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Environmental values, valuation methods, and natural disaster damage assessment

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Abstract

It is largely agreed that successful development depends on the rational use of natural capital (World Bank, 1998). In recent years, advances have been made to measuring progress toward "sustainable development" (Kunte *et al.*, 1998), and in applying valuation techniques to the analysis of the environmental impacts of investment projects and public policies, both in developed and developing countries (Barbier, 1998).

Natural capital is not exclusively endangered by human actions (or inactions). Environmental (quantity or quality) changes may also be induced by natural hazards which, besides altering the natural capital's intrinsic "productivity", may negatively affect people's "ability to exploit" environmental attributes.

In 1999, the UN Economic Commission for Latin America and the Caribbean (ECLAC) has published a *Manual for Estimating the Socio-Economic Effects of Natural Disasters* (ECLAC, 1991) which intends to be a tool aimed at professionals engaged in the valuation of natural disasters' socio-economic impacts.

ECLAC has recently undertaken a revision process, aimed at expanding the scope of the *Manual*. One of the objectives of this interdisciplinary upgrading process is to incorporate "environmental values" into disaster damage assessment.

The aim of this paper is to contribute to this process, by (i) illustrating the concept of environmental values from an economic perspective, (ii) providing an overview of valuation methods aimed at

quantifying these values in monetary terms, and (iii) making a preliminary attempt to identify a strategy for integrating environmental damages into natural disaster impact assessment.

The paper is organized as follows. The Section I provides a description of the economic concept of environmental values, the predominant taxonomies of environmental services underlying these values, and the general features of the valuation approaches proposed by economic literature. We end this Section by proposing a "valuation spectrum" which could be used as a sort of conceptual device for identifying the linkages between environmental values and valuation techniques.

In Section II we provide an overview of these techniques. After illustrating their rationale, potential advantages and caveats, we focus on the "environmental value transfer method" which, because of informational and resource constraints, appears to be a natural candidate for *post*-disaster damage assessment.

In Section III we focus on the incorporation of environmental values into natural disaster damage assessment, by illustrating its rationale, the potential intersections between environmental values and the ECLAC-Manual's damage categories, and some conceptual and methodological issues which need to be carefully addressed when trying to incorporate environmental values.

Section IV offers concluding remarks.

I. Environmental values and valuation approaches

1. The economic concept of environmental value

Although there are various interpretations of the term "environmental value", economists have primarily concentrated on monetary value, as expressed via stated or revealed individual preferences.

As synthesized by Pearce and Turner (1990), economic value is not an intrinsic quality of anything: it only occurs because of the interaction between a subject and an object. It follows that environmental attributes have value only if they enter at least one individual's utility function or a firm's production function. Attributes failing either of these criteria have no economic value (Hanley and Spash, 1993).

The main rationale behind measuring, in monetary terms, the costs (or benefits) of environmental (quantity or quality) changes, is to make them commensurable with other market values. In other words, the translation of individual preferences into monetary values is generally interpreted and recommended as an operational device for *ex ante* valuation ("cost-benefit analysis") of alternative courses of action entailing both expected environmental changes and alterations in the allocation patterns of other economic goods. Or to evaluate (*ex post*) the welfare impacts of actual environmental changes, in order to determine compensable damages or to assess the economic efficiency of restoration measures.

Despite the advocated operational nature of monetary valuation, many writers have questioned or strongly criticized the preference-related value theory underlying economic valuation and valuation techniques.

Some critics allege that the economic theory is based on a very narrow and simple definition of self-interest, and by so doing, it fails "to describe the environmental values people hold, the process of value construction, or the way individual values are aggregated into a social value" (Brouwer, 2000, p.138). Moreover, "ecocentric ideologies […] place primary emphasis on a distinction between instrumental value (expressed via human-held values) and intrinsic, non-preference-related value. They lay particular stress on the argument that functions and potentials of ecosystems themselves are a rich source of intrinsic value. This value would, it is argued, exist even if humans and their experiences were extinct" (Pearce and Turner, 1990, p.22).

However, as noted by Pearce and Turner (1990), the preference-related and the intrinsic-value distinction is not clear-cut. Individuals may capture part of the intrinsic value in their preferences, *e.g.* valuing "on behalf of" other species. Economists use the term "existence value" to encompass these notions. Similarly, the economic concept of "bequest value" is used to encompass intergenerational equity concerns.

2. Willingness-to-pay and willingness-to-accept

Environmental values are measured in money terms through the concept of individuals' willingness to pay (WTP) or willingness to accept (WTA) compensation for alterations in environmental conditions.

Of these two, the WTP approach has become the most frequently applied, whilst WTA empirical studies are relatively rare. As it will be illustrated later on, WTP is measured directly, by asking people to state a WTP amount, or indirectly, by assuming that this amount can be inferred by looking at the economic costs afforded to enjoy environmental services or at the costs incurred to acquire service substitutes.

Although it is beyond the scope of this paper to go into much detail with respect to the theoretical debate, and existing controversies about the WTP and WTA approaches, it is worth mentioning that the use of WTP rather than WTA was initially justified by an appeal to theoretical contributions (Willig, 1976) which suggested that the difference between the two welfare measures should be negligible.

However one of the earliest findings of stated preference studies was that WTP and WTA measures may differ radically, and cognitive psychologists have proposed theories explaining the substantial observed differences (Carson and Mitchell, 1993).

Moreover, another economist, Michael Hanemann (1991), has shown that the wedge between WTP and WTA can be large. The difference between the Willig's and Hanemann's theoretical findings is due to the fact that whilst the former focussed on the welfare impacts of a price change in a perfectly competitive market, the latter considered imposed quantity changes.

As argued by Carson *et al.* (2000), since changes in environmental conditions (*e.g.* natural resources damages) tend to fall into the category of imposed quantity changes, the difference between WTP and WTA measures can be very large.

Nevertheless, as mentioned before, the WTP approach has become the most frequently applied in empirical studies, and this is primarily attributable to the valuation techniques' intrinsic inability to provide reliable WTA estimates (Desvousges *et al.*, 1998; Brouwer, 2000).

Nowadays there is a broad consensus among economists about the desirable (the theoretically appropriate) welfare measure and the possible price paid –in terms of value assessment reliability– by using WTP instead of WTA estimates.

Firstly, if property rights in environmental goods and services are held by (are conventionally assigned to) people experiencing the effects of environmental changes, WTA would be the appropriate welfare measure instead of WTP (Desvousges *et al.*, 1998). This implies that the assignment of property rights "can have a substantial influence on the magnitude of the welfare measure [and] particularly when considering a reduction in an environmental service, the common practice of substituting a WTP estimate for the desired WTA measure can result in a substantial underestimate" (Carson *et al.*, 2000, p.21).

Secondly, the more unique the natural resource under consideration, the less close the WTP estimate is to the desired WTA measure, and the more substantial the underestimation of welfare changes.

3. Classifying environmental values

a) Use and non-use values

Following the anthropocentric approach predominantly adopted in the economics literature, natural resources may then be described as assets ("natural capital") the value of which stems from their service flows and their contribution to people's welfare.

This contribution may take on different forms. Understanding how people get utility from natural resources, i.e. why they may hold environmental values, and how alterations in conditions influence these values, are key elements to economic valuation and impact assessment.

When considering why individuals place values on a natural resource, a typical approach in the literature is to distinguish between those who use the resource ('s services), and those who do not (Freeman, 1993). The values held by the former group are generally termed <u>use values</u>, and may occur in many different ways.

Direct use values may derive from "consumptive uses" (*e.g.* fuelwood collection) and/or "non-consumptive uses" (*e.g.* hiking in the same forest), and may involve commercial (selling fuelwood or collecting visiting tolls) and/or non-commercial activities (home consumption of fuelwood or enjoyment of an open-access wilderness area).

Although the physical proximity is normally thought as being an essential part of use, some authors have argued that some kinds of "use" do not require the physical contact with the resource.

Randall and Stoll (1983), for example, have argued that there can be offsite uses, which they label as "vicarious consumption": *e.g.*, people can draw utility by looking in a magazine at pictures of a tropical forest.[1] This is what has also been sometimes referred to as indirect use value (Boyle and Bishop, 1987), although the latter term is more frequently, and meaningfully used to describe another category of values generated by natural assets.

[1] As argued by Freeman (1993), one valuation problem with so-called vicarious uses is that "the observable market transaction (*e.g.*, the purchase of a nature magazine) often entails the simultaneous or join use of many environmental resources, so that allocation of the market transaction to specific resources is not possible. Furthermore, vicarious use has the odd feature that use can occur even though the resource no longer exists, as through the viewing of films and photographs" (pp.268-269).

Indirect use values, also known as functional values, derive from "the natural interaction between different ecological systems and processes; in particular, the ecological functioning of one ecosystem may affect the functioning and productivity of an adjacent system that is being exploited economically" (Barbier, 1998, p.5). More generally speaking, indirect use values may be described as the benefits individuals experience, indirectly, as a consequence of the primary ecological function of a given resource (Torras, 2000). For example, the indirect use value of a wetland may arise from its contribution to filtering water exploited by downstreamer users (World Bank, 1998); forests may provide different off-site benefits, such as defense against soil erosion, flood control, or carbon sequestration; coastal wetlands may contribute to the protection of properties and economic activities against hurricane wind damages (Farber, 1987); and the use-value of a mangrove system may derive from its indirect support, as a breeding ground, for an offshore fishery (Barbier and Strand, 1998).

Besides use-values, it is largely agreed –or, at least, it is largely agreed by economists working in the field of environmental economics– that natural resources may also generate values which are unrelated to any actual, direct or indirect, use. So called non-use values "do not involve any observable behaviour; they are simply experienced "psychically". Consequently, nonuse values cannot be observed by market purchases or inferred trough actions" (Desvousges, 1995, p.4).

Non-use values, undoubtedly the most elusive component of a natural resource's total economic value, are said to arise from the psychological benefits people may derive from the mere knowledge the resource exists (*existence value*),[2] or from the desire to preserve natural capital in order to pass it to future generations (*bequest value)*.

Moreover, available taxonomies often include *option value* among non-use values. Option value may be defined as the benefits derived by an individual from preserving options for use of a particular resource when the individual is either uncertain about future use or faces uncertainty about the availability of that resource in the future.[3] [4] However, many authors have proposed not to include option value among non-use values, and to interpret it as a special case of use value, akin to insurance policy (World Bank, 1998), or have even argued that what is conventionally defined option value "is not a separate component of value; rather, it is an algebraic difference between two measures based on different perspective on valuation –an ex ante perspective focusing on option price and an ex post perspective focusing on realized surpluses. This option value can be either

[2] The concept of existence value was initially introduced by Krutilla (1967) in the context of irreversible allocation decisions involving unique natural environments. This was interpreted to imply that "irreversibility and uniqueness are required for existence values […] This is not so. Rather, Krutilla was simply arguing that existence values might be especially important in cases involving unique natural environments" (Kopp and Smith, 1993, p.322).

[3] *Quasi-option value*, a term coined by Arrow and Fisher (1974), is a related concept, and derives from the possibility that even though something appears unimportant now, information received later might lead us to re-evaluate it (World Bank, 1998). More precisely, the term was adopted to describe the welfare benefit associated with postponing a decision when there is uncertainty about the outcome of alternative choices and when at least one of the alternatives involves an irreversible commitment of resources.

[4] Various empirical studies, aimed at estimating the value of '"(quasi-)option benefits" may be found in the environmental valuation literature. In particular, most of them have focussed on the value of preserving *biodiversity* as, *inter alia*, a source of biological materials that may be exploited commercially for new industrial, agricultural, and pharmaceutical products (*e.g.* wild organisms gathered from natural habitats could provide cures for current diseases and future needs that are not yet known). Examples of these empirical works are the studies conducted by Adger *et al*. (1995), Fearnside (1997), and Grimes *et al*. (1994), which provide estimates of the option benefits of biodiversity maintenance in tropical forests (located in Mexico, Brazil, and Ecuador, respectively). However, Simpson (1997) has somehow questioned the emphasis placed upon biodiversity prospecting as a key argument in favour of conservation policies. In fact, available estimates, based upon pharmaceutical companies' willingness-to-pay to preserve biodiversity "hotspots", show that the estimated economic value of biodiversity for use in new product research is modest. "This does not imply that [biodiversity] is without [economic] value. [….] Biodiversity may be important for any number of commercial, ecological, aesthetic, ethical, or even spiritual reasons. However, when it comes to commercial prospecting among natural sources for new products, the value of biodiversity is not as high as some conservationists might suppose" (Simpson, 1997, p.5).

positive or negative depending on the particular structure of the uncertainty facing the individual" (Freeman, 1993, p.284).[5]

Although the distinction between use (or user) and non-use(r) values remains the predominant taxonomy, other classifications have been suggested in order to decompose the total value of a natural resource. This to encompass the variety of terms used in the literature to describe values not arising from resources' present uses as well as to avoid frequent conceptual overlaps between some non-use and use values' sub-categories. One of these alternative taxonomies is that which simply decomposes the total value into "direct use" and "passive use" value.[6] Following Carson *et al.* (2000), "direct use can be most easily thought of as requiring the agent to physically experience the commodity in some fashion" (p.3). Any other benefits not requiring this direct contact can be labeled as passive use value.

Although the *in situ* presence –the "physical contact" to the resource under consideration– may constitute an useful classification rule, another criterion may prove to be even more useful. This criterion derives from focusing on *whether or not individuals need to carry out an activity –entailing the use of other (marketed or unmarketable) economic goods– in order to get utility from a natural asset.*

The main advantage of this general classification criterion is that it probably allows a better understanding of the linkage between environmental values and valuation methodologies. In fact, as it will be better illustrated later on, the various methods developed to measure these values can be classified according to the way in which the values people attach to natural resources are assessed.

In particular, a group of methods tries to infer resource values by examining the purchases of related goods in the market place. In general, these related goods are factor inputs in the consumption (or production) activity required to get utility from a natural asset, or required to compensate environmental changes, in order to preserve the same level of utility (output).

Alternatively, other valuation methods do not rely upon information about individuals' purchases of natural resources' complementary, or substitute, goods, and try to measure the resources' value by directly asking people to state how much they are willing to pay to avoid (to undertake) negative (positive) environmental changes. These expressed preference methods are typically, although not exclusively, employed when analysts believe that the resource's total value would be severely undermined by (only) looking at the utility the individuals get from it by carrying out activities involving the use of other economic goods.

b) Other classifications of environmental services

So far we have mainly focused on the distinction between values deriving from using natural assets (' services), and assets' values which are independent of present or expected use. We have also argued that, from a valuation perspective, the presence or the absence of activities entailing the use of other economic goods may prove useful to draw an operational borderline between environmental services underlying use and non-use values. The reference to the presence of an

[5] Fisher (2000) provides an interesting critical review of the concepts of option and quasi-option value, as developed in the environmental economics literature, and discusses the interlinkages and overlaps between these concepts and the concept of option value developed by Dixit and Pindick (1994) in the field of investment decisions under conditions of uncertainty and irreversibility. Fisher's main conclusion is that, whilst the traditional concept of option value developed in the literature on environmental preservation (*e.g.* by Cicchetti and Freeman, 1971), is essentially static, related to risk aversion, and can be either positive or negative, the concept of quasi-option value due to Arrow and Fisher (1974) is, like the Dixit-Pindyck measure, dynamic, not dependent on risk aversion, and non-negative.

[6] The term "passive use value" was adopted in 1989 by a US Court (case *Ohio v. U.S. Department of Interior*) to encompass a number of terms such as non-use value, existence value, preservation value, bequest value, stewardship value, intrinsic value, and option value (Carlson *et al.*, 2000).

economic activity would imply that the “physical contact” is not a necessary condition to infer a resource’s use value, as long as individuals get off-site benefits from it through carrying out activities involving other economic goods.

Obviously, a natural asset may generate both non-use and (direct or indirect) use values. In particular, whilst nonusers can hold only nonuser values, users may hold both non-use and use values (Freeman, 1993). Moreover, as far as the latter are concerned, it may be useful to make a distinction between natural assets which support only one economic activity (*single use* resources) and assets which may (simultaneously) support many different activities (*multiple use* resources).

Besides the use/non-use distinction, and the single/multiple use one, an additional classification criteria also appears to be very useful when trying to assess the total value of a natural asset. We refer to the distinction between public environmental services and private environmental services.

Pure public services are those benefits flowing from a natural asset which can be enjoyed by one individual without detracting from the enjoyment opportunities still available to others (non-rivalry or indivisibility of benefits), and which cannot be withheld, at a reasonable cost, by the “owner” of the natural asset under consideration (non-excludability of benefits). On the contrary, excludable environmental services which cannot be enjoyed by one individual without affecting the other individuals’ enjoyment opportunities (from the same unit of service) are labeled pure private ones.

In between points along the spectrum of fully non rival/rival and costly/costlessly excludable services are called quasi-public/private goods. The latter term is used to encompass environmental services whose enjoyment by one additional individual does not affect others’ enjoyment up to a point, but beyond that point congestion –which may be interpreted as a “public bad”– reduces the enjoyment of all existing and potential beneficiaries; and/or services whose enjoyment can be technically controlled, but this control is not exerted because of the lack of well-defined property rights or the high costs of exclusion.[7]

As stated before, a natural asset may entail both non-use and (single or multiple, direct or indirect) use values, and the asset’s services, underlying these values, may be often placed along the private/public good spectrum.

Whilst environmental services underlying existence or bequest values are, almost by definition, public goods, those underlying use values often hold private or quasi-private/private features. However, some services underlying direct use (*e.g.* visual amenity benefits) or indirect use (flood control) values may display both non-rivalry and non-excludability, and may be labeled as pure public ones.

Various implications stem from environmental services’ private, public (or the combination of private/public) features. From a valuation perspective, the main consequence is that whilst private goods are marketable, goods holding public features are not exchanged in “normal markets”. It follows that whilst observable market prices provide useful –and sometimes sufficient– information for assessing the value of private environmental services, the social benefits arising from public environmental services cannot be directly deduced by market prices, and alternative approaches have to be adopted to infer their economic value.

[7] A variety of terms and concepts have been proposed in the economics literature to designate quasi-private/public goods’ subcategories. Goods whose benefits are excludable but partially rival (i.e. congestion may affect individuals’ enjoyment) are generally classified as *club goods* (Cornes and Sandler, 1986) (*e.g.* recreational fishing in a small lake). Goods whose benefits are nonrival but users can be excluded at a relatively low cost are often described as *toll goods* (*e.g.* navigation along a watercourse), whilst goods which are rival and the cost of exclusion is high are called *open access goods* (*e.g.* groundwater abstractions) (World Bank, 1993).

4. Valuation approaches

As it will be illustrated in the following section, various techniques have been developed to measure natural assets' values in order to assess the economic impacts resulting from alterations of conditions influencing the flow of goods and services these assets provide. Broadly speaking, these techniques can be grouped into three major valuation approaches.

The first one consists of exploiting the existence of a market price for an environmental good, in order to assess its economic value. If the observable prices are not distorted, then the economic value of (marginal) environmental changes can be valued by directly using existing market prices.[8] Obviously, if the natural resource of interest provides multiple goods and services, some (many, or even all) of which are unmarketable, this valuation approach would fail to provide reliable measures of the resource's value.

The second approach (surrogate market valuation) consists of measuring the value of unmarketable environmental services by looking at the market price (or the shadow price) of related economic goods. These related goods may consists of: (i) environmental services' complementary goods (i.e. goods required to enjoy environmental services); (ii) substitute goods (i.e. goods which may replace environmental services, or reduce/avoid the economic impacts of changes in service flow); (iii) other marketable goods providing indirect information about the environmental change's economic impacts. Again, the surrogate market valuation approach is potentially capable of providing reliable welfare measures only if the value of the natural resource under consideration is revealed by related market behaviour and market prices. This may occur for use values, but will never occur for non-use values. It follows that if a resource does not (only) provide benefits through its present (or expected) use, but because of its mere existence, the surrogate market valuation techniques are intrinsically unable to provide (reliable) value estimates.

Finally, the third approach (expressed preference approach) consists of directly asking individuals which value they attach to unmarketable environmental services, and to express their preferences towards changes in service flows. This approach is potentially able to estimate both use and non-use values, or simply –when applied in an holistic way– a natural resource's total value.

What are the main fields of application of the above–mentioned valuation approaches? Similarly to the private/public good spectrum, an analogous continuum can be used as a sort of conceptual device for identifying the linkages between environmental values, and valuation methodologies.

On the one extreme of the "valuation spectrum" we may place *environmental attributes, underlying non-use values, holding pure public features*: these values can only be assessed through expressed preference methods. On the other extreme of this spectrum we may place *private/quasi-private marketed environmental attributes*, typically underlying direct-use values, which can be measured by directly exploiting market prices.

In-between points along this spectrum we find *quasi-public/public unmarketable environmental attributes, underlying direct or, more frequently, indirect use values*, which can be assessed through surrogate market valuation or expressed valuation methods. The choice between these two valuation approaches mostly depends on (i) whether or not other relevant goods are involved in generating these values, and (ii) the economic nature of these related goods. When related goods hold private/quasi-private features (marketed or marketable goods), the surrogate valuation approach is potentially capable of inferring the value of unmarketable environmental attributes. On the contrary, when there are no relevant marketed goods involved, analysts must inevitably turn to expressed preference methods.

8 Even when the market of interest does not exhibit a competitive structure, observable market prices still provide useful 'baseline' information for identifying appropriate shadow prices and estimating the environmental values. If the environmental good of interest is potentially marketable, but prices cannot be observed (for example products harvested for home consumption) its economic value can be estimated by using the market price of close substitutes or the (opportunity) cost of harvesting.

II. Valuation techniques: an overview

1. Direct and indirect techniques

The valuation literature provides various taxonomies of techniques developed to measure the economic value of unmarketable environmental attributes. Here we will adopt the taxonomy proposed by Pearce and Turner (1990) and Turner *et al.* (1994) who classify the available techniques as "direct" (or "environmental demand curve approach") and "indirect techniques" ("non-demand approaches").

The direct techniques seek to directly measure the monetary value of environmental services. This may be done by looking for a surrogate market –typically the market of complementary goods or other factor inputs in the 'household's production function'–[9] in order to infer individuals' preferences, or by asking individuals to express their preferences. Following Pearce and Turner (1990) and Garrod and Willis (1999), the *travel-cost method*, the *hedonic price method*, and *the contingent valuation method* hold to the direct approach.

The indirect techniques do not seek to directly measure individual preferences. "Instead, they calculate a "dose-response" relationship between [say] pollution and some effect, and only then is some measure of preference for that effect applied" (Pearce and Turner,1990,p.142). According to Garrod and Willis (1999), because indirect techniques do not value the environmental commodity via a

[9] The concept of *household-production function* is briefly illustrated in Section II.4.

demand curve, they tend to fail to provide "true" valuation information and welfare measures. Although the literature does not provide an univocal and clear-cut classification of direct and indirect techniques, the so-called *production-function* and *cost-based valuation methods* are usually included in the latter group.

2. The production-function method

The production-function method (otherwise known as "change-in-productivity approach", "effect on production approach", or "valuing the environment as an input") seeks to exploit the relationship between environmental attributes and the output level of an economic activity.

The underlying assumption is that, when an environmental attribute enters a firm's production function, environmental changes' economic impacts may be measured by looking at the effect on production, and by valuing such effect at market (or shadow adjusted) output prices. As underlined in the previous section, the money estimates obtained in this way should not be interpreted as the "true" value measure, but as a proxy of the environmental change's ultimate welfare impacts.

The production-function approach (PFA) has been widely used, particularly to evaluate the impacts of environmental quality changes (*e.g.* acid rain or water pollution) upon agriculture (*e.g.* Adams et al. 1986) and fisheries (*e.g.* Kahn,1991). Other examples of application include analysis of the impacts of water diversion (Barbier,1998), and the valuation of the protection benefits provided by coastal wetlands against hurricane damage (Farber, 1987).

According to Barbier (1998), because of the direct dependence of many production systems in developing countries on natural resources and ecological functions, the PFA is considered widely applicable to many important economic and investment decisions in these countries.

Broadly speaking, the PFA consists of a two-step procedure. The first one is aimed at identifying the physical impacts of environmental changes on a production activity. The second step consists of valuing these changes in terms of the corresponding change in the activity's output.

Clearly, particularly at the first stage, co-operation is required between natural scientists, economists and other researchers, in order to determine the nature of the environment-production linkages (Barbier, 1998).

By indicating with Y the activity's output, with ENV the environmental variable(s) of interest, and with X_i (i=1……N) other inputs, the production function of a representative firm might look like:

$$\mathbf{Y = f(X_i, ENV)} \qquad \mathbf{(1)}$$

If $\delta Y/\delta ENV$ is positive, then a change in ENV (*e.g.* an increase or decrease in water pollution) will, *ceteris paribus*, decrease/increase output levels.

Broadly speaking, when Y is a marketed good, and the observable price is not affected by relevant market-failures, this price (or a shadow adjusted price) can be used to estimate the value of a change in ENV. Alternatively, this value can be estimated by looking at the changes in marketed inputs (Xi) required to maintain a given level of output.[10]

[10] When looking at the costs incurred to acquire additional (marketed) inputs, in order to mitigate the impacts of environmental changes upon a firm's output, the PFA becomes equivalent to some of the cost-based methods illustrated in the next section (namely, *averting behaviour models*). However, in the valuation literature, the latter term is generally used to describe valuation models focussing on households' substitution possibilities between environmental attributes and other goods or services.

Various quantitative methods have been used to estimate the economic costs (or benefits) of environmental changes affecting production activities. Following Hanley and Spash (1993), these methods can be classified as follows: (i) "traditional' type models" (or "historical approach"); (ii) "optimization models"; (iii) "econometric models".

The first method is quite simple, and its main advantage is that the informational requirements are relatively modest. Once the physical relationship environmental variables and the output level has been identified, the monetary value of the environmental change is estimated by multiplying the output change by the current output price. The main caveat of this method is that it ignores possible price changes. Although prices may be unaffected by marginal environmental changes, significant and widespread changes in environmental conditions could entail not-negligible price effects, so that the assumption of constant price could provide seriously biased welfare measures.

The optimization models require extensive data sets, but provide more detailed information, and allow indirect effects to be considered. In particular, quadratic programming models allow to treat both price and quantities and endogenous variables. However, because of their normative nature, discrepancies may emerge between the model solutions and reality, and identifying the source of such discrepancies may prove difficult.[11]

Finally, econometric models do not adopt a normative approach, but, by using observable data, and their variations over space or time (or both), try to get factual evidence about the inter-relationships of interest. "This applied work is objective in the sense that the results can be rigorously examined using accepted scientific and statistical methods, although ideological bias can be expected both in the selection of questions investigated and in the inferences drawn from factual evidence" (Hanley and Spash,1993, p.106).

Leaving aside the above mentioned specific possible caveats arising from the choice of the quantitative method, a number of more general problems may arise when applying the PFA. Following Barbier (1998) these potential drawbacks may be summarised as follows.

Firstly, as already mentioned, the price of the output can be heavily distorted, i.e. it may fail to provide a reliable proxy of the output's economic value. Besides market failures, prices may be distorted by fiscal policies (taxation or subsidization). Moreover, public regulatory policies (or the absence of appropriate regulations) may influence the values imputed to the environmental input (ENV). For example, when considering the impacts of an environmental change, say a change in a coastal wetland supporting an off shore fishery, if the latter is subject to open-access conditions, "rents in the fishery would be dissipated, and price would be equated to average and not marginal costs. As a consequence, producer surplus is zero and only consumer surplus determines the value of increased wetland area" (Barbier, 1998, p.8).

Secondly, applications of the PFA may be most straightforward in the case of a natural resource ('s services) supporting only one economic activity (single-use resources) than in the case of multiple-use resources.[12] In fact, when a natural resource supports many different economic activities, "applications of the production function approach may be slightly problematic […] and assumptions concerning the ecological relationships among these various multiple uses must be carefully constructed to avoid problems of double counting and trade-offs between the different values" (Barbier, 1998, p.8).

[11] As underlined by Hanley and Spash (1993), since optimization models describe the world as it should be, given certain assumptions, "when discrepancies arise between the [normative] model solutions and reality, the cause will be uncertain. Such discrepancies could be due to incorrect or inaccurate modeling of production activities, improper constraints or just the fact that the real world operates sub-optimally" (p.106).

[12] See Section I.3.

Finally, "for some valuation problems, choosing whether to incorporate intertemporal aspects of environment can be very important" (Barbier,1998, p.9). For example, in their study aimed at estimating the value of estuarine wetlands and mangroves in supporting off-shore fishery in the state of Campeche (Mexico), Barbier and Strand (1998) have adopted, and compared, a "static valuation approach", and a "dynamic valuation approach". The former valuation exercise assumes that fish stocks are always constant. The latter attempts to model the impact of a change in coastal wetland area on the growth function of the intertemporal fish harvesting process.

3. Cost-based methods

When the impacts of environmental changes do not (exclusively) manifest themselves through changes in firms' marketed outputs, information on related costs can be used to obtain estimates of the welfare impacts.

Various techniques, falling within the broad class of "cost-based approaches", have been applied to estimate the social rate of return of projects which were expected to entail significant environmental changes, or to assess the impacts of actual changes in damage assessment cases.

Broadly speaking, these techniques can be classified according to: (i) the nature of environmental changes; (ii) the effects of such changes; (iii) the individuals' ability to react to them; and (iv) the nature of the reactive actions.

a) Averting behaviour and relocation cost approach

Individuals may be able to react to environmental changes. For example, to avoid or reduce the health effects of increased water pollution, households may undertake averting expenditures such as buying bottled mineral water, spending energy (and time) to boil water, or acquiring water treatment equipment.

The averting behaviour approach exploits individuals' willingness-to-pay for avoiding (preventing or mitigating) the effects of negative environmental changes in order to infer the value of environmental quality. If the costs incurred to mitigate, or prevent, the effects of pollution can be estimated with a reasonable level of accuracy, the value of decreasing (increasing) environmental quality may be inferred by looking at the increase (decrease) in averting expenditure (AE).

This valuation method relies on various assumptions which affect its ability to provide reliable estimates of the 'true' value of an environmental change. These assumptions may be summarised as follows: (i) AE and environmental quality are close "substitutes"; (ii) AE is only explained by the environmental change of interest and does not generate additional benefits; (iii) AE is reversible.

These assumptions are unlikely to fully describe reality. When the AE under consideration is unable to fully offset a negative environmental quality change (i.e. AE and environmental quality are imperfect substitutes), the method provides an underestimation of the true welfare cost; on the contrary, if AE provides additional benefits, other than mitigating or preventing the effects of pollution, the method tends to over-estimate the true welfare cost of increased pollution (the benefits of decreased pollution). Moreover, as noted by Hanley and Spash (1993), AE may entail sunk costs, i.e. "investment in defensive equipment may be difficult to reverse, preventing households from moving to the position where the marginal costs and marginal benefits of pollution avoidance are equated" (p.99).

When relocation is the only averting/mitigating behavioural option, i.e. the only available option to avoid the impacts of a negative environmental change is moving to a different location, the economic value of this change can be estimated by exploiting the information on relocation costs (relocation-cost approach). For example, the value of an expected or actual increase in air pollution can be inferred by looking at the additional costs –*e.g.* additional transportation cost– individuals are prepared to incur (have incurred) by moving to an area, with less pollution, at a greater distance from their workplace (Garrod and Willis, 1999).

Similarly to the averting behaviour approach, the relocation-cost method may fail to provide true valuation information and welfare measures, i.e. it may involve an underestimation or an overestimation of the economic value of pollution, depending on whether or not, by moving, individuals are able to recover the same level of environmental services, and on whether or not relocation is driven only by the environmental quality at different sites.

b) Cost of illness and human capital approach

The cost-of-illness method has been quite frequently used to estimate the welfare effects associated to environmental changes involving changes in the level of morbidity. For example, this approach was adopted to estimate the economic benefits of pollution control measures undertaken in Santiago (Chile) to reduce the concentrations of air pollutants such as particulates, volatile organic compounds and nitrous oxides (World Bank, 1994).

The method can be applied when environmental changes have repercussions on human health, and when (it is assumed that) individuals are unable to react, i.e. when they may not undertake defensive actions (i.e. averting expenditures) to reduce health risks.

In these cases, the costs (benefits) of an increased (decreased) level of pollution can be estimated by using information on: (i) the relationship between environmental quality changes and changes in the level of morbidity; and (ii) the economic costs (benefits) associated with changes in the level of morbidity.

As far as the latter are concerned, besides medical costs, and other out-of-pocket expenses, any loss of earnings, due to an increase in morbidity, should be accounted for, in order to assess the welfare impacts of increased (decreased) levels of pollution involving health effects.

In principle, also non-market losses associated with sickness, such as pain and suffering to the affected individuals and other concerned, as well as restrictions to non-work activities, should be accounted for. However, these “intangible” effects are not in general taken into account, because of the difficulty to translate these effects into monetary values. This implies that the cost-of-illness estimates should, in general, be interpreted as lower-bound estimates of the “true” costs (benefits) associated to increased (reduced) pollution levels affecting health risks. Moreover, this method is intrinsically unable to evaluate the welfare effects of environmental changes which do not (exclusively) manifest themselves through changes in the level of morbidity.[13]

The so-called human-capital approach is an extension of the cost-of-illness method, in that the environmental changes’ impacts are assessed by looking at the relationship between environmental quality and mortality rates. However, this approach is much more problematic, in that it entails an estimation of the value of human life. This can be done by looking at the present value of an individual’s future income stream. But, besides the difficulty in predicting the expected life-time earnings, reducing the value of life to individuals’ expected productivity is extremely controversial, and some agencies have recommended not using this approach, and instead, to eventually use measures of the value of a statistical life based on willingness to pay estimates

[13] The same can be said for the averting behaviour method, which is unable to capture the impacts of environmental changes which do not involve expenditures aimed at preserving environmental use-values. Besides those use-values which cannot be preserved through affording economic costs, these valuation methods are intrinsically unable to capture non-use values.

"which includes much more that just lost productivity and is often 5 to 10 or more times larger than the straight human-capital estimates" (World Bank, 1998).[14]

c) Restoration cost approach

When restoring the environment to its original state –i.e. restoring a natural asset's original service flow– is technically feasible, the restoration cost may be used as a measure of the costs (benefits) of (avoided) negative environmental changes.

The restoration cost approach has been quite frequently used in cost-benefit analyses of new projects and public policies, and, in some countries, forms the basis of compensable damage assessment (*e.g.* in the United States, under the Comprehensive Environmental Response, Compensation and Liability Act, CERCLA) (Garrod and Willis, 1999).

Besides requiring that the costs to restore a natural asset ('s services) can be estimated with a reasonable level of accuracy, this approach –which cannot be applied to very unique and irreplaceable assets– implicitly assumes that restoration costs do not exceed the economic value of the asset ('s services).[15]

This assumption may not be valid in all cases. As argued by World Bank (1998), "it simply may cost more to restore an asset that it was worth in the first place" (p.6). More generally speaking, if environmental substitutes are available, and these substitutes can be acquired at a cost lower than the cost required to restore a damaged natural asset, then the restoration-cost method will provide an overestimation (an upper-bound estimate) of the "true" damage.

4. The travel-cost method

The travel-cost method (TCM) was designed and is generally used to value environmental attributes which are exploited to acquire recreation services.

The intuition underlying the TCM is simple. Even when entry to a recreation site is free of charge, individuals willing to enjoy environmental attributes generally need to afford economic costs. Besides out-of-pocket expenditures (transport costs) individuals need to use other "inputs" (other economic goods), such as time, to gain access to a recreation site.

By looking at the total cost afforded to gain access to the recreation site, the TCM tries to infer the demand for the site. Once this demand –i.e. the relationship between the cost of visiting a recreation site and the number of visits observed– has been identified, the total benefit recreators obtain can be calculated by using, as a welfare measure, the visitors' consumer surplus, i.e. the benefit visitors enjoy above the costs involved in carrying out the recreational activity.

The TCM can then be interpreted as a special case of the production function approach. More specifically, the TCM uses a "household-production framework": in fact, as a firm may combine environmental goods with other purchased inputs to produce marketable commodities, households may get utility by combining environmental attributes with other economic goods, to acquire recreation services.

Traditionally, TCM studies have used one equation to model the number of trips people take to a specific recreation site ("single-site models"), and have assumed that the number of trips is a function of travel costs, and that the travel cost is proportional to distance from the site. Moreover,

[14] Estimates of a statistical life based on WTP measures are available for many developed countries (World Bank, 1996). As argued by World Bank (1998), these measures might be used for other contexts, by adjusting available estimates using relative per capita GNP.

[15] See Section III.4 and III.7

a single-purpose trip has been frequently assumed. All these assumptions are "often valid in the case of [tourism within a country but] may not be valid for international tourism" (World Bank, 1998, p.9).

Moreover, one of the major drawbacks of the single-site models is their inability "to account for substitution among recreation sites [and their] inability to determine the importance of individual site characteristics. If there are substitutes for the site, an increase in travel cost would induce people to visit another site rather than forego recreation altogether […] Because the travel-cost model does not incorporate this substitution in any meaningful way, the method overstates the benefits of the recreation site (Desvousges *et al*, 1998, p.20).

"Multiple-site models" have been developed to overcome some of these drawbacks. However, even these models can only value a trip as a whole, and are unable to value changes of one specific environmental attribute of a site (Desvousges *et al*, 1998). This may pose problems when a valuation study is not aimed at assessing the value of a natural resource *per se*, but, say, at measuring the value of a negative environmental change. As noticed by McConnell (1993), for measuring environmental damages, "the successful use of the travel cost model requires not simply that the model itself reflects the demand for services of the public natural resource, but that the model accurately captures the change in demand for the service after the resource is injured" (McConnell, 1993, p.191).

TCM has been widely used to evaluate the use-value (recreational use value) of natural assets located both in developed and developing countries. As far as the latter are concerned, the main application is to valuing international tourists' willingness to pay for (visiting) wilderness areas. For example, Mekhaus and Lober (1996) have carried out a travel-cost study, aimed at assessing the benefits obtained by tourists visiting national parks and reserves in Costa Rica.

5. The hedonic pricing method

a) Underlying assumptions

Hedonic price valuation tries to measure the value of an unmarketed environmental service as a measurable component ("attribute" or "characteristic") of a marketed good (Anderson, 1993).

The method, which may be traced back to the characteristics theory of value developed by Lancaster (1966), relies on the proposition that an individual's utility for as a good is based on its attributes. As long as the latter include environmental attributes, by modeling individuals' willingness to pay for a particular good as a function of its characteristics, hedonic pricing tries to pick up the impacts of changes in environmental service flows upon individuals' utility.

The most common applications of the hedonic pricing method (HPM) try to exploit the relationship between property values –often, although not exclusively, residential property values– and environmental attributes of the neighbourhood (*e.g.* air quality, noise levels, access to recreational facilities, visual amenities).[16]

However, besides the so-called *property value approach* (World Bank, 1998), the HPM has been also applied to the labour market and wage rates: the *wage differential approach*'s underlying assumption is that an individual's choice of a particular job may be affected by the job's location's

[16] The majority of property value studies rely upon housing data. However, there are many applications of the hedonic method exploring interlinkages between environmental conditions and other assets' prices. In particular, various applications concerning cropland values may be found, trying to infer the value of environmental services such as fertility or access to water facilities. Examples include the hedonic studies conducted by Miranowski and Hammes (1984) and Ervin and Mill (1985) in the United States, to explore the effects of soil quality and erosion on cropland values.

surrounding environmental conditions or by the perceived risk of natural hazards (*wage-risk analysis*).[17]

b) Basic steps

The typical steps of an hedonic study may be broadly described as follows. The first one consists of selecting the environmental variable(s) of interest and of deciding the marketed good whose price is expected to provide information about the implicit environmental value(s) (henceforth, the "environmental price(s)").

As far as the dependent variable is concerned, either purchase or rental data may be used in property valuation studies, depending on data availability, data quality, and market conditions.[18]

Assuming purchase price (e.g. house price, P_H) is used as the dependent variable, the second step consists of identifying all other explanatory variables which, together with the environmental variable (ENV),[19] are thought to describe the property's attributes. The choice of the relevant attributes is potentially crucial (Hanley and Spash, 1993), in that failure to include property's relevant attributes correlated with some or all of the included characteristics, may lead to significantly biased estimates for these characteristics' implicit prices (i.e., *inter alia*, biased environmental prices).[20]

Two particular "omitted-variable-bias problems" should deserve specific and special attention.

The first concerns the question of so-called "averting behaviour" (Garrod and Willis, 1999), where owner-occupiers (landlords or tenants) spend money for preventing or mitigating neighbourhood's negative environmental conditions. As noticed by Kuik *et al.* (1992), the effects of averting behaviour (other than moving to a different location) are often neglected in hedonic studies because of the difficulties in acquiring detailed information.

The second problem has to do with the difference between actual and expected environmental changes. If an hedonic price study is aimed at inferring the value of an actual change in environmental conditions (say, the value a specific change in air quality), as long as property prices are also affected by expected changes (i.e. expected neighbourhood changes are one of the property's attributes) excluding expected changes from property prices' explanatory variables leads to omitted variable bias (Hanley and Spash, 1993).

Expectations about future benefits (or costs) associated to environmental changes do not give rise to the afore-mentioned omitted variable bias problem, as long as these expectations concern

17 Various wage-risk studies have been conducted in the developed world. Examples include lethal or non-lethal risks related to skin cancer (ozon layer), radiation concentration, soil pollution (toxic wastes), nuclear accidents (see Kuik *et al.*, 1992, table 2.4, pp.19-23). By contrast, according to Garrod and Willis (1999), in many developing countries the composition of labour markets make it unlikely that this approach would be useful in that context.

18 In particular, in countries with a large owner—occupier sector, it may be more appropriate and convenient to use purchase data, whilst in countries with a large rental sector, or which have little tradition of buying and selling houses, an *hedonic rent model* would be a more practical methodology (Garrod and Willis, 1999). Moreover, the choice of the dependent variable should account for the existence of market distortions or regulatory mechanisms (*e.g.* rent control) which may affect the estimated environmental prices' reliability.

19 ENV can represent a measure of either a quantity or a quality. The choice of what measure to use has important implications. For example, in the case of air quality, "is a seasonally weighted mean appropriate, or will a simple mean be sufficient? These sorts of questions usually require expert guidance to solve" (Garrod and Willis, 1999, p.106).

20 In hedonic studies using housing data, a number of property variables (*e.g.* number of rooms, structural integrity, etc.), neighbourhood socio-economic variables (job opportunities, ethnical composition, etc.) and variables reflecting non-environmental local amenities (access to public services, communications, etc.) are typically included among the explanatory variables.

future benefits –or costs– associated to actual (not expected) changes.[21] In fact, as argued by Garrod and Willis (1999), each environmental attribute is not valued with respect to the benefits it currently provides, but for the stream of future benefits which it will subsequently generate. In other words, "house prices should reflect the capitalized value of environmental quality to the home-owner" (Hanley and Spash, 1993, p.75).

Once the analyst has identified a plausible set of relevant property's attributes (C_i; i=1….N.), the next step consists of estimating an "hedonic price equation", holding the following general form:

$$\mathbf{P_H = f\,(ENV, C_i)} \qquad \mathbf{(2)}$$

The specification of function (2) plays a crucial role in hedonic studies. Since economic theory does not impose restrictions on the hedonic price function (Rosen, 1974), analysts may in principle adopt different functional forms, and "even for a given data set, criteria for functional form selection may be conflicting" (Hanley and Spash, p.79).[22]

Pioneering HPM studies have mostly adopted linear functional forms, which imply that the implicit prices of the property's attributes are constant.[23] In other words, the marginal cost (or benefit) of ENV changes would be independent of the level of ENV and of the composition of property's attributes.[24]

However, from Rosen (1974) onwards, many authors have argued that implicit prices are unlikely to be independent of the quantity of each property's attributes, since this would only occur if individuals were able to "re-package" property's attributes.

In other words, the hedonic price equation is expected to be non-linear, because "house buyers cannot treat individual housing attributes as discrete items for which they can pick and mix until their desired combination of characteristics is found. On the contrary, most properties embody a set of attributes which are not readily adjustable and homebuyers are limited in their choice to those properties available on the market" (Garrod and Willis, 1999, p.112).

Once the hedonic price equation has been specified, the environmental price, i.e. the value of a marginal change in ENV, is obtained by partially differentiating (2) with respect to ENV:

$$\mathbf{P_E = \delta P_H/\delta ENV = g(ENV,Ci)} \qquad \mathbf{(3)}$$

If all individuals were identical in every respect, *e.g.* all house buyers hold the same preference for a specific environmental attribute, (3) would give the (inverse) demand function for ENV. Otherwise, if we are to obtain an estimate of individuals' willingness to pay for given levels

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21 This does not imply that *environmental risks* (i.e. the value individuals attach to their exposure to natural hazards, rather than the value they assign to actual environmental changes) cannot be valued through hedonic methods. Probably the best known application of HPM to environmental risk is the study conducted by Brookshire *et al.* (1985) who examined the impact on property values of information on different levels of earthquake damage in residential areas of San Francisco and Los Angeles.

22 According to Garrod and Allison (1991), these criteria include: (i) parsimony (functional forms requiring as few parameter as possible; (ii) ability to allow clear economic interpretations of the results; (iii) ability to explain the observed data; and (iv) ability to make good predictions. In particular, as far as the potential trade-offs involved in choosing a particular functional form is concerned, Garrod and Willis (1999) argue that "the choice […] will depend on whether the principal objective of the study is to derive estimates of [implicit 'environmental prices'] or to generate conditional predictions of house prices or rents. The former objective requires careful consideration of the structure and parameterization of the hedonic price model whereas the latter demands close attention to the robustness of the model and its extrapolative plausibility" (p. 111).

23 For example, by using a loglinear functional form:

$$\ln P_H = \alpha \ln ENV + \beta \ln C_1 + \ldots + \gamma \ln C_N$$

by means of multiple regression we can get parameter estimates, and the estimated α "will tell us how much the property prices vary if we alter the value of the environmental variable [ENV]" (Pearce and Turner, 1990).

24 On the contrary, non-linearity would imply that as ENV increases (*e.g.* air quality increases), the property price rises but not at a constant rate. For example, if the property price rises, but a decreasing rate, the marginal cost of ENV falls as ENV rises. "An alternative possibility […] is that house prices rise at an increasing rate as [ENV] rises; this means that the marginal costs of [ENV] are increasing. Both scenarios are plausible" (Hanley and Spash, 1993, p.76).

of ENV, what is required is to see how this WTP varies according to individuals' characteristics (e.g. income, age, education, etc.). This requires a further statistical exercise (Pearce and Turner,1990).

This further exercise is aimed at getting a demand curve for ENV, by using the information acquired in the previous step, namely, by regressing P_E against ENV, and any socio-economic variables (S_j; j = 1….K) which may represent individuals' preference (WTP) for the environmental attribute of interest:

$$\mathbf{P_{Ej} = h\ (ENV,\ S_j)} \qquad \mathbf{(4)}$$

Once (4) has been estimated, the value of a non-marginal change in ENV can also be estimated, by measuring the appropriate area under (4), using area averages for S_j (j = 1….K) (Hanley and Spash, 1993).

c) Potential drawbacks

Although the application of the HPM has been widespread, there are a number of potential problems associated with this method.

The problems most frequently cited in the valuation literature may be summarized as follows: (i) the method's limits in completeness and comprehensiveness; (ii) restrictive theoretical assumptions; (iii) statistical problems; and (iv) data intensity.

As far as the ability to measure the environmental changes' overall impacts upon people's welfare is concerned, it is worth noting that, similar to other revealed preference methods, HPM assumes "weak complementarity",[25] which implies that HPM may only estimate well-perceived changes of a property's neighbourhood's environmental characteristics, and it does not estimate the impacts of ENV changes elsewhere (Kuik *et al.*, 1992).[26] More generally speaking, like averting-behaviour and travel-cost, the HPM is only able to pick up to provide value estimates of the impacts of environmental changes affecting the individuals' WTP for private goods. Thus, HPM is intrinsically unable to estimate non-use values (existence and bequest values), and is incapable of estimating the impacts of changes in service flows, underlying use values, which are not reflected by the selected marketed good's price (property prices or wages).

As far as other theoretical assumptions are concerned, HPM assumes that the private good's market (e.g. the housing market) is in equilibrium, the individuals are perfectly informed about the good's attributes (environmental attributes at every possible location), and are able to move to utility maximizing positions. Only when all these conditions are satisfied, HPM gives accurate estimates (of a sub-set) of environmental values (namely use-values reflected by the WTP for a related marketed good). Obviously, these assumptions are unlikely to fully describe reality (Hanley and Spash, 1993). In particular, property prices may be distorted by market failures and government interventions which, if ignored in hedonic studies, may seriously bias estimates of implicit environmental prices. Supply problems or other mobility restrictions, particularly in urban areas, may affect individuals' ability to satisfy their demand for environmental quality, and, by so-doing,

[25] Broadly speaking, *weak complementarity* (WC) (Mäler, 1974), means that if an individual does not use the marketed good (H), his/her marginal willingness to pay for ENV is zero. In other words, the individual places no value on ENV when the individual's consumption level of H is zero. For a more rigorous description of the conditions on the individuals' utility and demand function which must be satisfied to fit Mäler's definition of WC, see Freeman (1993, pp.270-271).

[26] The same consideration applies to *hedonic wage models*, because "wages are not generally paid as compensation for variations in environmental goods outside the workplace" (Garrod and Willis, 1999,p.8).

biasing value estimates. Similar problems may be encountered when conducting wage-differential and wage-risk analyses.[27]

We have already mentioned some statistical problems, leading to errors in HPM estimates: mis-specification errors related to the choice of the functional form for the hedonic price function (2), and omitted variable-bias problems, related to the choice of the function's argument. In particular, as far as the latter are concerned, we have called attention to errors in the estimation of the implicit price of an actual change of environmental conditions, attributable to omitting averting behaviour and individuals' expectations about future environmental changes.

Another potential statistical problem frequently cited in the evaluation literature is related to multicollinearity. Multicollinearity arises when two or more explanatory variables (or combination of variables) are highly, but not perfectly, correlated with each other (Pindyck and Rubinfeld, 1986). In hedonic valuation exercises, this is likely to occur. For example, some property's attributes, such as neighbourhood socio-economic variables, may be highly correlated with each other, and some of these variables may be closely correlated with the environmental variable(s) of interest. This will mean that the effects of some attributes on a property's price may be impossible to interpret individually (Garrod and Willis, 1999).[28] [29]

Finally, in order to carry out an hedonic price study –and to reduce or circumvent some of the afore-mentioned potential statistical problems– adequate data sources are required. For example, property valuation studies need open reporting of properties' prices and records of market transactions including information about properties' relevant attributes.[30] Since properties' environmental attributes (as well as many neighbourhood socio-economic variables) are unlikely to be recorded in property sales, additional information sources of neighbourhood data (e.g. GISs) are also required.

Because of their data intensity and data quality requirements, HPM has had a limited (but often successful) application in developing countries (World Bank, 1998).

6. The contingent valuation method

In the previous sections we have provided an overview of valuation techniques exploiting revealed preferences toward some marketed good, with a connection to the (non-marketed) environmental attibute(s) of interest, in order to gain insights about the latter's economic value.

In contrast, the stated (or expressed) preference approach, usually referred to as the contingent valuation method (CVM), consists of directly asking individuals the value they attach to environmental attributes, and to directly state their preferences towards environmental changes.

27 As argued by Garrod and Willis (1999), "the assumptions underlying the [hedonic wage models] of a fixed supply of jobs and a freely functioning job market where individuals chose jobs based on perfect information and with no mobility restrictions […] may not be valid when a shortage of jobs means that that individuals cannot satisfy their demands for environmental improvement because there are no suitable jobs available for them in areas of higher environmental quality" (p.101).

28 In this respect, Kuik *et al.* (1992) have argued that many valuation studies, namely property valuation studies, estimate a general indicator of "urban stress" rather than a well-defined indicator of environmental quality.

29 Besides available general statistical methods, the environmental valuation literature has developed and exploited various *ad hoc* approaches in order to address potential multicollinearity problems in HPM studies (see for example Feitelston (1992), and Powe *et al.* (1997)).

30 When official records of property transactions are not available (or they do not provide reliable) information may be collected from estate agents (see for example Dodgeson and Topham, 1990). However, in countries or regions which have little tradition of estate intermediation, or, more generally speaking, little tradition of selling and buying properties, such as houses, this alternative source of information is unlike to provide adequate data.

Once an appropriate survey instrument (questionnaire) has been prepared –and pre-tested– individuals' "bids" are obtained either by face-to-face interviewing, telephone interviewing, or mail surveys.[31]

CVM, originally proposed by Davis (1963), has been generally used for assisting public decision-making in order to evaluate projects or programs involving positive environmental changes: examples include investigations carried out to estimate the benefits individuals attributed to air pollution abatement in urban areas; to reduced health risks from water contaminants; to protection of wilderness areas and endangered species. CVM has also been applied, although much less frequently, for environmental damage assessments.[32]

The interest in CVM has increased over time: Carson *et al.* (1995) have identified more than 2,000 theoretical papers and applications dealing with the topic.

The vast literature, and the variety and inherent complexity of the methodological and technical issues involved, would suggest it were unwise to attempt to make a summary which risks providing a too narrow and incomplete picture. Given the objectives of this paper, we will then limit ourselves to illustrate the main potential advantages of CVM, and some of the technical issues and potential drawbacks more frequently cited in the literature.

The main potential advantage of CVM, with respect to revealed preference valuation techniques, consists of its potential ability to provide estimates of both use and non-use values, or, using a different taxonomy, of both "direct-use" and "passive-use" values (see Section I.3). In fact, whilst revealed preference techniques measure only environmental services' values which can be inferred by looking at other related marketed goods (i.e., generally speaking, direct-use values), CVM is potentially capable of capturing values, derived from environmental attributes holding quasi-public/public features, which cannot inferred through observable market behaviour.[33]

Leaving aside the debate revolving around whether or not non-use values –or the even broader category of passive use values– should be considered relevant to decision-making, and, in the affirmative, whether they should be monetized or left to the "political arena",[34] [35] much of the

[31] Face-to-face interviewing is generally considered the preferable way to conduct a CVM survey. Telephone and mail surveys are cheaper, but suffer from various potential drawbacks (see Carson, 1999, pp. 12-13).

[32] One of the best known and cited applications of CVM for damage assessments is the study concerning the 1989 Exxon Valdez oil spill (Carson *et al*., 1992).

[33] As noted by Carson *et al*. (2000), CV is not the only technique capable of capturing passive use value. Since "the fundamental problem in the economic valuation of environmental goods is the absence of a market […] any other members of the class of constructed markets […] such an actual referendum on whether to provide the [good] or a simulated market in which the good is actually provided […] can potentially be used for this purpose. The value of [the good] may also be inferred in some instances from voting decisions by political representatives" (p.5); however, one problem with using voting decisions by representatives "is that the vote of one [good] may often be tied to the provision of other goods or in response to the activities of special interest groups" (p. 6, footnote 8).

[34] For a summary of this debate, see Carson *et al.* (2000, pp.2-10). Following the Authors, "three camps hold fundamentally different positions on passive use values. They are: (1) passive use values are irrelevant to decision making, (2) passive use values cannot be monetized, and thus, can only be taken account of as a political matter or by having experts decide; (3) passive use values can be reliably measured and should explicitly taken into account" (p. 4).

[35] The use of contingent valuation, particularly its use for estimating non-use values in (compensable) damage assessment cases, has been strongly criticised by William Desvousges. In a testimony to the U.S. House of representatives (Desvousges, 1995), he stated that "nonuse damages cannot be reliably measured using CV" (p. 9). The Author argued that the common argument for including CV estimates of nonuse damages, i.e. that without it damages will be severely undermined, is fallacious, because it overlooks three significant facts. "First, the restoration of resource services required by [US] regulations, means that only forgone nonuse values during the interim period until recovery is complete are at issue. When restoration is complete, all nonuse values, if they were lost, would be restored […] Second, the concept of nonuse values was originally envisioned for permanent losses of unique resources. This situation has not arisen in most damage assessments. Typically, many substitutes exist for the injured resources, which suggests that any nonuse damages should be small. Moreover, the resources are recovering on their own and through the restoration activities […] Finally […] the transaction costs of trying to measure nonuse damages […] are often greater than the damages themselves. I think that society could be in a worse situation from an economic efficiency point of view by including nonuse damages than if they were excluded" (p. 9).

technical debate over CVM has focussed on the survey design, and on the economic criteria which the results of a CVM application should meet.[36]

One particular source of concern in the CVM literature has been “strategic bias”. This bias may result because the environmental changes for which respondents are required to state their bids (maximum WTP for a positive change, or to prevent a deterioration; or minimum WTA to give up a positive change, or to accept deterioration) often hold quasi public/public features (Hanley and Spash, 1993). Consequently, because the effects of these changes are non-excludable (see Section I.3), respondents may adopt a strategic behaviour in the form of a “free-riding” attitude.

However, as underlined by Carson *et al.* (2000), “the incentive structure for truthful preference revelation is closely related to the CV elicitation format used” (p.26). Progress has been made towards designing elicitation formats (setting up the “hypothetical market”), in order to avoid, minimize, or control the effects of freeriding. As noted by Hanley and Spash (1993), the available evidence tends to suggest that CVM studies are less prone to strategic behaviour than was once believed.

Another major focus of the technical debate has been comparing estimates from CVM surveys, and estimates from revealed preference methods. Available evidence shows that for quasi-public goods, such as outdoor recreation, CVM estimates tend to be lower, whilst for goods holding private features, surveys tend to predict higher hypothetical purchase levels than actually observed (Carson, 1999).

These differences –which may be at least partly explained by the economic nature of the environmental good, and by strategic bias or “hypothetical market error’ problems”–,[37] have led some authors to propose carrying out both a CVM and a revealed preference analysis, so as to acquire estimates which can be cross-checked, in order to get an idea about the robustness of the results (World Bank, 1998). However, cross-checking, besides further increasing the costs of acquiring value estimates, is a suitable option only if a valuation study is targeted at identifying environmental values which may be inferred through revealed preference methods (*e.g.* recreational values, through travel cost analysis).

There are several other relevant methodological and technical issues surrounding the implementation and the use of a CVM study. The interested reader is directed to two recent papers (Carson, 1999; Carson *et al.*, 2000), which provide an excellent literature review, and useful guidance both to CVM practitioners and users of CVM applications’ results.

However, before concluding this brief illustration of the CVM, it is worth drawing attention to the cost of CVM applications, and, consequently, on the impacts that resource constraints could have on the possibility to carry out a proper valuation exercise. Although many (public decision-makers or critics) believe that CVM is an easy even trivial task to ask individuals what they are willing to pay for a good, “a reliable CV survey is neither simple nor inexpensive to implement” (Carson *et al.*, 2000, p.37). Consequently, Carson *et al.* (2000) hold that “at this point in the development of CV, the key objective in terms of methodological development should shift to trying to determine how to reduce the cost of conducting CV studies while still maintaining most of the quality of the very best studies now being conducted. Development and research along these

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[36] As far as the economic criteria the results of a CVM study should meet are concerned, “much of this debate concerns the merits of particular tests and whether various phenomena are anomalies from the perspective of economic theory and, if so, whether they are peculiar to particular studies or CV practices […] or symptomatic of more general problems with CV” (Carson, 1999, p.4). Some of these tests are described by Carson *et al*. (2000), who also provide a literature review of the debate revolving around the consistency of CVM results with economic theory (see, in particular, pp.13-25).

[37] “Hypothetical market errors” are said to occur “if the very fact that respondents are asked for valuations in a hypothetical market makes their responses differ systematically from true values. If the effect leads to both over and under statement, then it is not bias we are faced with, but a random (that is, non-systematic) error term” (Hanley and Spash, 1993, p.61-62).

lines will be crucial in effectively incorporating the public' s preferences into the environmental decision making arena" (Carson *et al.*, 2000, p.37).

7. Environmental value transfer

a) Rationale and potential advantages

The term "environmental value transfer" (EVT) –otherwise known as "benefits transfer"–[38] refers to the process by which a demand function or value for one environmental attribute or group of attributes, obtained (by whatever valuation method) in one context (the *original study context*) is applied to assess environmental values in another context (the *transfer context*).

Using estimates obtained from past studies to evaluate the costs (or benefits) of new projects, environmental regulations, or other policies, is commonplace in public decision-making and benefit-cost analysis, and this approach has been formally recommended and adopted, by various agencies, for the economic valuation of environmental impacts. [39] [40]

EVT is generally advocated on the grounds of resource constraints and cost-effectiveness (Garrod and Willis, 1999). In fact, analysts can only rarely afford the luxury of implementing original studies, and transfer studies may provide an economical way to conduct research when a full-fledged study is not practical or necessary (Desvousges *et al.*, 1998).

Particularly when valuation is aimed at estimating compensable damages, a simplified approach is often motivated by the sensible desire to keep the expenses of investigation under the cost of damages of an event (Gardner, 1993). Furthermore, some events may involve transitory impacts (e.g. biological damages) which become unobservable before a study team is able to visit the affected sites (Garrod and Willis, 1999).

Although EVT is sometimes described as not a methodology *per se*, but simply as the transposition of estimates from one context to another context (World Bank, 1998), as argued by Desvousges *et al.* (1998) transfer studies demand "all the advanced skills required in original research and more […] Transfer analysts must employ great judgement and creativity both in manipulating available information and in presenting results to decision makers. They must also clearly expose the relative roles of data and assumptions, helping decision makers to understand the sources of uncertainties inherent in the estimates" (p.1).

[38] Following Brouwer (2000), the term environmental value transfer is used here instead of the popular term benefits transfer because available valuation methods (and, consequently, value transfer) may be used to estimate either economic benefits or economic costs associated with environmental changes. The more frequent use of the term "benefits transfer" is probably attributable to the fact that the transfer method has been widely applied to evaluate the impacts of regulations and projects aimed at preventing or mitigating harmful environmental changes.

[39] For example, the H.M.Treasury (1991) and the Asian Development Bank (1996) suggest that transferring available estimates can be a feasible approach for many applications, although both advocate caution in the use of such transfer. In particular, the Asian Development Bank recommends special caution when significant cultural differences exist and when the project to be valued is expected to have large environmental impacts.

[40] Value transfer is used in the United States under the CERCLA (Comprehensive Environmental Response, Compensation and Liability Act) to assess damages resulting from spills or accidents in marine and coastal environments. If the injuries are relatively moderate, a set of simplified procedures ("Type A rules") can be used. In particular, damage estimates may be obtained through a large computer model that has physical, biological, and economic components, by submitting incident-specific data, requiring minimum field observation. The sub-model for economic damages produces estimates based on reduced in situ values (Gardner, 1993). "Economic damages under "Type A" assessments are measured to account for injuries to commercial and recreational fisheries; waterfowl, shorebirds, and sea birds; for seals; and public beaches […] The reduction in use value is measured by the change in the value or the cost of harvesting; or from any change in the value of viewing or visiting the resource. Damages to waterfowls, shore and sea birds are assessed from use values for hunting and viewing, based on previous studies of waterfowl hunting and the change in visitor days at wildfowl refuges as a function of changes in bird populations" (Garrod and Willis, 1999 p.332).

However, in spite of fairly widespread use, until quite recently little professional discussion was available on how transferring information and estimates should be done or on the issues involved in developing transferable methods. To our knowledge, the first discussion of value transfer as a process is developed in Freeman (1984), and in 1992 *Water Resources Research* devoted a special issue (volume 28, number 3) to this subject, by collecting a set of papers aimed at defining standards and protocols for transfer studies. More recent contributions, sharing the same objective, include Desvousges *et al.* (1998) and Brouwer (2000).

Before illustrating the major steps and technical issues involved in a transfer study, it is worth briefly addressing the issue of the *level of accuracy* required in transfer analysis, and the relationship between the level of accuracy and the purposes of the transfer.

A transfer study relies upon demand functions or point estimates borrowed from previous studies, and, as stressed by Brookshire and Neill (1992), the transfer can be no more reliable than original findings upon which it is based. Sometimes, analysts must resort to low-quality studies, and nearly always to studies which were not designed with future transfer application in mind (Garrod and Willis, 1999).

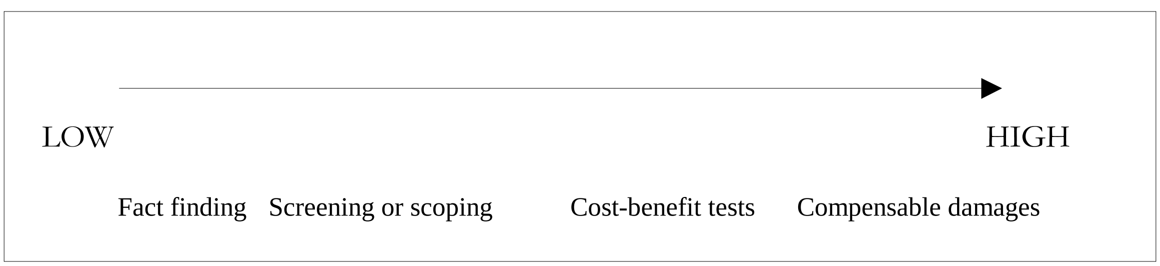
All this implies that transfer analysts must make a number of assumptions, judgements and *ad hoc* adjustments when transposing available estimates.

Although the same could be said for any valuation exercise, "the key question is whether the added subjectivity surrounding the transfer are acceptable, and whether the transfer is still informative. If not, the alternatives are to forego a quantitative analysis [i.e. to forego a monetary valuation of the environmental changes under consideration] or to conduct an original study" (Desvousges *et al.*, 1998, p.10).

To help answer such questions, Desvousges *et al.* (1998) have proposed a stylized continuum (see Graphic 1) which illustrates different possible purposes of the transfer and the level of accuracy required for each.

Graphic 1

A STYLIZED CONTINUUM OF THE LEVEL OF ACCURACY REQUIRED IN TRANSFER STUDIES



Source: Adapted from Desvousges *et al.* (1998)

If the purpose of a transfer study is simply fact finding –such identifying relevant environmental impacts, representative groups of affected individuals, typologies of resources uses and value categories– or a literature review is simply used as a screening tool for guiding the design of an original study, a relatively low level of accuracy is required.

On the contrary, "transfer studies that inform policy decisions, such as benefit-cost tests […] must meet a certain standard of accuracy. However, it is often sufficient if they obtain a bounded result. For example, benefit-cost tests often need only to determine whether or not the benefits are greater than costs; they may not need to establish an exact magnitude […] In contrast, at the highest standard of accuracy, an actual magnitude is required. In environmental economics, this category includes determining compensable damages in damage assessment cases (Desvousges *et al.*, 1998, pp.10-11).

b) Basic steps in a transfer study

As mentioned before, the increasing demand for environmental valuation and the increasing reliance on transfers studies, have provoked calls for standards and protocols. Two of these recommended protocols ("basic steps in a transfer study") may be found in Desvousges *et al.* (1998) and Brouwer (2000).

As illustrated in Graphic 2 and Graphic 3, the two proposed protocols exhibit many similarities and overlaps. However, leaving aside terminological differences, it is worth drawing attention on the emphasis placed by Brouwer on "stakeholder involvement" in various phases of the transfer process which, on the contrary, is not (at least explicitly) advocated by Desvousges *et al.* (1998).

This emphasis derives from what Brouwer (2000) considers as a caveat of the value transfer literature which, in general, does not question the transposition of values in itself, but mostly focuses on technical problems and techniques aimed at improving the "quality" of transfer estimates.

According to Brouwer, although these technical issues –some of which will be briefly illustrated in the next section– are important, they may overshadow more substantial methodological issues. As stated by the Author, "one the underexposed areas in […] environmental valuation so far is the assessment of the appropriateness of different valuation procedures in different environmental domains based on their underlying axioms and assumptions […] Instead of making assumptions a priori, more research efforts should be focused on the processes by which actual public attitudes and preferences towards the environment can best be facilitated and fed into environmental or other public policy decision-making. One way of making sure that the transfer (valuation) exercise generates socially and politically acceptable results is to get the stakeholders involved who are (going to be) affected by environmental change and whose values the researcher and decision-makers(s) are interested in. This stakeholder consultation process provides the researcher with an external valuation exercise and helps define the boundaries of monetary environmental valuation" (Brouwer, 2000, p.148).

Graphic 2

BASIC STEPS IN A TRANSFER STUDY (ADAPTED FROM DESVOUSGES *ET AL*., 1998)

STEP 1

IDENTIFY LINKAGES AND ORIGINAL STUDIES FOR TRANSFER

⇓

STEP 2

OBTAIN BACKGROUND INFORMATION

(E.G. BASELINE ENVIRONMENTAL QUALITY AND SOCIO-ECONOMIC DATA)

⇓

STEP 3

PERFORM PRELIMINARY ASSESSMENT OF COSTS (OR BENEFITS) AND

IDENTIFY NEW AREAS FOR ADJUSTMENT

⇓

STEP 4

TRANSFER EXISTING ESTIMATES OR MODELS AND ESTIMATE EFFECTS AT EACH LINKAGE,

OBTAINING PER-HOUSEHOLD COSTS (OR BENEFITS) IN EACH MARKET AREA

⇓

STEP 5

DETERMINE SET OF HOUSEHOLDS IN RELEVANT MARKET AND

OBTAIN TOTAL COSTS (OR BENEFITS)

Source: Author's elaboration.

Graphic 3

BASIC STEPS IN A TRANSFER STUDY (ADAPTED FROM BROUWER, 2000)

STEP 1

DEFINING THE ENVIRONMENTAL GOODS AND SERVICES

⇓

STEP 2

IDENTIFYING STAKEHOLDERS

STEP 3

IDENTIFYING VALUES HELD BY DIFFERENT STAKEHOLDER GROUPS

⇓

STEP 4

STAKEHOLDER INVOLVEMENT IN DETERMINING THE VALIDITY OF MONETARY VALUATION

⇓

STEP 5

STUDY SELECTION

⇓

STEP 6

ACCOUNTING FOR METHODOLOGICAL VALUE ELICITATION EFFECTS

⇓

STEP 7

STAKEHOLDER INVOLVEMENT IN VALUE AGGREGATION

Source: Author's elaboration.

c) Transfer studies: critical aspects and potential drawbacks

Various problems and potential drawbacks may emerge when conducting a transfer analysis. Some of them are briefly illustrated below, with reference to the three major steps of a transfer exercise: (a) identification and selection of candidate original studies; (b) synthesis of existing information; and (c) transfer of information.

(i) Identification and selection of original studies

Once the analyst has identified the relevant ecological and economic cause-effect relationships which are believed to drive changes in people's welfare resulting from the environmental changes which are expected to occur, or which have actually occurred, at the study site (the "transfer context"), the analyst has to undertake a search in order to identify previous studies that can potentially quantify such changes.

Although a literature search of published studies may reveal some potential candidates, some relevant studies may not appear though normal channels (e.g. working papers or special contractual studies). However, there are several useful bibliographies that include unpublished studies which would otherwise be difficult to find (e.g. Carson et al., 1995), and some databases are also available electronically. For example, a large online database has being compiled by Environment Canada, as a cooperative venture undertaken with the environmental protection agencies of the United States, Chile and Mexico, the World Bank, the European Union, and the Economy and Environment Program for South East Asia (http://www.evri.ec.gc.ca/evri).

Once a literature search or other available sources have revealed potential candidates for transfer, the analyst should evaluate their transferability and select the most appropriate one(s). Several criteria have been suggested to assess existing studies' transferability.

Besides their scientific soundness (Brookshire and Neill, 1992), special attention should be paid to the original studies' relevance: i.e., the original study context and the transfer context should match as closely as possible (Desvousges et al., 1998). In particular: (i) the magnitude of environmental changes and the affected 'environmental commodities' should be similar; (ii) the baseline environmental conditions should be comparable; (iii) the affected populations' socio-economic characteristics should be similar.

Furthermore, analysts should base their decision upon the original studies' *richness of details.*[41]

(ii) Synthesis of available information

Finding studies adequately satisfying the afore-mentioned general criteria may prove difficult. However, if analysts are able to pick up several useful studies, they face the problem of exploiting all the acquired relevant information in an efficient and sensible way.

[41] "To facilitate transferring entire equations, studies would ideally provide precise definitions and units of the variable in the analysis, as well as their means […] At the benefit or cost stage of the transfer, they would ideally include information on the available substitutes [for the environmental goods and services]. They would also reveal participation rates, the extent of the relevant geographic market [(see below, *Transferring information*)] and, in the case of travel-cost studies of recreation demand, report assumptions about the opportunity cost of time" (Desvousges *et al.*, 1998).

The simplest approach consists of using the bundle of selected studies in order to get a range of possible estimates (lower bound and upper bound estimates), or simple descriptive statistics (e.g. the mean and standard error).[42] A more sophisticated approach is to use meta-analysis techniques.

The term meta-analysis approach refers to the statistical analysis of a large collection of results from existing studies for the purposes of integrating the findings. Following Glass (1976), meta-analysis connotes "a rigorous alternative to the casual, narrative discussion of research studies which typify our attempt to make some sense of the rapidly expanding research literature" (p.3).

Meta-analysis should not be seen merely as a mean for producing point estimates to be transferred to a new context. As noted by Bergh et al. (1997) and Garrod and Willis (1999), meta-analysis may be a useful tool for exploring the factors, involved in the construction of environmental values, which have influenced variations within and across individual studies.

However, although several studies have used meta-analysis techniques to synthesize environmental valuation research,[43] meta-analysis is still a relatively underdeveloped field of enquiry, and only rarely do available applications attempt to provide information that is useful for transfers or for transferability assessment (Desvousges *et al.* 1998; Garrod and Willis, 1999).

(iii) Transferring information

After identifying relevant studies and synthesizing available information in some way, the next step consists of transferring such information, in order to get per-capita cost (or benefit) estimates, and, then, aggregated welfare measures. This can require ad hoc adjustments to the available estimates and may entail some arbitrary decisions.

As for the per-capita estimates, the analysts may improve the quality of the transfer by exploiting, on the one hand, secondary data concerning the transfer context's relevant features (baseline environmental conditions; socio-economic characteristics of affected individuals; availability of goods which may substitute environmental services; assignment of property rights),[44] and, on the other hand, information on the differences between the original context and the transfer one.

For computing aggregate welfare measures, the analysts should in principle identify all affected parties (all "stakeholders"). This in turn requires the identification of the environmental change's geographic and economic domain. In valuation literature's jargon, the relevant domain is sometimes described as the extent of the market (the *"relevant market size"*).

The geographical extent of the market cannot be merely deduced from the environmental change's "physical impacts" (*e.g.* the spatial distribution of water pollutants and, consequently, the number of potentially affected households or firms).

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[42] For example, in a recent study aimed at assessing the total economic value of Amazonian deforestation, Torras (2000) exploits previous studies which have focussed on specific forest value categories (direct use, indirect use, and non-use values), and calculates the annual per-hectare economic loss by using the mean of the estimates from these studies. By so-doing, the Author's estimated total annual value of a representative hectare of Amazon rain forest is 1,175 US $ (1993 prices). Although the methodology employed by Torras is quite crude, his paper provides valuable information about a large number of empirical studies, conducted in developed and developing countries (particularly in Latin American countries, like Peru, Ecuador, Costa Rica, Brazil, Mexico, Bolivia), aimed at estimating forest values.

[43] For a literature review, see Garrod and Willis (1999: 347-351); Desvousges *et al.* (1998: 28-36).

[44] Special attention should be paid to the assignment of property rights in the original study's and transfer context. If in the transfer context property rights are held (conventionally attributed to) by the individuals affected by environmental changes, transferred studies should in principle use willingness to accept instead of willingness to pay welfare measures (Desvousges *et al.*, 1998). However, as already noticed (see Section I.2.), WTA estimates are relatively rare in the valuation literature. This implies that, when interpreting the results obtained from exploiting available valuation studies, the analyst should be aware of the measurement errors deriving from possible differences in the assignment of property rights in the original and transfer context.

A careful examination of other factors which may affect the number of individuals actually affected (e.g. availability of substitutes for the polluted water body), and, more generally speaking, a proper understanding of the reasons why changes in environmental conditions may involve welfare losses (or gains) is also required. In fact, the same physical change may affect different people in different ways, depending on how this change affects the flow of environmental services, and depending on individuals' behaviour and preferences.

Obviously, the identification of relevant stakeholders (environmental values held by different stakeholder groups) cannot be left to the final stage of a transfer study. In fact, the selection of candidate original studies will be inevitably guided by the analyst's perception of the environmental values at stake. If, for example, a transfer study is aimed at ascertaining the social costs of an environmental change which is likely to affect only use-values (*e.g.* recreational activities), the analyst will carry on a literature search targeted towards finding studies aimed at estimating these values (*e.g.* travel-cost studies which have focused on similar recreational activities).

Nevertheless, when approaching the "aggregation step" (i.e. the estimation of total welfare impacts), the analyst might discover, or become aware, of other values (and stakeholders) involved, and this might require going back, and trying to find additional, or different information sources, for estimating these values.

Because the geographical extent of the market depends on the values held by the individuals experiencing an environmental change, it follows that the relevant market can have different "sizes" (local, national, international "markets"). As a simple rule of thumb, it can be stated that the market's size is correlated to the nature and the spectrum of values generated by a natural asset: assets providing only use-values (in particular, direct use values) are more likely to have a "local market" whilst the market's boundaries of assets providing (also) indirect-use and/or non-use values (*e.g.* a tropical forest) are generally wider.

It should also be noted that the relevant market's size may, or may not coincide with the political jurisdiction of the decision-making authority on behalf of which a transfer analysis is carried out. In this case, as noted by Desvousges *et al.* (1998), if the decision-making authority is only interested in the impacts within its jurisdiction –or within a specific geographic area falling within its political boundaries– looking only at these impacts may be appropriate. However, "if all affected parties are considered to be relevant, this may be a less accurate approach because the points where [costs or] benefits fall to zero may not necessarily correspond to these boundaries" (Desvousges *et al.*, 1998, p.42).

III. Natural disasters and environmental values

1. The rationale of environmental valuation

The term "disasters" means many different things. They are generally violent or unexpected occurrences, often accompanied by loss of life, material damages, and difficulties for the functioning of society and the economy (ECLAC, 1991).

Besides destroying or harmfully affecting physical (man-made) assets, "human", and "social capital", these occurrences may also seriously influence natural capital. "Environmental damages" (i.e., losses of environmental values) may either occur because of negative environmental (quantity or quality) changes,[45] or because of the (temporary or permanent) inability to exploit environmental services, or the increased costs of service enjoyment.

It is customary to divide disasters into two main groups: natural disasters and man-made disasters.

The former are related to meteorological, geo-tectonic, and biological events (Blaikie *et al*., 1994), and include phenomena such as floods, droughts, hurricanes, earthquakes, volcanic eruptions,

[45] The temporary or long-term environmental changes caused by a natural hazard are not always necessarily negative. For example, volcanic ashes and pyroclastic materials can be highly beneficial for agricultural activities; in some regions, floods are useful to plant growth, by enriching the soil and improving fertility; and tropical cyclones may bring benefits over drought-prone land (Blaikie *et al*., 1994).

mudslides and landslides, pest attacks. The latter group includes violent occurrences which are not natural in origin, such as explosions, fires, oil spills, releases of toxic substances, and collapses of dams.

Although some disasters undoubtedly have a natural origin, like geo-tectonic events, for other violent occurrences it may be difficult to draw a clear-cut borderline between “natural” and “man-made” disasters. For example, it is increasingly argued that some “natural” hazards, particularly those related to extreme meteorological events, are partly attributable to human activities (e.g. the emissions of greenhouse gases), although the dynamics of climate change is so stochastic and chaotic that is often difficult to identify the relationships between anthropogenic emissions and the observable (intensity and frequency of) extreme events. Similarly, some “man-made” disasters, like oil spills, may be partly attributable to natural hazards, such as unfavorable meteorological conditions.

Moreover, whatever the disaster’s origin, its consequences will not only depend on the hazard’s intrinsic severity and intensity, but also on people’s vulnerability and socio-economic resilience to violent and unexpected occurrences. In other words, even when the events which drive the disaster have a natural origin, their ultimate effects upon people’s welfare will generally depend on man-made physical, economic, and social “infrastructures”.

A proper understanding of the factors which determine a disaster’s effects, in terms of people’s welfare, is then important both in identifying and evaluating prevention measures aimed at reducing people’s vulnerability, and in determining, after each disaster, the type and amount of damage experienced by affected populations.

As long as a disaster affects, *inter alia*, environmental service flows, or individuals’ service enjoyment opportunities, “environmental valuation”, as defined and described in the previous sections, may contribute to improving the comprehensiveness of natural disasters’ socio-economic impact assessment. In fact, failure to account for disasters’ impacts upon environmental values may seriously undermine the reliability of *ex ante* cost-benefit tests, or the reliability of *post*-disaster damage estimates.

However, although the incorporation of environmental values is likely to improve (*ex ante* or *ex post*) damage assessment’s comprehensiveness, attention should also be paid to the opposite risk, i.e. to potential double-counting problems. These problems may arise when some environmental values, affected by a natural hazard, are already (implicitly) incorporated in other damage categories used to assess a disaster’s welfare impacts.

2. The *ECLAC’s manual*: classification of the effects of a natural disaster and criteria for evaluating damage

In 1991, the UN Economic Commission for Latin America and the Caribbean has published a *Manual for Estimating the Socio-Economic Effects of Natural Disasters* (henceforth, the *Manual*) which intends to be a tool aimed at professionals engaged into the valuation of natural disasters’ socio-economic impacts (ECLAC, 1991).

The *Manual* provides a classification of the effects of a natural disaster: (i) effects on property (direct damage); (ii) effects on goods and service production flows (indirect damage); and (iii) effects on the behaviour of the main macroeconomic aggregates (secondary effects).

Direct damage, which more or less coincide with the disaster or occurs within hours of it, is defined as “all damage sustained by immovable assets and inventories […] and comprises […] total or partial destruction of physical infrastructures, buildings, installations, machinery, equipment, means of transport and storage and furniture, and damage to cropland, irrigation works and dams" (ECLAC, 1991, p.12).

Indirect damage, beginning almost immediately after the disaster and possibly extending into the rehabilitation and reconstruction phase, "is basically damage to the flows of goods that cease to be produced or the services that cease to be provided during a period of time beginning almost immediately after the disaster and possibly extending into the rehabilitation and reconstruction phase […] Indirect damage is caused by direct damage to production capacity and social and economic infrastructure. [It] also includes the costs or increased costs of providing services as a result of the disaster, and losses of income as a result of the impossibility of providing such services (which will, in turn, reflected in the secondary effects)" (ECLAC, 1991, p.13).

The *Manual* provides various examples of indirect effects measurable in monetary terms, such as increased overheads as a result of the destruction of physical infrastructures or inventories; losses of production and income; increased costs because of the need to use alternative means of production (e.g. because of having to use road diversions); or the cost of health campaigns to prevent epidemics. The *Manual* also mentions indirect effects "which could be measured in monetary terms were it not for the time pressures on the analyst" (ECLAC, 1991, p.14). These include, *inter alia*, "environmental changes" (ECLAC, 1991).

Finally, *secondary effects* are defined in the *Manual* as the disaster's impact on the behaviour of macroeconomic variables (GDP, balance of trade, level of indebtedness, foreign reserves, etc.). "Their measurement complements the measurement of direct and indirect damage, since it is carried out from a different standpoint. Secondary effects reflect the impact of direct and indirect damage and must not be added to it" (ECLAC, 1991, p.15).

As far as the criteria for evaluating direct and indirect damages are concerned, the *Manual* provides detailed guidelines for various natural disasters' potential impacts. However, leaving aside the details, the recommended general evaluation criteria may be summarized as follows.

As far as *direct damage* is concerned, "it is advisable to value at equivalent replacement cost totally destroyed capital stock or buildings earmarked for demolition. This involves taking into account the functional equivalence of the destroyed capital asset, in other words, the cost of replacing it with other stock offering similar operating characteristics". On the other hand, "*indirect damage",* [46] to flows of goods or services will be evaluated at producer or market prices, as appropriate" (ECLAC, 1991, p.21).

3. Natural disasters and environmental values: a tentative taxonomy

As underlined in Section I, an environmental attribute (or a group of environmental attributes) does not have an economic value *per se*. It has a value only if it enters at least one individual's utility function or a firm's production function. If so, a natural resource's total economic value may be, in principle, decomposed into use-values and non-use values. Obviously, the relative weight of these value elements vary across resources, and over space and time, depending on people's perceptions, behaviour, and preferences.

From an operational point of view, it may then prove useful to address the valuation of a natural disaster's impacts upon environmental values, by decomposing them into impacts upon use-values and impacts upon non-use values.

46 Italics added by the author.

Impacts upon use values stem from alterations of the (net) benefits derived by exploiting environmental attributes, usually in conjunction with other economic goods. Broadly speaking, natural hazards may alter these benefits in two different ways:

(i) by inducing temporary or permanent environmental (quantity or quality) changes thus altering a natural asset's "intrinsic productivity";

(ii) by altering people's "ability to use the environment"; this typically occurs when man-made capital's partial or total disruption impedes, or make it more costly, to exploit environmental attributes entering firms' production functions, households' production functions, or both.

For the sake of convenience, we term the former category of natural disasters' impacts upon use values direct environmental damages, while the latter one indirect environmental damages.

Examples of direct environmental damages include soil erosion caused by floods; watercourse diversion; losses of natural habitats, such as forests or wetlands –generating direct, indirect, single, or multiple-use values– caused by hazards such as landslides, volcanic eruptions, or coastal storms.

Examples of indirect environmental damages include the disruption of water-distribution networks or water-treatment facilities caused by an earthquake, harmfully affecting water-related use values (*e.g.* loss of agricultural or industrial production; increased health risks or increased public/private averting expenditures). Or the disruption of communication networks and means of transports (like roads, bridges, ports, airports), could make it temporarily impossible to carry out productive and commercial activities entailing the use of environmental goods and services, or impede non-commercial recreational activities.

As far as non-use values are concerned, since, by definition, they arise from the psychological benefits people derive from the mere existence of a natural resource (and/or from intergenerational equity concerns) –i.e. these values are not generated through carrying out an activity involving other economic goods (namely man-made capital)– they can be affected by a natural disaster only if it entails environmental changes (i.e. direct environmental damages).

In short, a natural hazard may affect environmental values in two ways. Directly, by inducing environmental (quantity or quality) changes affecting use values and/or non-use values (direct environmental damages). Or, indirectly, by affecting people's "ability to use" environmental attributes (indirect environmental damages).

4. Intersections between environmental values and ECLAC' s damage categories

As underlined in Section III.2, the *Manual* provides a conceptual and operational distinction between natural disasters' *direct* and *indirect damages*. The former include damages to physical assets. The latter encompass welfare impacts related to changes in the supply (or in the supply cost) of marketable goods and services, as a result of man-made capital's (partial or total) disruption.

On the other hand, in the previous Section, we have provided a tentative taxonomy of natural disasters' impacts upon environmental values ("environmental damages"), by making a distinction between use and/or non-use values affected by environmental changes (direct environmental damages), and use values affected by changes in the ability to exploit environmental attributes (indirect environmental damages).

Although drawing a clear-cut borderline between direct and indirect environmental damages may sometimes prove difficult, we believe this distinction could be exploited as a sort of conceptual device in order to identify appropriate welfare measures and valuation approaches, as well as to identify possible overlaps between natural disasters' environmental damages and other damage categories.

Our notion of direct environmental damage is broadly similar to the ECLAC's definition of *direct damage*: whilst the latter encompasses the physical effects upon man-made capital, the former is aimed at encompassing a disaster's effects on natural capital (impacts upon natural resources' "intrinsic productivity").[47]

Both physical effects result in changes in (man-made or natural) capital's service flows, and, by so-doing, affect people's welfare. In principle, the welfare impacts should be evaluated by looking at the (present) value of decreased capital's "dividends", i.e. decreased benefits attributable to these changes over time (see Section III.7).

Alternatively, as a proxy of the "true" welfare cost, the capital's restoration cost can be used as a measure of damage, provided restoration is feasible, and the analyst believes that the economic cost incurred in restoring the asset to its original state is not greater than the benefits the damaged/destroyed assets provide(d).[48]

This is the valuation approach recommended in the *Manual* for assessing the costs of man-made capital disruption. However, in order to avoid the underestimation of damages, the *Manual* also recommends taking into account the changes in the flows of marketable goods and services (*indirect damage*), attributable to the disruption of physical assets, which occur until the assets' rehabilitation or reconstruction.

However, extending *sic et simpliciter* this valuation approach to natural capital –i.e. using natural capital's restoration cost as a proxy of direct environmental damages– may be more problematic. This because (i) the restoration of a natural asset's original "productivity" may be technically unfeasible; (ii) when technically feasible, the natural capital's rehabilitation and restoration phase may last longer than the average time required to restore man-made capital ('s services).[49]

Turning now on our notion of indirect environmental damages, since it is intended to encompass welfare losses not attributable to natural capital's disruption, but losses attributable to changes in people's "ability to use" potentially available environmental services, some of these damages could be masked by other ECLAC's damage categories, namely under *direct damage*.

In fact, indirect environmental damages typically arise because of the partial or total disruption of other forms of capital, like physical infrastructures. In principle, the welfare costs of capital's disruption should be, in principle, evaluated by looking at the losses in the capital's service flows. If evaluated in this way, the welfare costs would (should) also include the loss of benefits people will experience as a result of the impossibility to exploit environmental attributes, or as a result of the increased exploitation costs.

However, the *Manual* has made a choice in favour of man-made capital's restoration cost, as a proxy measure of a disaster's effect upon physical assets. As already noted, this operational choice underlines the assumption that restoration will allow the recovery of benefits whose values are (believed to be) at least equal to restoration cost. Consequently, man-made capital's restoration-

[47] The term "productivity" is used here in a broad sense, in that it refers to the "production" of people's welfare. If interpreted in this way, losses of natural capital's productivity do not only occur when a disaster affects environmental attributes entering firms' production functions, but also when a disaster affects attributes entering individuals' utility functions, by so doing altering people's "quality of life" (*e.g.* reduced recreational activities, increased health risks, losses of non-use values).

[48] See Section II.3 (*Restoration cost approach*).

[49] In the *Manual*, "the rehabilitation and restoration phase […] has been set at a maximum of five years [although] any calculations of its effects should, in any case, extend to the period needed to restore all or part of production capacity" (ECLAC, 1991, p.13)

cost measures (*direct damage*) should also be interpreted as a proxy of the indirect environmental damages attributable to man-made capital's disruption.[50]

Obviously, if man-made capital's restoration is not immediately carried out, or it does not allow the immediate recovery of people's ability to use natural resources, the indirect environmental damages, occurring during the restoration phase, should be added to the man-made capital's restoration cost, unless they are already implicitly accounted for in *indirect damage.* This occurs when indirect damage already include disasters' effects, such as losses of marketable outputs and income, or the costs of health campaigns to prevent epidemics, which arise because of people's temporary inability to exploit environmental attributes during man-made capital's restoration phase.

To summarize, there are undoubtedly intersections between ECLAC's damage categories, and a disasters' effects upon environmental values which, for the sake of convenience, we have termed direct and indirect environmental damages (see Graphic 4).

In fact, both ECLAC's notions of *direct* and *indirect damages* may potentially encompass either some of the welfare effects arising from natural capital's "disruption" (direct environmental damages), or some of the effects resulting from changes in people's ability to use potentially available environmental services (indirect environmental damages).

It follows that attention should be paid to avoiding double-counting problems, which could emerge by treating environmental damages as a separate "value component" of a disaster's socio-economic impacts.

On the other hand, the emphasis placed by the *Manual* upon losses of marketable goods and services –as a result of capital's disruption– tends to provide a too narrow definition of a disaster's *indirect damage*. Although this emphasis can be defended on the grounds of the technical difficulty of incorporating many use values, not to say non-use values, in damage estimates, the price paid, in terms of damage assessment's comprehensiveness, by only including environmental damages which manifest themselves through changes of marketable outputs, could be relatively high.

5. The choice of environmental valuation method

Adherence to the restoration-cost approach, as advocated in the *Manual*, in order to assess a disaster's economic effects upon man-made capital, and extending this approach to natural capital, do not rule out the problem of assessing environmental damages occurring during the (man-made or natural) capital's restoration phase. Although, when feasible (and actually carried out), restoration rules out the need to estimate long-term, "irreversible" environmental damages, if restoration does not allow to immediately recover people's ability to use the environment (indirect environmental damages), or the environment's intrinsic productivity (direct environmental damages), the welfare losses occurring during the interim period should be identified, and possibly measured, in order to get a comprehensive damage assessment.

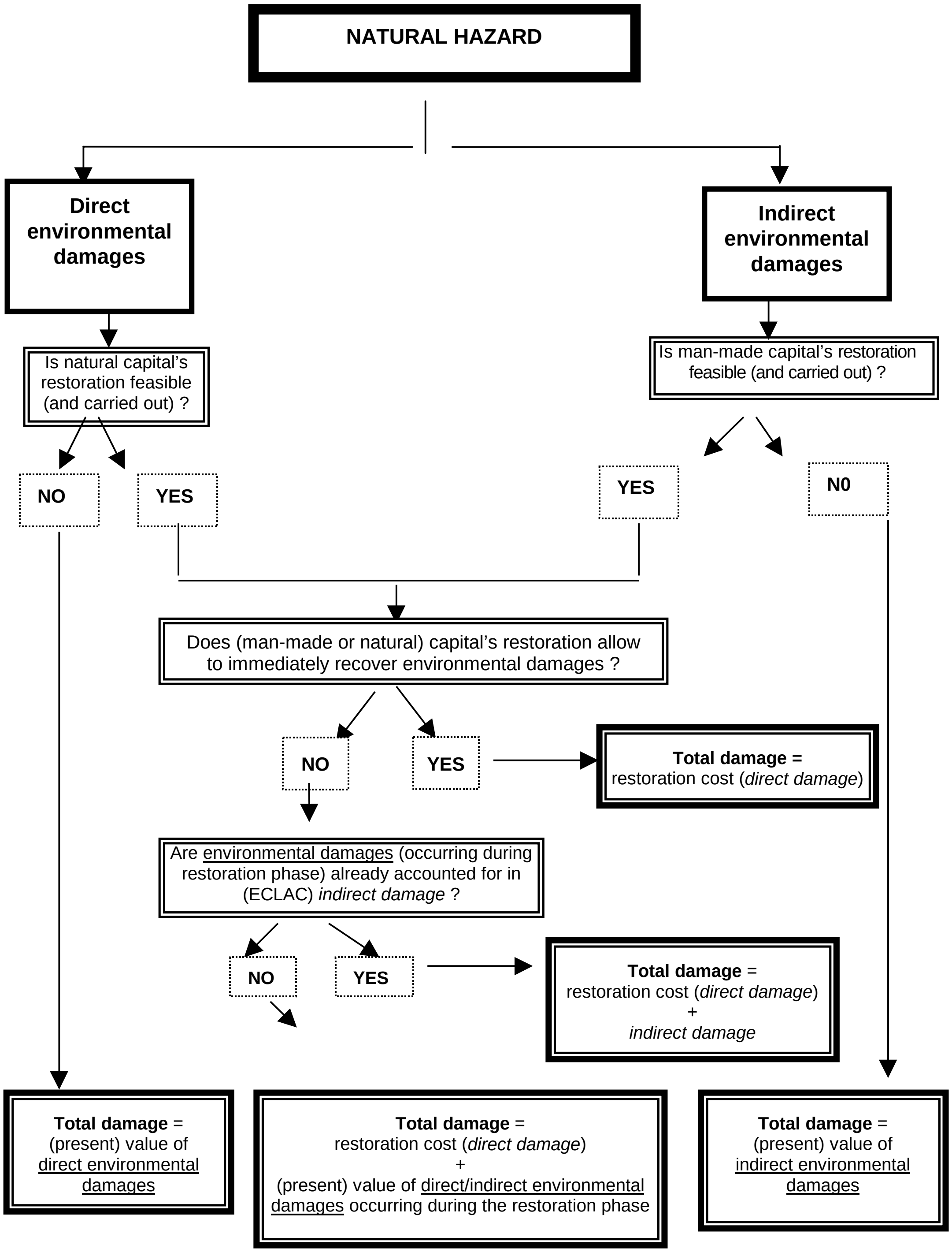
[50] To clarify this statement, let's suppose that to exploit (i.e. to derive economic benefits, B_t , from) an environmental attribute (ENV, *e.g.* surface water of a given quality), a physical asset (K, *e.g.* a water distribution network) is required:

$$B_t = f(ENV,K)$$

where B_t is the per unit of time of, say, agricultural production's economic value. For the sake of simplicity, let's assume that if K=0, then B_t=0 (alternatively, it could be assumed that if K=0, crop production can still be carried out, but farmers need to exploit an alternative environmental input, say groundwater, by affording higher costs). Let's now assume that a natural disaster does not affect ENV, but it involves the disruption of the distribution network (K=0). In principle, the economic damage should be assessed by looking at the (present) value of lost benefits (PV). Alternatively, the disaster's economic cost can be evaluated by looking at the network's restoration cost (C). Assuming the investment in restoration is "instantaneous", restoration is economically efficient if C≤PV. In this case, if restoration is actually carried out, and it allows previous economic benefits to immediately recover, adding PV ("indirect environmental damage") to C (*direct damage* in ECLAC's jargon) would imply an overestimation of the disaster's economic damages. If C>PV, in principle, restoration should not be carried out; however, if restoration is carried out, C still provides an indirect measure (actually, an overestimation) of PV, and, again, adding PV to C would imply double-counting.

Graphic 4

ENVIRONMENTAL DAMAGES AND ECLAC'S DAMAGE CATEGORIES



Source: Author's elaboration.

Generally speaking, the choice of estimation technique to evaluate natural disasters' impacts upon environmental values depends on a number of criteria and circumstances.

It obviously depends on the purpose of the valuation study (*ex ante* appraisal of mitigating measures, or *ex post* damage assessment) and on the level of accuracy required.

The choice also depends on the environmental value categories involved. In fact, different types of environmental services have different types of values attached to them: (direct, indirect, single, multiple) use values and/or non-use values.

As illustrated in Section II, most of the available valuation methods are intrinsically unable to estimate all these value categories. In particular, the techniques assuming "weak complementarity" (such as travel cost or hedonic pricing) are unable to provide non-use value estimates, and some of them are "specialized" in the evaluation of particular use-values (e.g. travel cost: recreational values; hedonic pricing: values arising from the environmental features of a property's neighbourhood; averting behaviour: values related to health risks, etc.)

Similarly, the choice of technique will depend on the economic nature of the affected environmental services (private, quasi-private/public, public services), and/or the economic nature of the related goods involved in generating use values. As underlined in Section I.4, the value of unmarketable environmental services, underlying use values, may be estimated through surrogate market valuation methods only if: (i) these services enter a firm's production function (see Section II.2.), or (ii) households' demand for these services can be inferred by looking at related marketed goods (see Sections II.3-II.5).

Last, but not least, the choice of valuation technique will be inevitably affected by the technique's intrinsic data intensity, and by the availability, or the possibility of acquiring, at a reasonable cost and in due time, the required information.

As far as the informational constraints are concerned, the situation varies according to the valuation context. Differences exists between developing and developed countries, and, perhaps more importantly, differences exist between countries which are more inclined to environmental valuation, and countries which have little tradition (and interest) of incorporating environmental impact assessment into public decision-making processes.

Moreover, the constraints vary according to the purpose of the valuation study. If the aim is to carry out an *ex ante* valuation of natural disasters' mitigating measures, and analysts do not face too stringent resource constraints, they could try to implement a properly targeted valuation study, in order to incorporate environmental value estimates into project appraisal.

However, if the aim is to carry out a *post*-disaster damage assessment, analysts are undoubtedly more unlikely to be able to or allowed to afford the luxury of implementing a full-fledged original study, and the exploitation of previous valuation studies and available value estimates may constitute the only viable option.

Moreover, when dealing with *post*-disaster assessment, besides resource constraints, reliance upon previous studies and available estimates may be justified on the grounds of the technical difficulty of acquiring adequate and comprehensive information about a disaster's environmental impacts. In fact, as already underlined in Section II.7, some events may involve transitory environmental changes which become unobservable before a study team is able to visit the affected sites.

Although the *environmental value transfer method* appears to be a natural candidate for estimating *post*-disaster impacts upon environmental values, special attention should be paid when importing estimates, borrowed from previous valuation studies, into a natural disaster's impact assessment.

The first word of caution concerns the avoidance of double-counting problems. As underlined in the previous Section, some disaster-related environmental damages may already be implicitly captured in other damage estimates. In particular, besides capital's restoration cost (*direct damage*), losses of environmental values may be already embedded in *indirect damage* estimates.[51]

Other elements of caution derive from the intrinsic, general potential drawbacks of the environmental value transfer approach. In particular, as already stressed in Section II.7, special attention should be paid to the original studies' relevance: i.e., the original study context and the transfer context (the "natural disaster's context") should match as closely as possible. In particular: (i) the magnitude of environmental changes and the affected environmental attributes should be similar; (ii) the baseline environmental conditions should be comparable; and (iii) the affected populations' socio-economic characteristics should be similar.

The need to rely upon relevant valuation studies undoubtedly reinforces the arguments in favour of developing, and expanding the scope of *ex ante* economic analysis of natural disasters' mitigation measures. In fact, *ex ante* studies, conducted for countries or regions which are more likely to be exposed to (specific types of) natural disasters, could supply, as a by-product, value estimates which could be subsequently exploited in damage assessment. In turn, *post*-disaster investigations could provide insights for improving the quality of estimates used in *ex ante* valuation studies.

6. Damage for whom? The natural disaster's relevant market size

When considering only property damages, or damages which manifest themselves through changes in marketable goods, the identification of affected persons is relatively straightforward: the natural disaster's geographical boundaries coincide with the area in which these "material losses" are detectable.

However, when considering a disaster's impacts upon environmental values, the identification of the "victims" may be much more problematic.[52]

These problems arise from the fact that a natural hazard may affect environmental attributes which do not exclusively enter the production (or utility) functions of people experiencing *direct* or *indirect damages*, as defined in the *Manual*.

Generally speaking, when considering environmental values, the "relevant market size" is correlated to the spectrum of values a natural asset generates: whilst assets only providing goods and services underlying direct-use values tend to have a "local market", the market size of assets

[51] In fact, if a natural disaster involves changes in the flows of marketable outputs, totally or partially attributable to natural capital's disruption (or to changes in people's ability to use environmental attributes) , ECLAC's *indirect damage* estimates, based upon the "market value" of these changes, can be interpreted as an application of the "production-function approach" (see Section II.2).

[52] The term "victim" and "affected person" are used as synonymous in the *Manual*, which makes a distinction between "primary", "secondary" and "tertiary" victims. "*Primary victims* and homeless persons are those in the population segment affected by the direct effects of the disaster and includes the dead, injured and crippled (the primary trauma victims) and those who suffered material loss, including those accruing from production and income losses, as a direct and immediate consequence of the disaster […] This segment of the population is that found within the territory affected in the moment in which the disaster occurs" (ECLAC, 1991, p.36). "Those population segments which suffer the indirect effects of the disaster are secondary and tertiary victims. The difference between the two groups is that the *secondary victims* are found within the boundary of the affected territory (or very near) and the *tertiary victims* re found outside or far from it" (p. 37). (Italics added by the author).

providing services underlying indirect-use values and/or non-use values is wider, and may go well beyond the community experiencing detectable “material losses”.

It follows that when trying to incorporate environmental values into natural disasters’ socio-economic assessment, analysts face a strategic problem, which cannot be simply solved on empirical grounds, because it may require a “political” decision: who cares about the environmental changes a disaster produces? Which impacts upon stakeholders of natural assets (‘ services) have to be accounted for in damage assessment?.

If damage assessment is aimed at evaluating all welfare impacts of a disaster, wherever they might occur, the empirical problem the analyst faces is to identify the points where environmental damages fall to zero. These may prove very difficult when a natural asset provides services, holding public features, of national or international significance.

On the contrary, if the aim is to assess the disaster’s welfare impacts occurring within a given area (a specific “jurisdiction”), the analyst faces the following problem: which environmental values, besides those related to private environmental services or services holding “local public features”, should be accounted for?.

As far as the latter decision is concerned, the answer cannot be univocal, in that it partly depends on the assignment of *property rights* to environmental values.

As underlined in Section I, many environmental services, particularly those underlying indirect-use values and non-use values, hold public features. This means that, besides being non-rival, the benefits flowing from these services cannot be withheld by the “owner” of the natural resource providing these services (say, the region or the country “hosting” the resource, *e.g.* a tropical forest).

However, despite the technical non-excludability, some services’ beneficiaries might be willing to contribute to natural resources’ conservation, if they perceive that a complete “free-riding” attitude could harmfully affect resources’ service flows, because of the lack of adequate conservation incentives.

This phenomenon does not represent an hypothetical scenario. For example, the Global Environmental Facility (GEF) program –undertaken by the World Bank, the United Nations Development Programme, and the United Nations Environment Programme– was created to provide grant financing to countries to undertake activities that generate global benefits but which are not in the country’s direct interest (Dixon and Pagiola, 1999).

This tendency towards encouraging the provision of off-site environmental benefits is likely to continue and expand, because it appears to be the only effective strategy for filling the gap between the increasing international demand for natural resource conservation, and the lack of local/national incentives to undertake conservation activities except insofar as they generate *appropriable benefits*. This process could be further speeded up by initiatives such as the establishment of an “international market for carbon emission permits”: in particular, developed countries could be willing to pay for activities undertaken abroad which allow to increase carbon sequestration, so as to fulfil their abatement targets.

Grant financing to countries holding natural resources generating global environmental benefits is, *de facto*, equivalent to (conventionally) assigning these countries a sort of “property right” on these external benefits. Consequently, similarly to traditional marketable goods, such as oil or minerals, these resources may become a source of additional revenues, and resource damage could imply an actual, or potential, financial loss.

It follows that, even if damage assessment is only aimed at evaluating the impacts affecting the country (or region) directly experiencing the effects of a natural disaster, if the hazard undermines a country's ability to take advantage of the international willingness to pay for external environmental benefits, this economic loss should, in principle, be accounted for in the disaster's damage estimate.

7. Time and discounting

If natural resources are interpreted as economic assets, whose value stems from their service flows, environmental damages calculated for a period of one year do not measure total damages. On the contrary, the essence of damage assessment is to determine the change in a natural asset's economic value, i.e. the impacts of a natural disaster upon the asset's (expected) service flows.

This requires: (i) identification of the starting point at which loss of environmental services commenced, and the future point at which loss will cease, (ii) estimation of the annual welfare losses, and (iii) the choice of a discount rate.

Whilst "from the standpoint of economic analysis, the time at which damages should first be measured is when they first occurred, and consideration of the monetary value of damages should cease when loss in service stops" (D'Arge, 1993, p.252), choosing an appropriate *discount rate*, particularly when dealing with long-term environmental damages is much more problematic.

In fact, despite the fact that the use of discounting for collective decision-making has been a topic of extensive theoretical debate [53], disagreements continue, and still "the search for a "perfect" formula to specify the social time preference rate [appears to be] futile (Feldstein, 1964, p.247). All we know is that, given an expected time stream of environmental damages, the higher the discount rate, the lower the estimated present value of total damages will be.[54]

The difficulty of selecting an appropriate discount rate may be partly circumvented if –as advocated in the *Manual* for man-made capital - a "political" choice is made in favour of restoring the natural capital's productivity, provided rehabilitation is technically feasible and actually carried out. Besides overcoming the difficulties in valuing fully environmental damage, using the

[53] This issue has been highly debated, particularly in the cost-benefit analysis literature. With few exceptions, the discounting process of future costs and benefits has been accepted as a logical analytical tool, in that using a positive discount rate appears to be consistent with pure time preference rate: when choosing between present and future consumption, individuals usually show a willingness to pay more for immediate consumption than for deferred consumption. Consequently, much of the debate – which has benefited from the contributions of notable economists, including Marglin (1963), Feldstein (1964), Sen (1967), and more recently Arrow *et al.* (1996) - has not focussed on the discounting process in itself, but on whether or not market-based time preference rates ("interest rates") should be used in collective decision-making, and how "social" rates should be measured. The environment and natural resources literature has enriched this debate, either by requestioning the fundamental arguments for discounting (*e.g.* Ferejohn and Page, 1978), or by providing additional arguments for adjusting the "conventional" social discount rates, in order to better reflect the interests of future generations, and/or to encompass environmental "risks" and "irreversibilities". While there are authors who have even argued that "if an economy faces increased production possibilities in the future because of depletion of non-renewable resources, government [should] favour more heavily the consumption of future generations […] by using a negative real social discount rate" (Just *et al.*, 1982, p.306), the majority seems to be in favour of lowering (positive) discount rates, particularly when dealing with public projects involving significant long-term environmental impacts.

[54] Let us indicate with B_t the annual estimated environmental damage induced by a natural disaster. Let us assume that annual damages are not expected to vary over time, and will last indefinitely ($B_t = B$; $t=0, \ldots \infty$). It follows that, by indicating with r the (constant) discount rate, the present value of total environmental damages (PV) will be equal to B/r. For example, if B=$1000, if the appropriate r is thought to be 5%, PV will be $20,000, whilst if the appropriate rate is thought to be 10%, PV will fall to $ 10,000.

restoration cost as a conventional measure of damage is broadly consistent with the "sustainability criterion" in favour of maintaining natural assets' productivity "intact".[55]

However, using the restoration cost as a proxy of environmental damages may involve an overestimation or underestimation of the disaster's "true" social cost.[56] This may occur if the timing of restoration, and the welfare losses occurring during the rehabilitation phase, are not appropriately accounted for in damage assessment.

In fact, if restoration is not immediately carried out –and/or is diluted over time– the estimated restoration cost should be discounted back to the present, i.e. it should be multiplied by an appropriate discount factor, in order to avoid an overestimation of the disaster's social cost. On the other hand, if restoration is "immediately" carried out, but it does not allow environmental services to recover at once, the welfare losses occurring during the *interim* period should be added to the restoration cost. If these losses were ignored, reference to the restoration cost alone would underestimate the disaster's social costs.

To clarify these issues, let us consider three alternative scenarios. First, let us assume restoration (at a total cost *C*) is immediately carried out (t=0), but it allows the natural asset's productivity to recover only at time t=n (n>0); during the *interim* period, the affected people are expected to experience the following annual welfare losses: B_t (t=0, …n). In this case, the natural disaster's social damage (*SD*) will be:[57]

$$SD = C + \sum_{t=0}^{n} \frac{B_t}{(1+r)^t} \qquad \textbf{(5)}$$

Let's now assume that restoration will be carried out at time t=n, and, once carried out, it will immediately allow the recovery of natural asset's productivity. In this case:[58]

$$SD = \frac{C}{(1+r)^n} + \sum_{t=0}^{n} \frac{B_t}{(1+r)^t} \qquad \textbf{(6)}$$

Finally, let's assume restoration will be carried out at time t=n , but the natural asset's productivity will not be recovered until t=n+s (s>0). In this case:

$$SD = \frac{C}{(1+r)^n} + \sum_{t=0}^{n+s} \frac{B_t}{(1+r)^t} \qquad \textbf{(7)}$$

55 According to Markandya and Pearce (1994), although arguments for lowering conventional discount rates do contain persuasive elements, adjusting the discount rate for evaluating public projects involving environmental impacts, "is a clumsy way of handling these legitimate concerns" (Markandya and Pearce, 1994, p.46). They argue that the problem of intergenerational justice – one of the main focuses of the "environmental critique" to the discounting process- could be best addressed by other means, namely by imposing a "sustainability constraint", i.e. by requiring that no project should reduce the stock of natural resources unless there is some compensating increase elsewhere. According to Markandya and Pearce, the cost of the compensating measures should be included as a cost in the project that generates the initial effect on natural capital. "However, project cost and benefit flows will be discounted at the unadjusted opportunity cost rate" (Weiss, 1994, p.11).

56 We leave aside the issue whether or not restoration is economically efficient, i.e., put differently, we assume that the social value of resources required for restoring a natural asset's productivity is not greater than the future welfare costs which would emerge without restoration (see footnote 50).

57 We assume that *C* refers to a cost-effective restoration investment, and that the investment's only positive social effects are in terms of removing environmental damage.

58 We assume that postponing restoration does not involve a change in restoration cost. If the restoration cost is expected to increase because of inflation, it should be appropriately deflated before multiplying it by a discount factor determined by using a real discount rate. More generally speaking, if all elements used for the computation of SD (*C* and B_t) are expressed in real terms, by appropriate r we mean an appropriate real discount rate; otherwise, unless these elements are appropriately deflated, a nominal discount rate has to be used for computing SD.

If either the investment in restoration is postponed or it is immediately carried out, but it does not allow the environmental services to immediately recover, an appropriate discount rate (*r*) has to be identified in order to compute SD. However, if *n* (or *n+s*) is relatively "small" –and people experiencing welfare losses during the restoration phase can be broadly identified– selecting an appropriate *r* appears to be less conceptually difficult than selecting a discount rate when facing long-term environmental damages involving time horizons extending over multiple generations.

In fact, if restoration can be essentially complete, and the rehabilitation phase does not last "too" long, most of the conceptual issues surrounding the discounting process –such as intergenerational equity concerns, uncertainty about future preferences and welfare losses, as well as uncertainty about the discount rate itself–[59] are ruled out, in that, under such circumstances, restoration would neutralize long-term environmental damages.

As argued by D'Arge (1993), for "near-term future damages", the desired rate would be the one for the damaged group. If our interpretation is correct, in practical terms this means that, when dealing with "short-term" environmental damages, professionals engaged in natural disaster damage assessment should use a "standard" (social) discount rate, i.e. the rate used by cost-benefit analysts for evaluating public projects affecting a similar community, over a similar time-span.

Obviously, the longer the restoration phase becomes, the more difficult it becomes to identify an appropriate discount rate. In this case, if it were only for the difficulty of estimating accurately long-term damages, adherence to a "precautionary principle" would dictate adjusting the "standard" discount rate, by lowering it in order to encompass uncertainty about future welfare losses stemming from irreversible, or almost irreversible environmental damages.

How "low" the discount rate, when dealing with long-term damages, particularly with irreversible losses of services flowing from "unique" natural assets, should be, is difficult to say. Since the search for a perfect formula appears to be futile, some arbitrariness appears to be inevitable.

What matters, however, is the transparency and coherence of the overall damage assessment process. This implies, on the one hand, that the assumptions behind the choice of adjusting the discount rate should be spelled out (*e.g.* the assumptions about future preferences, and technological changes' impacts upon natural resources' use and substitutability). On the other hand, the discount rate's adjustment(s), for addressing specific long-term or irreversible environmental damages, should be consistent with the choices made for evaluating other dimensions of a natural disaster's socio-economic impacts (namely, impacts attributable to man-made capital's disruption).

[59] Whilst the traditional debate on discounting has focussed, *inter alia*, on the uncertainty about future economic impacts (say, about future environmental damages), and various authors have argued that such uncertainty would justify lowering the discount rate (for a critique, see Markandya and Pearce, 1994, pp. 36-41), some recent contributions have focussed on the uncertainty about the discount rate itself (Weitzman, 1998; Pizer, 1999; Newell and Pizer, 2000). Through simulations, Pizer (1999) shows that uncertainty about future discount rates leads to the use of lower-than-average effective rates. The decline in the effective rate is especially dramatic as t becomes large (Weitzman, 1998). According to Newell and Pizer (2000), these results have "potentially huge implications for the valuation of benefits in the distant future, such as those associated with mitigation of climate change, long-lived infrastructure, reduction of hazardous and radioactive waste, and biodiversity benefits that are discounted to a pittance when the discount rate is treated as if it is exactly known" (pp.3-4).

IV. Summary and conclusions

The aim of this paper was to contribute to the *ECLAC-Manual*'s revision process, by illustrating the concept of "environmental values" from an economic perspective, and by making a preliminary attempt to identify a strategy for incorporating "environmental damages" (i.e. "losses of environmental values") into natural disasters' damage assessment.

We have argued that expanding the scope of damage assessment, in particular, through the incorporation of environmental damages, does not require a substantial re-definition of the present *Manual*'s categorization of natural disasters' socio-economic damages. Rather, ***it requires: (i) a broadening of the notion of "capital" available to a society, and (ii) interpreting economic damage as that which is the disaster's impact upon capital's service flows.***

This approach appears to be *inter alia* consistent with the "sustainability literature", which, despite disagreements and ambiguities about the meaning of sustainable development, has legitimately emphasized the need to adopt a broad concept of capital, by including not only man-made physical assets, but also "human capital", "social capital" and "***natural capital***" (Pearce and Atkinson, 1998). The latter usually refers to environmental attributes entering individuals' utility functions, firms' production functions, or both.

Although describing natural resources as assets whose economic value stems from their service flows might appear to non-economists as a too narrow and questionable approach, we believe it allows one to address damage assessment consistently. In fact, adherence to this approach implies that, similarly to a disaster's impacts upon man-made capital, the impacts upon natural capital should be, in principle, evaluated by looking at the economic costs stemming from reduced environmental services (or, alternatively, by looking at the natural capital's restoration cost).

However, valuing natural assets and, consequently, assessing the economic costs stemming from reduced environmental services, is not easy, particularly when dealing with assets whose societal value does not merely stem from their role in the production of marketable outputs. Although the economic literature supplies a sophisticated technical armamentarium for estimating the value of unmarketable environmental services, applying valuation techniques is neither simple nor without cost. This is especially true in developing countries and, more generally speaking, in countries which have little tradition in incorporating environmental impact assessment into collective decision-making processes.

The intrinsic difficulties surrounding environmental valuation are exacerbated when dealing with natural disasters' damage assessment, since, in this case, analysts are more unlikely to be able to, or allowed to afford the luxury of implementing a full-fledged original study for estimating (lost) environmental values.

Exploitation of previous valuation studies may often constitute the only viable option, but special attention should be paid when relying upon the so-called ***value transfer method***. In particular, special attention should be paid to the original studies' relevance: i.e. the original study context and the transfer context should match as closely as possible. This reinforces the argument in favour of developing and expanding the scope of *ex ante* economic analysis of natural disasters' mitigation measures, which, as a by-product, could supply value estimates which could be subsequently exploited in *post*-disaster damage assessment.

We have also highlighted three specific issues which, in our opinion, deserve attention when trying to incorporate environmental damages into disaster impact assessment.

Firstly, attention should be paid to avoiding ***double counting problems***. Whilst failure to account for disaster-related environmental changes may involve a substantial underestimation of natural disasters' total economic costs, considering environmental damages *per se* could imply an overestimation of a disaster's welfare impacts.

In this respect, we have proposed a tentative taxonomy of "environmental damages", by making a distinction between damages occurring because of environmental (quantity or quality) changes altering the natural assets' intrinsic "productivity" (direct environmental damages); and welfare losses stemming from changes in people's "ability to use" environmental services (indirect environmental damages).

Since the latter can be often traced back to man-made capital's total or partial disruption, if the overall economic impacts of man-made capital disruption are already (explicitly or implicitly) included in damage assessment, treating welfare losses stemming from changes in people's ability to exploit environmental attributes as a separate damage category would imply an overestimation of the disaster's economic costs. Obviously, if some significant indirect environmental damages –e.g. increased health risks or reduced non-commercial recreational activities– are unaccounted for when assessing the economic impacts of man-made capital's disruption (indirect damage in ECLAC's jargon), they should be in principle identified and evaluated in order to obtain a comprehensive estimate of the disaster's total economic costs.

Similar considerations apply when, as advocated in the *Manual,* man-made capital restoration cost (*direct damage*) is used as a proxy measure of (future) welfare losses. If man-made capital's restoration allows the full recovery of people's previous ability to exploit environmental attributes, then indirect environmental damages should not be treated as a separate component which is then added to the disaster's economic costs. Obviously, if capital restoration does not allow immediate recovery of indirect environmental damages, the welfare losses, occurring during the *interim* period, should be added to the man-made capital's restoration cost.

Secondly, attention should be paid to the natural disaster's geographic and economic domain (***the disaster's relevant "market size"***). If the aim of damage assessment is to assess a disaster's impacts upon a specific community, say a country or region, analysts face the following problem: which environmental values, besides those related to "private" environmental services, or services holding "local public features", should be accounted for?.

The answer cannot be univocal, in that it is partly related to the actual assignment of "property rights" to environmental values. In this respect, we have argued that, even if damage assessment is only aimed at evaluating the economic impacts affecting the country directly experiencing the effects of a natural hazard, if the hazard undermines the country's ability to take advantage of the international willingness-to-pay for external environmental benefits, this financial loss should, in principle, be accounted for in the disaster's damage estimate.

Finally, attention should be paid to ***the timing of environmental damages and the related issue of discounting***. If a natural resource is interpreted as an asset, whose dividends are the benefits experienced by individuals over time, environmental damages calculated for a period of one year do not measure total damages. On the contrary, the essence of damage assessment is to determine the change in the value of the asset, and this requires, *inter alia*, selecting a discount rate, in order to get a consistent sum of lost dividends.

Choosing appropriate discount factors is not easy, especially when dealing with long-term environmental damages, or irreversible losses of unique natural assets' services. However, the difficulty of selecting a discount rate may be partly circumvented if, as advocated in the *Manual* with reference to man-made capital, the natural capital's restoration cost is used as a proxy of economic damages.

If restoration is feasible, essentially complete, and actually carried out, and if the rehabilitation phase does not last "too long", most of the problems surrounding the choice of an appropriate discount rate are, *de facto*, ruled out. In this case, a "standard" (social) discount rate should be used for computing the present value of welfare losses during the restoration phase.

On the contrary, if natural capital restoration is either unfeasible, or the rehabilitation phase lasts many years, adherence to a "precautionary principle" would dictate adjusting the discount rate, by lowering it in order to encompass uncertainty about future welfare losses stemming from irreversible, or almost irreversible environmental damages.

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