ENVIRONMENTAL VALUES, VALUATION METHODS, AND NATURAL DISASTER DAMAGE ASSESSMENT

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1. Introduction and the scope of the paper

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 changes may also be induced by natural hazards which, besides altering the intrinsic natural capital’s ‘productivity’, may negatively affect people’s ability to exploit/enjoy environmental goods or services.

In 1999, the UN Economic Commission for Latin America and the Caribbean has produced a Manual for Estimating the Socio-Economic Effects of Natural Disasters (ECLAC, 1999) 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 post-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 value measures into post-disaster damage estimates.

The paper is organised as follows. The following Section 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 3, 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 4, which concludes the paper, we focus on the incorporation of environmental values into natural disaster damage estimates, 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 into damage assessment.

2. Environmental values and valuation approaches

2.1. 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 synthesised 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 profitability of restoration measures.

Despite the advocated operational nature of monetary valuation, many writers have questioned or strongly criticised 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.

2.2. 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, mostly for practical reasons, 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.

2.3. Classifying environmental values

• 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 **use values**, 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.

**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 defence 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).

¹ As argued by Freeman (1993), one valuation problem with so-called vicarious uses is that “the observable market transaction (for example, the purchase of a nature magazine) often entails the simultaneous or joint 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” (p.268-269).
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.

**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 realised

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\(^2\) 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).
surpluses. This option value can be either 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 labelled 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 economic goods - in order to get utility from a natural asset.

The main advantage of this classification criterion rule 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

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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. The Author’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).
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’ demand for 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 economic activities involving the use of other economic goods.

• **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 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* 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 labelled 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.  

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 (e.g. visual amenity benefits) or indirect (e.g. flood control) use values may display both non-rivalry and non-excludability, and may be labelled 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).
2.4. Valuation approaches

As it will be illustrated in the following section, various techniques have been developed and applied 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. 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.

Broadly speaking, 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.

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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 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 cost of harvesting.
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 services, underlying non-use values, namely existence and bequest values, holding pure public features. As stated before, these values can only be assessed through expressed preference methods.

On the other extreme of this spectrum we may place quasi-private/private marketed marketable environmental goods and services, 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 services, 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; (ii) the economic nature of these related goods.

When other economic goods are involved, and they hold private features (marketed or marketable goods), the surrogate valuation approach is potentially capable of measuring unmarketable resources’ services. On the contrary, when there are no relevant private goods involved, analysts must inevitably turn to expressed preference methods.
3. Valuation techniques: An overview

3.1. Direct and indirect techniques

The literature provides various taxonomies of the valuation techniques developed to measure the economic value of unmarketable environmental goods and services. 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 former 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 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.

3.2. 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

9 The concept of household-production function is briefly illustrated in section 3.4
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 $\text{ENV}$ the environmental variable(s) of interest, and with $X_i$ ($i=1\ldots N$) other inputs, the production function of a representative firm might look like:

$$Y = f(X_i, \text{ENV}) \quad (1)$$

If $\delta Y/\delta \text{ENV}$ is positive, then a change in $\text{ENV}$ (e.g. an increase or decrease in air 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 $\text{ENV}$. Alternatively, this value can be estimated by looking at the changes in marketed inputs ($X_i$) required to maintain a given level of output.\(^\text{10}\)

\(^{10}\) 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
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.

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11 As underlined by Hanley and Spash (1993), since optimisation 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 modelling of production activities, improper constraints or just the fact that the real world operates suboptimally” (p.106).
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 ‘true’ economic value. Besides market failures, prices may be distorted by fiscal policies (taxation or subsidisation).

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\(^\text{12}\).

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).

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.

\[3.3. \text{Cost-based methods}\]

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, ex ante, the social rate of return of projects which were

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\(^{12}\) See section 2.3.
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.

- **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).

Finally, 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.

- **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 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.\textsuperscript{13}

The so-called \textit{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 “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).\textsuperscript{14}

\begin{itemize}
  \item \textbf{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.

  This 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 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).

  This assumption may not be valid in all cases. Ass 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

\end{itemize}

\textsuperscript{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.

\textsuperscript{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.
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.

3.4. 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’, 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, 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.

### 3.5. The hedonic pricing method

- **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 a good is based on its attributes. As long as the latter include environmental attributes, by modelling 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)\(^{15}\).

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 surrounding environmental conditions or by the perceived risk of natural hazards (*wage-risk analysis*).\(^{16}\)

- **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.\(^{17}\)

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)\(^{18}\), 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

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\(^{15}\) 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.

\(^{16}\) 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.

\(^{17}\) 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.

\(^{18}\) 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).
included characteristics, may lead to significantly biased estimates for these characteristics’ implicit prices (i.e., *inter alia*, biased environmental prices).

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 future benefits – or costs - associated to actual (not expected) changes. 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 capitalised 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<sub>i</sub>; i=1,…,N.), the next step consists of estimating an ‘hedonic price equation’, holding the following general form:

\[
P_H = f(ENV, C_i) \quad (2)
\]

---

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

20 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.
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).21

Pioneering HPM studies have mostly adopted linear functional forms, which imply that the implicit prices of the property’s attributes are constant. 22 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.23

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:

\[ P_E = \delta P_H / \delta ENV = g(ENV,C_i) (3) \]

If all individuals were identical in every respect, e.g. ll house buyers hold the same preference for a specific environmental attribute, (3) would give the (inverse) demand

---

21 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 parameterisation of the hedonic price model whereas the latter demands close attention to the robustness of the model and its extrapolative plausibility” (p. 111).

22 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 \( \alpha \) “will tell us how much the property prices vary if we alter the value of the environmental variable [ENV]” (Pearce and Turner, 1990).

23 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).
function for ENV. Otherwise, if we are to obtain an estimate of individuals' willingness to pay for given levels 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:

$$P_{Ej} = h(ENV, S_j)$$ \hspace{1cm} (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).

- **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 summarised 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’ $^{24}$, 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).$^{25}$

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).

$^{24}$ 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).

$^{25}$ 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).
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 maximising 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, biasing value estimates. Similar problems may be encountered when conducting wage-differential and wage-risk analyses.26

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). 27 28

26 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).

27 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.

28 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)).
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. 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)

3.6. 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 attribute(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.

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.

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

\[29\] 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.

\[30\] 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).
endangered species. CVM has also been applied, although much less frequently, for environmental damage assessments. 31

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 2.3).

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 the values derived from environmental attributes holding quasi-public/public features, which are not revealed by observable market behaviour (passive use values). 32

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’, 33 much of the technical debate over CVM has focussed on the survey design, and on the economic criteria which the results of a CVM application should meet. 34

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31 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).
32 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).
33 For a summary of this ‘philosophical’ debate, see Carson et al. (2000, pp.2-10).
34 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
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 2.3), 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 free-riding. 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.

A major focus of the technical debate has been on the choice of the question formats used: binary discrete choice question versus an open-ended question. “The argument made by some is that if agents had well defined preferences for the good, both formats should result in similar results [...] The counter argument, which comes from the economic theory on mechanism design, is that incentives for truthful preference revelation are different for these two formats and, as a consequence, one should expect the estimates should be different” (Carson, 1999, p.9).

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’ problems35- 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

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35 ‘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).
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 lines will be crucial in effectively incorporating the public’ spreferences into the environmental decision making arena” (Carson et al., 2000, p.37).

3.7. Environmental value transfer

• Rationale and potential advantages

The term environmental value transfer (EVT) – otherwise known as benefits transfer\(^{37}\) – 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).

\(^{36}\) Italicics added by the author.

\(^{37}\) 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 economic benefits or 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.
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.\(^{38}\)\(^{39}\)

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 \textit{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 \textit{per se}, but simply as the transposition of estimates from one context to another context (World Bank, 1998), as argued by Desvousges \textit{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).

\(^{38}\) 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.

\(^{39}\) Value transfer is used in the United States under the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) 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 Fig. 1) which illustrates different possible purposes of the transfer and the level of accuracy required for each.

**FIG. 1 – A STYLIZED CONTINUUM OF THE LEVEL OF ACCURACY REQUIRED IN TRANSFER STUDIES**

<table>
<thead>
<tr>
<th>LOW</th>
<th>HIGH</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fact finding</td>
<td>Compensable damages</td>
</tr>
<tr>
<td>Screening or scoping</td>
<td>Cost-benefit tests</td>
</tr>
</tbody>
</table>

Source: Adapted from Desvousges et al. (1998)
If the purpose of a transfer study is simply fact finding – such as 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).

**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 Fig. 2 and Fig.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 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).

- **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.

A. 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 through 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 been 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](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.
Fig. 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)

Fig. 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
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.40

B. 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.

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)41. 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).

40 “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).

41 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.
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, 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).

C. 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); 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 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).

42 For a literature review, see Garrod and Willis (1999: 347-351); Desvousges et al. (1998: 28-36)
43 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 2.2.), 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’ (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).
4. Natural disasters and environmental values

4.1. 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, 1999).

Besides destroying or harmfully affecting human, man-made and social ‘capital’, these occurrences may also seriously influence natural capital (‘s productivity). Losses of environmental values may either occur because of negative environmental (quantity or quality) changes, 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, 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 unfavourable 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

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44 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).
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.

### 4.2. The ECLAC’s Manual: classification of the effects of a natural disaster and criteria for evaluating damage

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

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, 1999, 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, 1999, 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, 1999, p.14). These include, inter alia, environmental changes.

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, 1999, 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 summarised 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 to flows of goods or services will be evaluated at producer or market prices, as appropriate” (ECLAC, 1999, p.21).  

45 Italics added by the author.
4.3. Natural disasters and environmental values: a tentative taxonomy

As underlined in Section 2, 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.

The former stems from alterations of the (net) benefits derived by exploiting environmental attributes, usually in conjunction with other economic goods. On the contrary, non-use values, arising from the psychological benefits people derive from the mere existence of a resource (and/or from intergenerational equity concerns), are not reflected in any ‘market behaviour’, i.e. these values are not generated through carrying out an activity involving other economic goods.

Broadly speaking, natural hazards may affect use values in two different ways: (i) by inducing temporary or permanent environmental changes thus altering a natural asset’s ‘intrinsic productivity’; (ii) by altering people’s ‘ability to use the environment’ (the economic costs people have to afford to exploit available environmental goods and services).

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

Examples of direct environmental impacts 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.

Indirect impacts arise from man-made capital’s partial or total disruption which may impede, or make it more costly, to exploit environmental services entering firms’ production functions, or ‘households’ production functions’, or both.

For example, the disruption of water-distribution networks or water-treatment facilities, caused by an earthquake, would harmfully affect water resources’ use-values (loss of agricultural or industrial production; increased health risks; increased public/private averting expenditures). Or damages to 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 are not derived from carrying out activities involving the use of other economic goods, they can be affected by a natural disaster only if it entails environmental changes.

In short, as summarised in Fig.4, a natural disaster may affect environmental values in two ways. Directly, by inducing environmental (quantity or quality) changes affecting use values and/or non-use values. Or, indirectly, by affecting people’s ‘ability to use’ a natural asset(s services).

4.4. Intersections between environmental values and ECLAC’s damage categories

As underlined in Section 4.2, the ECLAC’s Manual provides a conceptual and operational distinction between natural disasters’ direct and indirect damages. The former include damages to man-made capital. The latter encompass welfare impacts
related to changes in the supply (or in the supply cost) of marketable goods and services.

On the other hand, in the previous Section, we have provided a tentative taxonomy of natural disasters’ impacts upon environmental values, by making a distinction between values affected by environmental changes (direct environmental impacts), and values affected by changes in the ‘ability to exploit’ environmental attributes (indirect impacts).

Although drawing a clear-cut borderline between direct and indirect ‘environmental’ impacts may prove sometimes 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’ impacts upon environmental values, and other damage categories.
[INSERT HERE FIGURE 4]
Our notion of direct environmental impact 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.

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 benefits attributable to these changes over time.

Alternatively, as a proxy of the ‘true’ welfare cost, the capital’s restoration cost can be used as a measure of damage, provided the analyst believes that the cost incurred in restoring the asset to its original state is not greater than the benefits the damaged/destroyed assets provide[d]. This is the valuation approach recommended in ECLAC’s Manual, which, however, in order to avoid the underestimation of damages, recommends to take also 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 may be problematic. This because (i) the restoration of a natural asset’s original ‘productivity’ may be technically unfeasible; (ii) many environmental goods and services are not exchanged in normal markets; (iii) 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].

Nevertheless, some of the welfare impacts deriving from environmental changes (direct environmental impact) may already be implicitly captured by ECLAC’s concept of indirect damage. In fact, according to the Manual, this damage category does not exclusively encompass temporary changes in the flows of marketable goods and services attributable to man-made capital’s disruption, but also changes attributable to other effects of a disaster.

As far as the latter are concerned, the Manual explicitly mentions disasters’ effects upon social and economic infrastructures as a potential additional source of indirect damage. An explicit reference to environmental changes would allow us to complete the picture, and would more clearly point out that ECLAC’s notion of ‘indirect damage’

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46 See Section 3.3 (Restoration cost approach).
47 In the ECLAC’s 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, 1999, p.13)
encompasses whatever (temporary or permanent) damage to the flows of goods and services, deriving from (man-made, social, and natural) capital’s alterations.

Obviously, not all environmental changes’ impacts upon people’s welfare are comprised into the ECLAC’s notion of indirect damage, since it only comprises changes in the flows of marketable goods and services.

It follows that if a natural disaster involves environmental changes which do not (exclusively) affect production activities (marketable outputs), their impacts upon people’s welfare will be, *de facto*, left uncovered by the ECLAC’s notion of indirect damage. This is the case for environmental changes affecting non-use values, and often the case for changes affecting environmental attributes entering ‘household-production functions’.

As far as the latter are concerned, some impacts upon use-values held by households might be however implicitly included in ECLAC’s indirect damage estimates. For example, health-related values may be implicitly accounted for, as long as these estimates include an evaluation of post-disaster additional costs of health campaigns required to avoid/reduce the harmful impacts of, say, increased water pollution; or recreational values may be implicitly captured by estimates of ‘production’ losses, experienced by the tourism industry.

However, if the indirect damage estimates do not account for all monetary costs imposed by a natural disaster, and/or if the environmental changes’ welfare impacts do not manifest themselves through additional monetary costs, a portion of the environmental direct impacts will be inevitably ignored in the damage assessment, unless a full-fledged environmental valuation exercise is carried out.

Turning now on our notion of indirect impacts, 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 exploit potentially available environmental services, some of these impacts could be masked under ECLAC’s damage categories, namely under direct damage.

In fact, these indirect impacts typically arise because of the partial or total disruption of other forms of capital, like physical infrastructures. As previously argued, the welfare costs of capital’s disruption should be in principle evaluated by looking at the losses in the capital’s service flows, until the asset’s restoration. If evaluated in this way, the welfare costs would also include the loss of benefits people will experience as a result of the impossibility to exploit environmental goods and services, or as a result of the increased exploitation costs.
However, the ECLAC’s Manual has made a choice in favour of capital’s restoration cost, as a proxy measure of damage effect. 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, the man-made capital’s restoration-cost measure (direct damage) should be also interpreted as a proxy of the indirect environmental impacts attributable to man-made capital’s disruption.

To summarise, 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 impacts (see Fig.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 impacts), or some of the effects resulting from changes in people’s ability to use potentially available environmental services (indirect impacts).

It follows that attention should be paid to avoiding double-counting problems, which could emerge by treating the environmental dimension of a natural disaster as a separate ‘value component’ of a disaster’s socio-economic impacts. 48

On the other hand, the emphasis placed by the Manual upon marketable goods and services, 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 impacts related to ‘productive’ uses of the environment, could be relatively high.

4.5. The choice of environmental valuation method

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.

48 In this respect, it is worth noting that the Manual’s inclusion of environmental changes among a natural disaster’s possible indirect effects is somehow misleading, in that, from a conceptual point of view, these changes tend to fall within the Manual’s category of direct effects (i.e. effects upon capital’s service flows).
The choice also depends on the environmental value categories involved (see Fig. 4). 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 3, 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 ‘specialised’ in the evaluation of particular use-value’s sub-categories (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 2.4, the value of unmarketable environmental services, underlying use values, may be estimated through surrogate market valuation methods only if these services enter a firm’s production function, or if households’ demand for these services can be inferred by looking at related marketed goods.

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 3.7, 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 already underlined, some ‘losses of environmental values’ may already be embedded in other damage categories. In particular, when a natural hazard affects the flows of environmental services (and/or the ability to use services) entering firms’ production functions, the resulting welfare impacts may already be captured by indirect damage estimates. In fact, if a natural disaster involves changes in the flows of marketable outputs, totally or partially attributable to natural capital’s disruption, ECLAC’s damage estimates, based upon the ‘market value’ of these changes, can be interpreted as an application of the production-function approach illustrated in Section 3.2.

Other elements of caution derive from the intrinsic, general potential drawbacks of the environmental value transfer approach.

In particular, as already stressed in Section 3.7, 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.
4.6. Damage for whom? The natural disaster’s relevant geographic and economic domain

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. 49

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 ECLAC’s 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 providing services underlying indirect-use values and/or non-use values is wider, and may go well beyond the geographical area experiencing direct and indirect damages.

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 ‘policy’ 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 the environmental changes’ welfare impacts fall to zero. These may prove very difficult when a natural asset provides services, holding public features, of national or international significance.

49 The term ‘victim’ and ‘affected person’ are used as synonymous in the ECLAC’s Manual. The Manual 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, 1999, p.36). “Those population segments which suffer the indirect effects of the disaster are secondary and tertiary victims. The difference between the tow 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).
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.

As underlined in Section 2, 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.

However, despite the technical non-excludability, some services’ beneficiaries might be willing to contribute to natural capital’s conservation, if they perceive that a complete ‘free-riding’ attitude could harmfully affect the 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 global 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 marketable environmental 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.
REFERENCES


