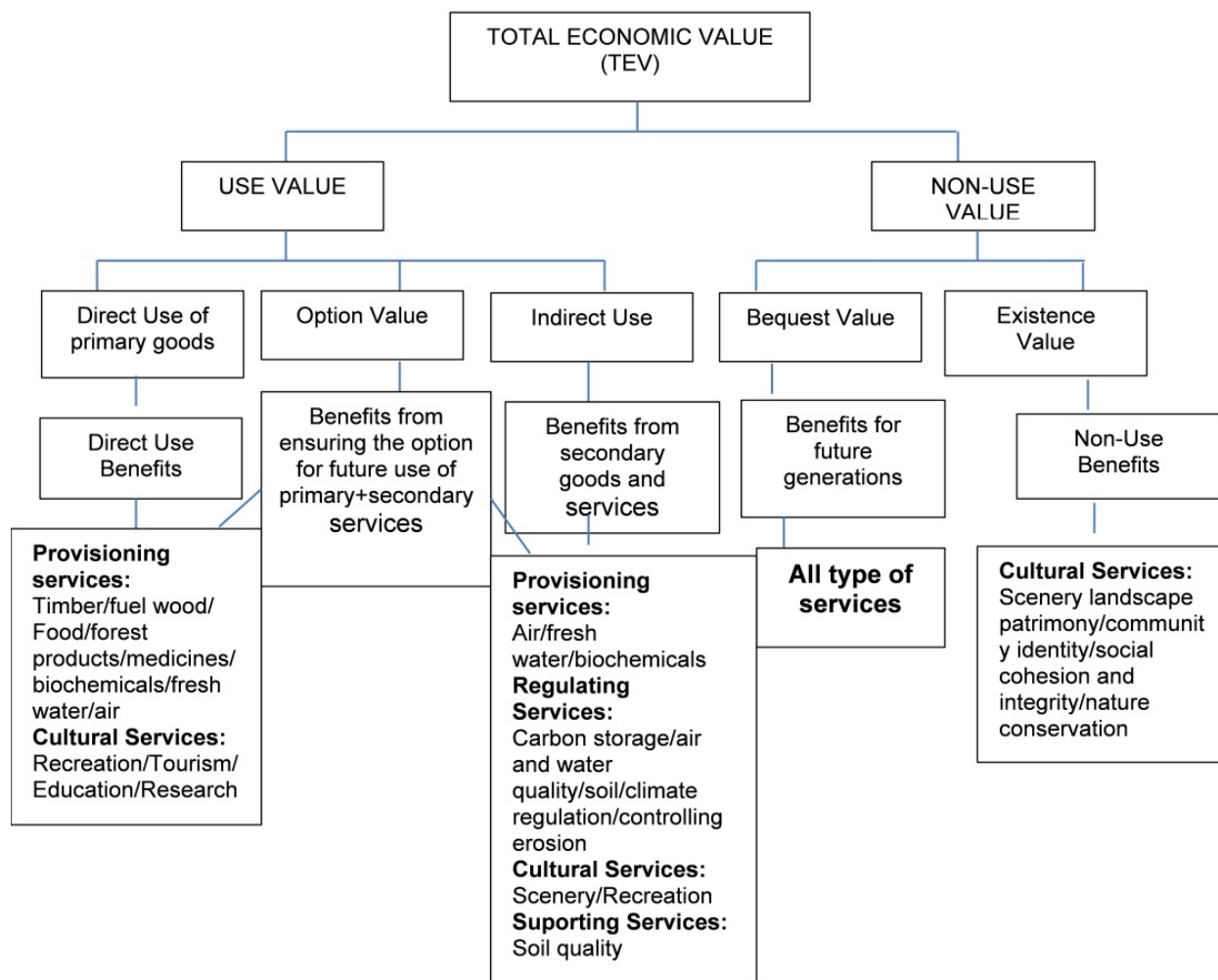


Figure 1. TEV Taxonomy and Ecosystem Services (adapted from [25])

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 the strategic, commercial and scientific value of ecosystems (see for instance [27] for more details and [18] for their interesting parabolic perspective of ecosystem and biodiversity in terms of Noah's Ark). Indirect Uses are related with the use society makes from ecosystem functions (or secondary functions) indirectly, 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). Indirect Use values may further include "Vicarious Use value" addressing the possibility that an individual may gain satisfaction indirectly, from pictures, books, or broadcasts of natural ecosystems even when not able to visit such places. Option Values are related with individual willingness to pay a premium to ensure future ecosystem availability and usage. Some authors also refer to a Quasi-Option Value which reflects the individual willingness to pay a premium to ensure the maintenance of the ecosystems and respective services to get more accurate scientific information in the future. This type of benefit relates with individual perceptions concerning the

irreplaceability degree of some ecosystems and with the lack of information about its functioning. Given this type of uncertainty, some individuals prefer to conserve ecosystems instead of destroying them until the moment society will have more accurate information about their real ecological value. Passive (or 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. Finally, Bequest Value considers individual willingness to pay a premium to ensure that their heirs will be able to use the ecosystem in the future.

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 (for a more comprehensive understanding of this debate see for instance [23]). Nevertheless there is actually a growing trend towards using the TEV measure on the grounds that theoretically there is no need to adopt a dichotomy that involves the adoption of arbitrary assumptions. See [28- 29] for a more comprehensive understanding of the total and non-use values discussion. Advances in ecological economic models and theory also seem to stress the value of the overall system as opposed to individual system components only. Currently, ecosystems or environmental resources as a whole are increasingly recognised as assets providing sets of 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 [20-21]. This point to the value of the system itself 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 [23].

The economic value of an ecosystem to some individual, thus relates to TEV that individual puts on the ecosystem. Individual's TEV is not an absolute value because economics provides valuations only in comparative terms. When economists say they are valuing an ecosystem, they are really defining a trade-off between two situations involving a change: e.g. maintenance or non-maintenance of the ecosystem. The economic value of the ecosystem is the amount an individual would pay or be paid to be as well off with the ecosystem or without it [30-31]. Economic value is therefore an answer, mostly expressed in monetary terms (although not necessarily), to a carefully defined question in which two alternatives are being compared. The answer (the value) is very dependent on the factors incorporated in that choice: the object and the circumstances of choice [32]. Economics defines objects of choice as any tangible or non-tangible object, process or activity that can be described as allowing choice and 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 is 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 (to accept or not the political option to conserve the ecosystem). It is clearly fundamental 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) specify the rights of assignment; iii) define the mechanism

of choice, that is the manner through which the individual will exercise choice: by voting, through private market transactions or other unspecified behaviours.

The object and circumstances surrounding such choice define its context. In the case of ecosystems, value depends on the ecosystem location and the level of human presence, the actual or threatened level of degradation as well as the degree to which natural services provided can be substituted by other substitute ecosystems. This substitutability is a highly important concept within economic valuation as objects with significant numbers of close substitutes are not rated as valuable as those with few or even no substitutes. In the case of ecosystems, the degree of substitutability is relative depending on factors including the scale and level of aggregation and the time-scale involved. 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 were first proposed by Mäler [33-34] as an extension of the standard theory of welfare measurement related to market price changes formulated by Hicks [35]. 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 (for instance, a conservation policy) generates an improvement in individual well-being (that is 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 (for instance, some ecosystem destruction, related with a particular human activity) in well-being (that is, 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 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 wellbeing; or WTP^E/WTA^C where the individual faces deterioration in well-being.

Mäler used the following basic model of individual utility to define welfare measures. 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 , and for non-market environmental amenities denoted q . 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 . 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 $\mathbf{x}$ and $\mathbf{q}$. 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 being a merely empirical question [36]. The individual faces a budget constraint based on their disposable income m , and the prices of market commodities, $\mathbf{p}$. The individual maximisation utility problem of decision is then formalised as:

$$\begin{aligned} & \max_{\mathbf{x}^*} U(\mathbf{x}, \mathbf{q}) \\ & \text{subject to } \sum p_i x_i = m \end{aligned} \quad (1)$$

meaning, each individual is trying to decide what is the affordable quantity of good X (x^*) that must be purchased in order to get the maximum level of utility (or satisfaction), given his/her per capita income, the prices of the good, and the level of amenity q , all remaining constant. The solution of problem (1) yields a set of ordinary or marshallian demand functions for $\mathbf{x}$ denoted $x_i = g_i(p, q, m)$, for $i = 1, \dots, N$ individual and an indirect utility function denoted $U(\mathbf{x}, \mathbf{q}) = \varphi(p, q, m) = U[g_i(p, q, m); q]$. Each individual marshallian demand function expresses the relationship between the changes on the individual purchases of good X as its price rises, holding individual income and the amenity constant; and each indirect utility function expresses the changes on each individual utility (or satisfaction), for different purchases of the good X . The dual problem of each individual i is an expenditure minimisation model defined by:

$$\begin{aligned} & \min_{x_c} \sum_i p_i x_i \\ & \text{subject to } U(\mathbf{x}, \mathbf{q}) = U \end{aligned} \quad (2)$$

The dual problem states that individual i wants to make the lowest possible expenditure (denoted $p_i x_i$) with X , to maintain his/her utility or satisfaction at a particular level denoted U , for all level of prices. The solution of the dual problem (2) yields a set of compensated or Hicksian demand functions for $\mathbf{x}$ denoted $x_{ic} = h_i(p, q, U)$, and an expenditure function (3):

$$m = e(p, q, U) = \sum_i p_i h_i(p, q, U). \quad (3)$$

Each compensated demand function shows how the quantity demanded of good X changes as the price rises, holding utility and the amenity q unchangeable. The expenditure function shows the relationship between the minimal expenditures with the good X necessary to achieve the specific level of utility.

Let us now suppose $\mathbf{q}$ is going to change from the current state $\mathbf{q}^0$ to another different state $\mathbf{q}^1$ so that the individual i will have to choose between the state $\mathbf{q}^0$ or the state $\mathbf{q}^1$, while holding constant the individual per capita income m and the prices $\mathbf{p}$. If he or she chooses $\mathbf{q}^0$, the level of utility is given by $U^0 = U^0(p, \mathbf{q}^0, m)$; if he or she chooses $\mathbf{q}^1$ the utility is given by $U^1 = U^1(p, \mathbf{q}^1, m)$. The welfare change associated with the change of the utility level from U^0 to U^1 can be measured using Mäler's Compensation Variation (CV) or Equivalent

Variation (EV) measures, defined respectively by $CV = e(p, q^0, U^0) - e(p, q^1, U^0)$ and $EV = e(p, q^0, U^1) - e(p, q^1, U^1)$, as they are illustrated in Figure 2.

Utility change Hicksian measures	Ecosystem improvements $q^0 < q^1 \Rightarrow U^0 < U^1$	Ecosystem degradation or destruction $q^0 > q^1 \Rightarrow U^0 > U^1$
CV	WTP ^C	WTA ^C
EV	WTA ^E	WTP ^E

Figure 2. The relationship between the Hicksian CV and EV utility measures and the WTP and WTA Malher's utility measures

If $U^1 > U^0$, that is if there is an improvement of the individual's utility or level of satisfaction associated with an improvement of the ecosystem, utility CV measure are WTP^C to secure that change, or EV measure WTA^E to forgo it. If $U^1 < U^0$, that is if there is a decrease of the individual's utility or of the level of satisfaction associated with the depreciation or destruction of the ecosystem, utility CV measure are WTA^C to tolerate the damage, and EV measure WTP^E to avoid that change. Given the duality between the indirect utility function and the expenditure function, the fundamental monetary measures are, therefore, given by equations (4) and (5), respectively:

$$WTP^C \text{ or } WTA^C = CV = e(p, q^1, U^0) - e(p, q^0, U^0) = \int_{q^0}^{q^1} \frac{\partial e(p, q, U^0)}{\partial q} \quad (4)$$

$$WTP^E \text{ or } WTA^E = EV = e(p, q^1, U^1) - e(p, q^0, U^1) = \int_{q^0}^{q^1} \frac{\partial e(p, q, U^1)}{\partial q} \quad (5)$$

In short, when environmental change is positive and individual's utility rises, he/she will be able to pay a maximum amount of money given by WTP^C to secure the environmental improvement, or he/she will accept a minimum amount of money given by WTP^E to forego the environmental improvement. When environmental change is negative and individual's utility declines, WTA^C is the minimum amount of money that must be given to the individual to compensate him from the environmental damage effects and WTP^E is the maximum amount of money the consumer has to pay to stop the implementation of the economic decision that will be the cause of the environmental damage.

WTP (equivalent or compensated) and WTA (equivalent or compensated) may or may not produce differing monetary information for the same object under valuation. The difference between the two measures is explained by the income elasticity (that is the sensitivity of the quantity demanded at a given price to income changes) and the budget share of the ecosystem. The smaller the income elasticity or the smaller the budget share, the closer are the measures to each other. WTP is equal to WTA where there is no income effect and individual is neutral to losses and gains, and where there are perfect substitutes of the object under valuation. Generally, WTA is greater than WTP which stems from the different welfare measure definitions and contexts of choice: as proved by numerous empirical

estimations of WTP/WTa, WTa is always greater than the WTP because the latter is limited by the individual's budget restriction and by the existence or non-existence of substitutes, while the former is not. See [37-38] for a more comprehensive discussion about the WTP and WTa differences and the consequences they have on the valuation of environmental services.

For purposes of economic valuation, the ecosystem and its components (q) are considered to be public or quasi-public reproducible natural assets, like structures or equipment. By the fact of being natural assets, they have direct and indirect use values, and passive uses as well, including option, quasi-option and existence values. As people experience satisfaction (utility) with the existence and services of ecosystems, the value of that reproducible natural asset is necessarily linked to its generated value flows of services and passive uses (it is assumed that the satisfaction or utility $U(q^1)$ generated by the new state of the ecosystem is a flow that will occur during a particular time period of time). Hence, according to the asset analogy, TEV is equal to the discounted sum of values (or benefits, or utilities) of those services and passive use benefit flows, so that:

$$TEV = \sum_{t=0}^T \frac{TEV^t}{(1 + \rho)^t} \quad (6)$$

where T is the relevant period of time during which the ecosystem generates the flow of amenities; TEV^t is the mean WTP (or WTa) each individual wants to pay for the amenity's improvement (or be compensated for the destruction of the amenities) estimated for the relevant population (that is, for the individuals that are going to suffer the effects of the amenity's change) at a t point period of time period T; and ρ is the discount rate used to discount the cash flows provided by the ecosystem during the T time period considered.

3. Estimating ecosystems TEV

WTP and WTa are though the fundamental monetary measures of value in economics for market and non-market commodities. When economists set about estimating the individual's WTP/WTa (or the value) for the market goods and services, they use actual, observed, market-based information, because preferences for private goods are revealed directly when individuals purchase them on the market. However, because ecosystems and most of the services they produce, like the ones associated with indirect use, option or existence values, are not market tradable, we thus have to elicit individual preferences directly in terms of WTP/WTa by use of questionnaires, such as Contingent Valuation (CV). CV provides the means to estimate natural resource value or loss and is the only current method that produces estimations of the ecosystem's TEV, including all sort of the marketed and non-marketed ecosystem services. For a detailed description of the Contingent Valuation method, see for instance [39] or, more recently, [40]. If you want to apply the method in order to estimate the value of some ecosystem or the value individuals put on some ecosystem's changes, you may consult [39-41]. For a more detailed description of other economic valuation methods, see for instance [38, 42-43]. So, to estimate (6), one has to go

through the following steps: i) to estimate WTP/WTa and TEV via actual individual responses gathered by the CV method; ii) to estimate TEV, using aggregate values of stated WTP/WTa discounted by a certain rate, in accordance with the natural asset analogy.

3.1. The use of the CV method and estimate reliability

The use of CV methods to estimate TEV for environmental services has been one of the most fiercely debated issues within environmental economic valuation literature until the 90's of the last century, more specifically the validity and reliability of CV estimates issue and the inclusion or non-inclusion of passive users as a TEV component [44,16]. Detractors argue that respondents provide answers inconsistent with the basic assumptions of utilitarian rational choice; they question the seriousness of CV answers because the results of surveys are not binding. And a more extreme position even holds that the economic concept of value itself has no link to reality. This has led to the supposition that responses might bias CV value away from theoretical welfare measures. Defenders acknowledge that early applications suffered from many of the problems critics have noted [39]. However, recognition is required of how more recent and more comprehensive studies have dealt and continue to deal with those objections.

The key to a successful evaluation method is that it must be assessed in terms of how closely it represents an accurate measurement of the real value. The closer the real values are to the estimated, the more accurate the valuation method is. If WTP/WTa an ecosystem 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. See [39-40] for a more comprehensive description of these methodological CV problems; and their potential effects upon estimates. See also [16] for a comprehensive survey of literature on such issues.

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. The bias between the theoretical, and the CV estimated WTP/WTa, grows where the latter tends to systematically diverge from the former. The concept validity relates to the CV application process that involves numerous issues that must be resolved mainly based on individual judgement of the CV implementing entity. Because WTP is not observed, inferences as to validity are based on indirect evidence related both with content validity of CV study design and execution, and construct validity dealing with the degree to which the estimated money measure relates to other theoretical measures. To assess the content validity involves examining study procedure content. This involves four steps. Firstly, the researcher defines the scenario that would lead a theoretical consumer to reveal his or her WTP/WTa. More precisely, this is the CV phase where the elements of choice or details of the transaction are presented to the respondent. The transaction must be adequately defined in order to be clearly understood by the participant. The second step is

vital to controlling the extent to which participants really understand the proposed transaction as communicated through the scenario defined in the first step. This obliges the introduction of qualitative research procedures to support CV survey design. The third and fourth steps refer to the appropriate statistical and econometric techniques that must be applied to elicit unbiased, higher content validity estimates of WTP/WTAs. Contributions towards improving CV studies, from both the theoretical and empirical perspective, have been drawn from the differing fields of academic social science research – economics, psychology, law and politics (see [32] for a more comprehensive study of this issue) - but the most important was that of NOAA Panel Report [45]. Recognising the impossibility of externally validating estimates produced by CV studies, the NOAA Panel recommended researchers adopt an *ex ante* analysis of the results in place of an *ex post* analysis, by focusing discussion on how to improve the theoretical and empirical quality of studies thereby improving the accuracy of CV valuation by strengthening result reliability. The Panel guidelines for the study design phase are set out defined for three aspects of the transaction: the good, the payment, and the valuation context. In the case of ecosystem valuation surveys, respondents need to know about the attributes of the ecosystem [46-48], the level of provision of those environmental attributes “with and without intervention” and if there are undamaged substitute commodities. As for payment, the Panel recommended the use of the WTP valuation format and the definition of the “payment vehicle” which may include taxes, property taxes, sales taxes, and entrance fees, changes in the market prices of goods and services or donations to special funds. As for the transaction context, it is important 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 use for such provision (e.g. majority vote, individual payment). Researchers must allow respondents the opportunity not to vote. The Panel recommends a conservative survey design, a referendum style choice, and voting choices followed by open-ended questions asking about reasons for voting one way or another. The Panel’s recommendations on survey design are for the most part almost standard practice except: i) the use of a referendum format in substitution of the open ended question eliciting the maximum WTP; ii) and the opportunity to the respondent to choose not to vote. These Panel guidelines are standard practice for any high quality survey. It is recommended to use probability sampling, in-person interviewing, to minimise non-responses, to make careful pre-testing and to examine interviewer effects.

By recognising the impossibility of independent verification of the CV results, the Panel suggested that besides the survey design and administration guidelines, an alternative test of CV reliability must be drawn from the economic theory of rational choice to monitor the rationality of individual WTP responses. This test is called the scope (embedding) test and requires that the stated survey WTP should be related to the size of the object of choice. If the WTP response is inadequate to the scope of the object of choice then the findings of the CV survey are unreliable. The scope test is based on the weakest form of rationality among individual choices. It is reasonable to suppose that more of a good is always better to the individual if not satiated and that he/she is willing to pay more for more of that good. Also, it is reasonable to assume that WTP will decline although not abruptly for additional

amounts of the good [45]. The NOAA Panel concluded that the information provided by CV surveys, where in full compliance with the recommendations, can be considered “as reliable by the standards that seem to be implicit in similar contexts, like market analysis for new and innovative products, and the assessment of other damages normally allowed in court proceedings” [45].

More recently there has been a trend to include expertise from other disciplines such as marketing research, survey research, cognitive and social psychology (see [46] for a comprehensive introduction to the psychological perspective on economic valuation to improve the CV methodology) both from the theoretical and empirical point of views. To read more about psychologist’s criticism of the utilitarian approach to ecosystem valuation, see for instance [49] , where the problem related with the existence of lexicographic preferences is discussed, and [50] , for a review of the some empirical evidence of such preferences for environmental goods. The importance of all these contributions to survey research is almost intuitive because CV is broadly survey valuation method based.

3.2. Estimating TEV using individual stated WTP

As mentioned earlier in the chapter, the capital asset pricing approach views the ecosystem value as an asset value (natural and reproducible) at a particular point in time as the discounted value of all future services the ecosystem will provide as stated by equation (6). A common economic approach is to assume a rate of discount and further assume that the flows of services provided by the ecosystem during the relevant period of time T will be constant, and that the value of flows will increase in line with the expected rate of inflation. Under these assumptions, one is left with the task of valuing the services at some point in time t discounted at some discount rate, ρ . The problem here is what rate of discount is to be chosen and what flows are to be considered.

The answers to either question are not straightforward to economists because the valuation of ecosystems or the decisions involving nature related sustainability decisions are characterised by several dimensions in which issues are more demanding for economists than those raised by environmental economics. One is the time dimension. The long-term period T applied to ecosystems, not inferior to 50 years and sometimes as long as one, two or even several centuries, is much longer than normally considered in economic analysis, which poses a particular challenge for the economist’s traditional practice of discounting.

A second dimension is related with uncertainty. Over such a long time scale, it is logical to expect that individual preferences will change due to technical and social changes that also affect the flow of individual WTP across time. On the other hand, society’s ecosystem services demand is already so great that trade-offs among competing services have become a rule. This increasing demand is compounded by the increasingly serious degradation of ecosystem capacities to provide services, seriously diminishing the prospects for sustainable human development [51]. It is though important to incorporate the ecosystem TEV into this

pressing trend for scarcity. As one cannot ask people to be futurists as to their future WTP patterns, one way of achieving this is to use appropriate discount rates and discount methods that allow for incorporating growing future ecosystem scarcity. The default criterion that has been used for ranking environmental conservation projects is provided for by the discounted utilitarian approach introduced by Bentham in the nineteenth century, and has been dominating more due a lack of convincing alternatives rather than any intrinsic accuracy. According to it, one must choose the greatest present discounted value of net benefits. The project must be rejected where it provides a negative net benefit. A positive utility discount rate ρ reflects what economists refer to as a positive time preference [52], a widespread desire to consume today rather than save for the future. This is a way by which the market penalises investments with long-term payoffs: any investment with high up-front costs and a long stream of future benefits will dramatically undercount future benefits. This is precisely what happens when one has to value ecosystems by discounting the flow of the services they provide over time: at any positive discount rate, the present value of any ecosystem is almost irrelevant and it thus becomes irrational (from the economic point of view) to be concerned about extinction or conservation. And yet societies obviously are worried about such issues and actively continue to consider how to devote substantial and scarce financial resources to them.

How important then is our concern as economists about the time dimension to ecosystem valuation frameworks? The opinion of some authors like Ramsey [53] or Harrod [54] (the original proponents of modern dynamic economics), and that of Heal [55] on environmental subjects, is that discounting is ethically indefensible (see chap. V in [18], for a comprehensive theoretical approach to this issue). There is however empirical evidence suggesting the legitimacy of discounting future utilities even where it is not certain that discounting future utilities in the evaluation development programmes is the same as discounting benefits of values [55]). Where empirically proven that people consider some positive discount rate we have to conclude that the traditional discounted utilitarianism approach to value benefits is not particularly suitable to clearly (as far as possible) capturing individual future concerns over the future environment. This is because decision-makers and cost-benefit applying researchers generally use market rates of discount much higher than the appropriate efficient sustainable rate thus depreciating future flows. As a consequence of this controversy, researchers have, in some cases, begun to apply lower discount rates to long-term, intergenerational projects [56]. Others use a declining rate in the future. Unfortunately, both methods result in time-inconsistency problems as long-term projects in the present become near-term projects in the future (see [18] for a more comprehensive discussion of this issue). More recently, attempts have been made to include the uncertainty and its persistence dimensions into the discount rate discussion that seems to dramatically increase the expected net present value of future payoffs [57]. We may conclude that in any calculation of TEV as defined in equation (6), we have to choose the appropriate relevant period of time, discount rate, and method of discounting in order to reflect intergenerational preferences and uncertainty into the valuation process.

4. So just what is ecosystem TEV useful for?

A clear answer to this question suggests it be broken down into other two questions. Firstly, given the existence of controversy towards theoretical, methodological, and empirical aspects of ecosystem TEV, what is the usefulness of a monetary measure for ecosystem conservation decisions? Secondly, where the economic value is a theoretical, abstract measure experiencing a somewhat exacerbated controversy, what is the point of estimating it and using it for conservation issues? Let us begin with the first.

When government classifies some region as a protected area, biological resources of that region may be put under threat because the responsibility of management is transferred from people who live inside or close to the protected area to governmental agencies, located far away from the region. However, the direct costs of protection typically fall on the inhabitants who otherwise might have benefited from exploiting the Nature. Thus, there is a problem of justice and of intergenerational equity, which may have severe consequences if inhabitants have an economic disadvantage and, as a consequence, environmental regulations may be easily evaded or avoided.

National governments and local administration always faced difficulties in protecting their natural areas because the implementation and management of protection and conservation programmes are expensive tasks. Although society has been demonstrating the value it places upon intergenerational environmental transfers, by accepting to bear the opportunity cost of setting aside land and resources in their natural or almost natural state only for scientific, educational, recreation, and biodiversity protection or sustainability purposes, protected areas generally have been facing budget problems, and insufficient personnel. The aforementioned financial shortages, together with rising economic pressure over environment, are on the alert to the necessity of adopting new mechanisms to improve Nature protection sustainability. Economists, biologists and management conservationists they all increasingly recognise the urgent need for alternative approaches to Nature management to introduce incentives or stimulus specifically used to incite and/or motivate economic, social and administrative agents (the stakeholders) to comply with Nature conservation strategies, to divert resources (e. g. land, capital, and labour) towards Nature conservation, and to incentive the participation of those groups and agents that work in or live within protected areas, to adopt sustainable development options.

Market-based incentives (including ecosystem valuation) are considered as the more adequate to perform the task. The basic underlying idea is that Nature has to be incorporated into the market system because Nature exhibit quasi-pure public good characteristics, an open access's free rider problem, together with lack or insufficiently stated property rights [12,17,58-59]. Because of these market failures individual users have too little incentive to conserve species and ecosystems because they earn immediate, high monetary benefits from exploiting Nature, without paying the full social and economic costs of its depletion. Instead, individual users transfer these costs to society to be paid either now or in the future. Therefore, although users may benefit collectively from managing Nature in

a sustainable way on a protected area based conservation management scheme, from the individual point of view he/she can be better off by cheating and increasing their own use. As for the property rights problem it often happens that over exploited environmental resource and landscape tend to be the ones with the weakest or even non-existent ownership. However, in more developed countries like some Europeans, it often happens that biodiversity has private owners. But in the absence of markets, the private owner of the natural resource is not enabled to capture the benefits of Nature conservation. Hence, the private rational option will be to exploit the resources to extinction or its irreversible destruction. This bunch of market failures and its negative consequences upon environment explain why an effective and sustained government intervention is required to meet the values and needs society expects to have from conservation, all together with market forces [12-13, 19]. By using market-based incentives like fees, rewards, fines, compensations for ecosystem services, subsidies, and loans (or credit), land banks, revolving funds or daily wages, financing stimuli, voluntary action stimuli, and involvement of government authorities will be more incentivised. Therefore an increasing of compliance with the Nature conservation legislation is to be expected: firstly, because they may give local communities the financial means to develop sustainable activities; secondly because they must reduce economic pressure on environmental rich lands, by incentive locals to devote them instead to biological conservation and to concentrate economic and social activities on less biological rich lands; thirdly because they must conserve traditional knowledge and cultural systems; and fourthly because they must compensate locals for possible foregone income associated with the protection option.

As for the ecosystem's economic valuation, many economists and non-economists consider it as a very important conservation tool, an incentive measure, a support for conservation decision-making, and one of the ways to ensure that the private profitability gap between sustainable and unsustainable use of ecosystem services is narrowed or even closed [2, 21]. They see ecosystem valuation as a support for improving political and judicial decision-making within contexts of accruing environmental degradation, especially when compounded by increasing social demand for environmental services. Ecosystem valuation can also play a beneficial role in government land use, conservation, and tax planning, particularly when there is a growing interest in incentive or compelled conservation by private owners and property developers [60]. The basic economic decision methodology more commonly used to make cost-effective choices between different investment alternatives, the Cost-Benefit analysis, has often been under-utilised (or non-utilised at all) for environmental purposes because the monetary value of ecosystems is not took into account as an important variable. The absence of recognition towards the monetary value of ecosystem's and the services they provide to society is a common flaw of the decision process which happens because the existence of that value itself is not easily recognised by the stakeholders; besides, ecosystem's valuation is very demanding from the technical and the financial points of view. Further, economic ecosystem's valuation is important, as economic conservation instruments based on cash payments or tax breaks require the estimation of enough compensation to offset habitat losses elsewhere. Additionally,

evaluation will be particularly important whenever ecological assets are not traded, or when it is necessary to make a comparison between ecosystems with different characteristics and, therefore, different values (ecosystem's value depends crucially on: the ecosystem location; its relationship with human activities and expected ecosystem changes over time), bringing confidence to ecosystem trading schemes in terms of the maximisation of net social benefits [61].

Strong theoretical, empirical, and jurisdictional progress has been made in the last three decades in the USA and Europe [62-64, 2], concerning the estimation and appliance of the economic values estimations to Nature conservation. Recently, one the more important efforts to improve the use of ecosystem valuation as a conservative management tool, was borne at the Heiligendamm Summit on 6-8 June 2007, where the G8 countries and the 5 major leaders newly industrializing countries endorsed a proposal suggested by the German government during the meeting that took place in Potsdamin on 20 March 2007 to make a study to analyze the global economic benefit of biological diversity, the costs of the loss of biodiversity and the failure to take protective measures versus the costs of effective conservation (the Potsdam Initiative for biodiversity). The study called The Economics of Ecosysems and Biodiversity (TEEB), was appointed to Pavan Sukhdev, and the preparatory work was initiated by the German Federal Ministry for the Environment and the European Commission, with the support of several other partners; hosted by UNEP, several countries and organizations have joined TEEB. Several reports were already produced (<http://www.teebweb.org/InformationMaterial/TEEBReports/tabid/1278/Default.aspx>) concerning the evaluation of costs of the loss and decline of Nature worldwide and their comparison with the costs of effective conservation and sustainable use. The main issue of the study, consisted in making visible the value of ecosystems and services to facilitate development of cost-effective policies [22].

There isn't, however, a one-hundred per cent consensus about the relevance of using ecosystem valuation as an incentive tool in conservation policies. Some economists argue that evaluation is neither necessary nor sufficient for conservation defending the key theme of conservation as making this policy more attractive than any alternative usage through translating the social benefits of ecosystems into income [17]. In their opinion, this does not necessarily oblige evaluation. However, one can question this opinion by asking how one can translate those ecosystem social benefits into income efficiently, in the absence of markets, invisibility of the ecosystem's monetary values, within contexts of growing financial resource scarcity, social equity promotion, and efficiency concerns? It clearly seems that valuation is the foundation to achieve that issue. Further, economic valuation constitutes a useful tool for educating and involving local populations and stakeholders by highlighting the connection between the ecosystems' underlying biophysical properties and benefits associated with the active or passive use of its services. It enables the justification of conservation projects encouraging local inhabitants and stakeholders to accept and to comply. If local people are conveniently alerted to the true economic value of their land, including scarcity, they will anticipate a higher future value and hold back from economic activities that may not be compatible with conservation. This behaviour will be smoothly

endogenised by local inhabitants and stakeholders where accompanied by decisions that transform conservation decisions into sustainable income for local communities. In sum, environmental valuation can be seen as an economic conservation instrument that promotes ethics and fairness in conservation policies. Through environmental valuation and improved economic compensation mechanisms, society will reinforce its right to defend social ecosystem services even when such a defence may sometimes imposes severe economic and social restrictions on the use of land that belongs to other people independent of their social-economic development expectations.

Being such an important instrument for convincing people to comply and participate in conservation policy, there remains ongoing debate and a persistent reluctance as to the reliability, and validity of the monetary measures like the one defined by equation (6). Philosophers, ecologists and even some economists not subscribing to the idea of using environmental economic values as conservation tools have their own ethical and philosophical, technical and methodological arguments. It is unquestionable that ecosystem's TEV is an abstract, theoretical measure and does not measure the real value (but what is really the real value of something, one may questioning) of the thing as commonly understood. Economists tend to structure their thinking around models of perfect rational agents, with fixed preferences, making decisions in order to maximise certain well defined individual objectives, and this is very different from the paradigms characterising the natural sciences or even other social sciences (see [65] where both economic and ecological meaning of value, are compared). As a consequence of this hyper-abstract economic model of thinking, economic value is a *relative* not an *absolute* measure. It is the answer to a question involving a choice: how much an individual is willing to pay (to accept) for a unit more (or to forego) of an environmental resource without changing his or her current (future) wellbeing. To say one thing has greater economic value than another is an alternative way of saying that, under the circumstances, this would be chosen in preference to that. A correct understanding of what the economic value concept really means helps in understanding the value paradox (the paradox value is one of the arguments used by psychologists to demonstrate the incoherence of the economic monetary measure and its valuation shortcomings): articles that command great prices are often things that common people consider as being of little intrinsic or useful value, take diamonds or paintings for example. On the other hand, absolutely essential goods are often available at a negligible, or even no price, like water or landscape, at least in regions where such natural resources are abundant. There is, however, no inconsistency where goods that are, overall, immeasurably useful being worth less than the non-essential as economists determine marginal values: a good is worth more than another when the individual is not yet satiated and it is more difficult to obtain an additional unit than another unit of something else. People generally consider a diamond much more valuable than a litre of water, even while knowing the latter's vital importance, because they recognise the rarity of diamonds and the relative abundance of water. In short, the economic money measure of value basically reflects the scarcity of the good being valued under certain circumstances but not value from a common sense perspective. The economic money measure of value also reflects important issues that

affect scarcity as perceived by the individual such as the existence of substitutes as well as the context surrounding the object of choice. In short, one may say that the theoretical, abstract weaknesses of economic valuation, identified by some commentators, is simultaneously its strength when used to improve decisions involving scarce non-market transacted resources, like ecosystems. Thus, while the level of abstraction and technical rigour of the theory underlying the definition of economic value are seen by some as handicaps, others consider them as trump-cards when there is a need for credible support to political and judicial decisions. However, these are complex methodological problems and the high level of economic and econometric expertise needed to estimate ecosystem values lies at the source of another set of criticisms. It is also a very expensive process. Nevertheless, these technical, methodological, or budgetary contrasts must not be used as arguments to turn our backs on ecosystem valuation although they must be taken into account during the political decision-making process when conservation decisions involve very important ecosystems at the local, national, or regional levels.

In short one must not agree with the argument any number is better than no number, used by some proponents, as it denotes an indefensible, very resigned attitude. When talking about economic values, economists are not referring to *any* number but to *a* number rigorously and theoretically defined and carefully applied in order to capture the value people put on the object of choice under certain circumstances of choice. Rather than saying any number is better than no number is to say an economic number is better as a minimum reference number, than any number at all.

5. Conclusions

In this chapter, we have discussed the advantages and disadvantages of ecosystem economic valuation as an incentive measure for enforcing local community co-operation in conservation decisions and management. The utilitarian approach allows value to arise in a number of ways depending on individual use of ecosystems. Hence, economists have generally settled for taxonomy of total ecosystem value interpreted as Total Economic Value (TEV) that distinguishes between Direct Use Values and Passive (Non-use) Values. TEV is a relative value and an answer mostly expressed in monetary terms to a carefully defined question in which two alternatives are compared. This answer depends on elements of choice defining the prevailing context. As ecosystems are not purchased on markets, one has to elicit individual preferences directly by the use of questionnaires such as Contingent Valuation (CV) to assess the individual's WTP and WTA relevant monetary measures. Complex criteria and "rules of evidence" such as those suggested by the NOAA's Panel must be applied to guarantee the reliability and validity of the CV data. To calculate TEV based on individual CV data and the asset analogy, time and uncertainty have to be considered when discounting service flows. One concludes that TEV may be a useful tool as an incentive, a support for decision-making, and as a tool for education and information. The fact of being a very abstract instrument, and very demanding from the theoretical and technical points of view, becomes an advantage. To date, it is still the only existing, carefully

defined and applied and somehow reliable way of society knowing how much an ecosystem is worth within a market-based scenario.

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