Improving Benefit Transfer for Wetland Valuation: Income Adjustment and Economic Values of Ecosystem Goods and Services[1,2]

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Abstract

To save time and resources, benefits transfer is widely used in economic valuation of ecosystems goods and services. However high uncertainties are involved in the value estimated between two countries with different levels of income and differences toward the environment. This paper surveys the method of benefit transfer, in particular in its application to wetlands´ ecosystem goods and services valuation and proposes a new methodology for income adjustments that produces more reliable estimates than those currently used.

Keywords: Benefits transfers, income elasticity of willingness to pay, wetlands´ ecosystem goods and services, environmental Kuznets curve, economic valuation methods

JEL: H4, Q5, O4

1.Introduction

Earth's ecosystems produce goods and services which satisfy different necessities of people and in this way they determine in a crucial manner human welfare. Moreover, in the absence of ecosystems, their indispensable ecosystem services and functions would cease to exist and, with them, the provision of goods and services they generate would also disappear, which would render life on Earth impossible. There are, therefore, good enough reasons to care about the appropriate conservation of ecosystems and the future adequate provisions of the good and services they provide. The mounting deterioration of the planet ecosystems currently observed (MEA 2005; EC 2008; CBD 2010) is mostly the result of an inadequate appraisal of ecosystems' contribution to human wellbeing, which provokes their undervaluation and lesser priority to their care and conservation than they deserve given their relevance for human current welfare and future survival. The undervaluation of ecosystems and of ecosystem goods and services is in large part provoked by their common-pool resource characteristic and the absence of formal markets in which their relatives scarcities are properly gauged and assessed.

In fact, in economic terms, ecosystems are valuable precisely because the goods and services they provide positively affect human welfare. Moreover, the decisions that individuals and society make in the scarcity context they ought to live 'reveal' their relative valuations of these goods and services relative to other goods and services, such as those produced by the economic system.

Economic science has developed different techniques to reveal and measure the value of goods that do not have explicit markets and that therefore, do not have explicit market prices either. These techniques use actual and/or constructed decisions that individuals and society make in the scarcity context they ought to live, to revealing their relative valuations of these goods and services vis à vis the other goods and services. The use of these estimating techniques allow to calculate quantitative expressions of the individual and social valuations of the different goods and services provided by the ecosystems, which provide extremely valuable information, not only on the relative appreciations by people of these ecosystem goods and services, but also on their relative scarcities and the relative willingness of people to care for their conservation.

The methodologies of economic valuation – such as travel costs, hedonic prices, stated preferences, contingent valuation, etc. – use information on related goods that do have markets or that is obtained from specially designed surveys applied directly to those from whom we are interested in revealing or determining their valuations. The technique to be used in each case depends on the type of ecosystem good or service we want to valuate and the type of contribution it makes to the wellbeing of individuals or society. Examples of applications of these methodologies have been growing in the last decades both in developed and developing countries (Bateman, 1999; Bateman and Willis, 1999).

However, using the methodologies mentioned above for policy decision making and implementation is generally time consuming and expensive. The growing demand for using cost-benefit analysis to assessing prospective policies to be implemented and the advantages of benefit transfer (BT) in terms of time and money saved when economically valuing ecosystem goods and/or services involved with these policies have triggered an increased interest in the BT technique for policy analysis.

Benefit transfer is the transference of the economic value of an ecosystem good or service estimated in an original study to a new study demanding a rapid and efficient assessment of the same ecosystem good or service in a different location.

Good reviews of traditional methodologies of benefits transfer exists and they can be found Navrud (2004) and Navrud and Ready (2007). The focus of this paper is on recent advances in BT methodologies, their applications to wetland ecosystem's goods and services and, specially, on a proposed novel methodology – which is both empirically consistent and well founded in economic behavior –to adjusting the willingness to pay (WTP) for an ecosystem service provided by a wetland estimated in an original study to transferring it to a new context (location)..

The paper is organized as follows. In Section 2 we describe the Total Economic Value framework for economic valuation and briefly summarize the current techniques for economic valuation. Section 3 describes the Benefit Transfer (BT) methodology. Section 4 discusses modern insights of benefit transfer methodologies including declared subjective happiness assessments, a new technique that is emerging with the most recent advances in the field of behavioral economics. Section 5 addresses the issue of economic valuation of wetlands and the ecosystems services they provide. Section 6 presents a new approach for income adjustment of unit values transferred. Section 7 summarizes and presents concluding commentaries.

2. The total economic value approach and how to estimate it

2.1 Total economic value approach

The total economic value (TEV) is a methodology to put a monetary metric on the benefits provided by natural resources. These benefits are not only those provided directly through the direct enjoyment of people such in the cases of consuming water, foods, fibers or medicinal plants, for example, but also can be less tangible (material) benefits such as those derived from recreation or aesthetic pleasure (scenic beauty). If there is a willingness to sacrifice some goods in order to enjoy goods and services provided by nature, a value can be attached to those goods and ecosystem services.

The TEV is the sum of all this direct and indirect components attached to a particular ecosystem service. Following Perman et al (1995) and Adamowicz (1995) the TEV is composed of two main components: use and non-use values. Use value corresponding to those values assigned by a person to the direct, indirect or optional use of an ecosystem good or service. Non-use value is a value attached to a good or service although they are not directly enjoyed.

Use value is disaggregated into the following: *direct use value* that corresponds to the value attached to direct use of goods and services provided by the resource. These values include consumptive use of the resource such as hunting, fishing, timber collection and so on. This consumptive use is relatively easy to estimate because in general it is associated with some private goods traded in the market (timber, fruits, fiber and so on). In addition, use-value includes non-consumptive uses such as hiking, camping, boating and nature photography (Fausold and Lilieholm 1996) which are more difficult to estimate due to the absence of well-defined markets. *Indirect use values* are the benefits indirectly accounted for the use of the resource. These include particularly those goods and services derived from some of the natural functions performed by the ecosystems, such as soil conservation, flood prevention, water purification and regulation, etc. The functions provided by forest ecosystems are included in this category, for example. *Option Values*: This concept relates to the possible use of the resource in the future (for example the value assigned to a resource with potential use in pharmacological product or potential use as energy supply). *Bequest values*: is also a category of use values, and correspond to the value assigned to the good and services in order to preserve them for future generations. It considers making guesses about the preferences of future generations and including them in the present generation preferences (Kula 1994).

Non-use values: only existence value is included in this category. *Existence value* is the value attached to a good or ecosystem service even if it will never be used or visited and therefore, it is a value assigned to its mere existence. Some goods or services provided by wetlands, such as biodiversity and habitat, are included in this category of value (Merlo and Briales 2000, Perman et al. 1995).

2.2 Economic valuation techniques

There are different methodologies to economically valuing goods and services provided by ecosystems in general and wetland's ecosystems in particular. All of them differ in its validity for the case at hand, their theoretical underpinning and their informational requirements and feasibility (Bishop, 1999). In the remaining of this section, a summary of the main characteristics of the different valuing methods is presented. For a more detailed account of these characteristic the reader is referred to Navrud (2004), Navrud and Ready (2007), and Brander et al (2006):

Market prices

It is the simplest methodology because it assigns a value to several goods and services provided by ecosystems based on market prices already existent. Wood, food, fiber and material extraction from ecosystems are example of goods whose economic valuation is based on this methodology (Constanza et al 1997, Raphael and Jaworski 1979)

Travel cost

This methodology is mostly applied when ecosystems provide recreational services. It consists in estimating travel expenditures incurred by those who visit the zone with recreational motives making possible to estimate the demand for recreational uses. In order to apply this methodology it is necessary to know of all complementary goods that contribute to satisfy the recreational motivations. In addition it is also necessary to take in consideration the cost of time invested traveling to the zone. An important advantage of this method is the use of market information because all components of the travel cost are taken from real markets (Ramdial 1975, Cooper and Loomis 1993).

Hedonic prices

Based on this methodology it is possible to valuing environmental attributes as part of the attributes of a heterogenous good (housing, professional activities, etc.). For example, a house located in a zone with favorable environmental conditions (clean air, natural surroundings, landscape, etc.) is more valued due to the presence of these characteristics than other houses located in places without them. This methodology uses econometric techniques to calculate the increase in the value in houses provoked by each particular characteristic (Lupi et al 1991, Doss and Taft 1996).

Production function

This method is also known as input-output, change in productivity method or doses-response function. Based on this methodology is possible to estimate indirect use values with regard to their contribution to market activities. It is based on a production function in which natural capital is an argument and therefore, this method is only useful to determine the value of an environmental service the ecosystem provides to an existent activity (.Acharya and Barbier 2000; Bell 1997, Nunez et al 2006, Figueroa and Pasten 2008).

Contingent valuation

This method is based on responses of people to direct questions about their willingness to pay for an environmental improvement or their willingness to accept an environmental worsening. It has an advantage over the other valuation techniques since it allows for the estimation of non use value. Its theoretical bases are welfare theory and the assumption of consumer rational behavior (Farber 1988; Bateman and Langford 1997).

Valuation method based on costs

These methods are based on the estimation of the cost to maintain, provide or restore a good or ecosystem service provided by ecosystems. Examples of this type of method are replacement costs, precautionary spending, and opportunity costs. The replacement cost is a technique that measures the benefits provided by the service due to the estimation of reproduction costs to the original levels of provision if those are lost (Breaux et al. 1995; Emerton and Kekulandala 2002). Defensive costs are estimates of spending on prevention or defensive measures to avoid the degradation or loss of values of environmental services. Opportunity cost is a methodology to estimate the value of the alternative use of the ecosystem (Leitch and Hovde 1996; Sathirathai and Barbier 2001). These techniques must be used with caution because do not represent the true willingness to pay for a good or environmental ecosystem service (Bishop, 1999)

3. The benefit transfer methodology

Benefit transfer (BT) is the valuing methodology focused by this paper and it is employed to estimate ecosystem economic values by transferring available information from a site where a study was realized to a place where the valuation has to be performed under the assumption that characteristics in both sites are similar. For example, the scenic value in a lake in a particular region can be estimated using information provided by a study already existent of the scenic value of a lake with similar characteristics in a different place.

The main challenge for the use of benefit transfer is, therefore, to determine whether the prevalent conditions in the place, moment or context in which the original value was determined are similar or not to the existent conditions in the place, moment or context characterizing the good or ecosystem service being valued, such that it is justifiable to assume the value obtained in the first case can be applied in (transferred to) the valuation currently performed.

Benefit transfer usually is employed when is too expensive or time consuming to produce primary economic valuation studies. Is for this reason that the method has triggered an increasing interest and the literature has expanded rapidly in the last years. This method is more reliable when the original site and the site object of the transfer are similar in quality, location, and population characteristics, when the environmental changes are similar, and when the original study was soundly realized and used appropriate economic valuation methodologies (King et al 2007) In this sense, it would have to be taken into consideration that the maximum exactitude and reliability attributable to the benefit transfer method are those of the original study.

3.1 Who use it?

BT is used by people who have to inform decision making in a not expensive and time consuming setting. For example, the U.S. Oil Spill Act recommends transfer of values to assess the damages resulting from small spills or accidents by the mean of transferring value estimates from several sources (see Navrud 2004, Brander 2006).

For policy valuation, benefits transfer was first used by Costanza et al (1997) to estimate the economic value of the world ecosystem´s services. BT is also the predominant methodology proposed ExternE (European Commission, 1995, 1999, 2003) which provides a framework to estimate the economic value of externalities associated to projects, programs and policies to be implemented within the context of the European Union. Krupnick et al (1995) use BT to value the health benefits from air quality improvement in Central and Eastern Europe. Brenner et al (2009) used the methodology to estimate the non-market value of ecosystem´s services provided by the Catalan coastal zone in Spain. Anieski and Wilson (2005) estimate the value of Canada´s

Boreal Ecosystem and Niemi and Lee (2002) estimate the economic benefits to protect the natural resources in the Sonora Desert. Shahwahid and McNally (2001) used benefits transfers to estimate the terrestrial and marine resources of Samoa.

In forest, BT has been used to estimate the Total Economic Value (TEV) of forests in Mexico (Adger et al. 1995) and, by Pearce and Pearce (2001) to estimate the TEV of forest ecosystem services at global scale. Deluchi (2002) compares benefit transfer to hedonic price method to elicit the health and visibility cost of air pollution. In developing countries the BT methodology has been used in Mexico to estimate the impact of climate change (Ibarraran and Rodriguez 2007), health impacts from power plant emissions (Lopez et al. 2005), damage costs of air pollution (MacKinley 2005) environmental impact of electrical power plants (Macias and Islas 2010), and for the economic valuation of the ecosystem services provided by the National System of Protected Areas in Chile (Figueroa 2009 and 2010).

Benefit transfer is often used in estimating the economic value of a statistical life. In this kind of models estimates derived from one setting, generally job-related accidents are transferred to a somewhat different scenario such as risk for atmospheric pollution (see Hammit et al 2011),

3.2 Types of BT

The benefit transfer methodology consists in the transferring of the economic valuation of a given ecosystem or environmental good or service that have been estimated in given site, moment and/or context (the study site) to the same ecosystem or environmental good or service in a different site, moment or context (the policy site) where its valuation is needed. To make this transference three main approaches exist: transferring the unit value, transferring the benefit function and employing meta-analysis.

Unit value transfer (UVT)

This type of BT consists in just transferring the adjusted or un-adjusted unit value reported by the study site to the policy site. Obviously, it ought to be the case that both sites (i.e., locations, moments and contexts) are similar. As the main advantage of this method is its simplicity, it has become one of the most used for policy analysis.

Perhaps one of the best know examples of ecosystem valuation using unit value benefit transfers is the work of Constanza et al (1997) which valuates the world's ecosystems using point values taken from different studies made in different parts of the world to assigning a unitary value to all the ecosystems of the planet. Another well-known example is Extern-E, (European Commission, 1995, 1999, 2003) a methodology developed by the European Community to estimate the economic value of the environmental impacts provoked by project and polices in the energy sector. The valuation methodology rests mostly in the use of simple unit value transfers. Other studies that estimate externalities of projects and policies are for example, the U.S. Environmental Protection Agency (EPA) guidance for estimating the value of a statistical life (VSL) which relies in the unit values reported by Viscusi (1992, 1993); the Department of Transportation Guide for calculating the VSL, which draws value from one study by Viscusi (2004) and four meta-analyses (Miller 2000, Mrozek and Taylor 2002, Viscusi and Aldy 2003, and Kochi et al 2006). The US Department of Homeland Security relies in the Viscusi (2004) paper. Some studies of VSL in low income countries have used transferred values adjusted by income; for example Lvovsky et al. (2000) use EPA-VSL estimates for their analysis of six Asian cities. In addition, the World Bank (2006) makes reference to European studies applied to Pakistan. With regards to its simplicity, there exist several studies showing that there is not a clear advantage using more sophisticated methods of benefit transfers instead of the simpler UVT (Barton 2002).

If the transfer is taken from a different country than the country of application, an estimation of the difference in purchasing power have to be addressed according to differences in relative prices between both countries. Also if it is within the same country but in different moments of time, an adjustment for inflation has to be performed. However, even if there is no need to adjust for purchasing power or inflation, still differences in income will provoke differences in willingness to pay that in turn imply differences between the WTP in the study site and the policy site. Which is the proper way to make these adjustments for income is one of the main concerns in this paper (see section six below).

Benefit function transfer (BFT)

The simple unit transfer value does not consider specific information from the policy site; therefore, an alternative available methodology is the so called *Benefit Function Transfer* (BFT) which, instead of transferring the point value estimates from the original study site, transfers the whole benefit function estimated in the study site. Then the average characteristics of the policy site are plugged into the benefit function to obtaining the new values to be transferred. In this fashion information from the policy site is incorporated in the calculation of the values transferred which are estimated using as much data as possible from the policy site. In this case the information requirements are more demanding since it is necessary to have available specific information from the policy site on all the covariates included in the original regression (age, sex, income, etc.) and about the characteristics of the policy site itself (location, distance from the coast, extension and so on). The implicit hypothesis when transferring the benefit function is that the underlying preferences are similar between the study and the policy site. Early authors using BFT proposed to transfer the entire demand equation for recreation based on the travel cost method rather than just the willingness to pay value estimated from those demand functions (Burt and Brewer 1971; Brown and Hansen 1974; Ciccheti et al. 1976; Dwyer et al. 1977). Ciccheti et al. (1976) for example, proposed to estimate the benefits of a new sky resort at Mineral King in the Sequoia National Park, California, based on existing demand functions for other ski resorts. The authors replaced the values of the independent variables in the original demand functions estimated by the values of the site object of the study. Loomil (1992) tests the null hypothesis of equal coefficient in the demand for ocean salmon sport fishing in Oregon and Washington and for freshwater steelhead fishing in Oregon versus Idaho rejecting the null hypothesis of equal coefficients. Hellerstein and Feather (1997) develop national estimates of the non-market water-based recreational benefits of reductions in soil erosion through the use of BTF.

Meta-analysis

Meta-analysis summarizes information from several valuation studies averaging their values expecting that this procedure will provide more accuracy than simple unit value transfer. Its main objective is first to test hypotheses with respect to the effects of the explanatory variables on the value of interest; and, in second place, to use the estimated a meta-analysis model to predict values across time and space (Bergstrom and Taylor 2006). A model of meta-analysis is developed by Bergstrom and Taylor (2006), and Boyle et al (1994) employ one for valuing the benefits of underground water. Moreover, meta-analyses to elicit the value of statistical life were developed by Miller (2000), Mrozek and Taylor (2002) and Viscusi and Aldy (2003). Other efforts include the Loomis and White (1996) valuation study of endangered species, and the Brouwer et al. (1999), Woodward and Wui (2001) and Brander et al (2006) meta-analyses of wetland valuation studies.

4. Some recent advances in Benefit Transfer

4.1 Parameter calibration approach

The parameter calibration approach recognizes that individual's willingness to pay is ultimately defined by preferences and, as a result of this; the methodology specifies utility functions and therefore allows for estimates that are consistent with economic theory. It determines the relevant parameters of a utility function from a set of studies and consequently it makes it possible to obtaining empirical estimates of the WTP for an environmental improvement derived from the underlying theoretical utility function. The researcher using this methodology must be prepared to make assumptions about the specific form of the utility function. According to Smith et al (2002), in first place, the method is theoretically consistent with the economic theory of preferences; secondly, it offers observable predictions that can be contrasted with the empirical evidence; third, when different estimates for the same case exists due to methodological differences, the parameter calibration approach uses the definition of each method in a unified framework to reconcile differences; and fourth, it provides a framework to take into account size effects. The methodology has been applied by Smith et al. (2000) to evaluate the impact on recreational activities due to improved water quality; and by Smith et al. (2002) to evaluate the effect of water quality in lakefront property values in Maine.

4.2 Life satisfaction approach

Closely related to the technique of parameter calibration described above is the use of happiness surveys as a novel method of economic valuation. Under this methodology reported values of happiness - or a function of it- resemble the utility functions generating the WTP for environmental quality. The marginal impact of both income and environmental quality on happiness can be estimated and the marginal rate of substitution between income and environmental quality is an estimation of the marginal willingness to pay for an environmental improvement. The methodology have been used to obtaining economic valuation of flood disasters (Luechinger and Raschky, 2009), climatic conditions (Frijters and van Praag 1998; Rehdanz and Maddison 2005; Becchetti et al. 2007; Brereton et al. 2008), airport noise nuisance (van Praag, and Baarsma 2005), proximity to infrastructure (Brereton et al. 2008), urban regeneration schemes (Dolan and Metcalfe 2008), droughts (Carroll et al. 2009), crime (Cohen 2008), terrorism (Frey et al. 2009) and air quality (Welsch 2002, 2006, Di Tella and

MacCulloch 2007, Levinson 2009, Luechinger 2009a, Luechinger 2009b, MacKerron and Mourato 2009).

4.3 Bayesian benefits transfers (BBT)

This methodology is a middle point between unit value transfer and doing a whole survey in the policy site. It consists in carrying out a small and not expensive survey in the policy site and using Bayesian methods to update the prior information coming from the study site with information from the survey. This approach is justifiable as long as preferences and distribution of socioeconomic characteristic of the population are different between the study and the policy site. Simple value transfer assumes identical preferences and distribution of socioeconomic characteristics between the study and the policy site. Benefit function transfer is less restrictive since it allows for a different distribution in the site of implementation but it still assumes identical preferences. In contrast, Bayesian benefit transfer allows for both different preferences and a different distribution of the characteristics in both sites (Lehr 2005).

The first work applying BBT is Parson and Kealy (1994); they use a small sample of Milwaukee county residents in the State of Wisconsin in order to valuate an improvement in the quality of water for recreational uses by Milwaukee residents. The authors pull the small sample together with a broader sample of the State of Wisconsin to estimate benefits based in a random utility model. Lehr (2005) analyzes willingness to pay for the creation of an artificial recreational lake using an small sample in the policy site whose WTP estimated is updated with information from a similar project developed in Kovenhavn, Denmark. In Leon et al. (2002) past information on the benefits of national parks in Spain are combined with on-site sample information to obtain more accurate results (see also Leon et al 2003). Kochi et al (2006) applies the methodology to the estimation of the value of a statistical life for environmental policy analysis. In a recent paper Dekker et al (2011) use Bayesian meta-analysis for empirically estimate factors of correction for out of context benefit transfer of value of statistical life (VSL) i.e. a value estimated in the context of road safety to be applied in the context of atmospheric pollution for example. Other applications of BBT are in Leon and Vazquez-Polo (1998), Moeltner et al (2007) and Leon-Gonzalez and Scarpa (2008)

4.4 Unit Transfer with income adjustments

Unit value is the less expensive and time-saving form of benefit transfer and there is not strong evidence of being outperformed by other methods such as benefit function transfer, meta-analysis, Bayesian meta-analysis, etc. Even though income is the main mechanism of adjustment, there are other forms of adjustment that can be applied to the values being transferred. For example, in the cases of VSL, adjustments are made for external costs (Robinson and Hammit 2010) and for age in a study for Canada (Jenkins et al. 2007). In an analysis conducted by the World Bank (Lvovsky et al. 2000), income is clearly the main variable to adjusting the unit values being transferred. Due to the fact that differences in income between the study and the policy sites is in general the main concern for adjusting transferred values, section 6 below is devoted to analyzing the current practice of income adjustment and proposing a novel methodology firmly grounded in microeconomic fundamentals and economic behavior.

5. Ecosystem services valuation

As it was mentioned before, economic science has methodologies available today for calculating or revealing the valuation of ecosystems and ecosystem goods and services, a great deal of which is not traded in the market and, therefore, there exist no explicit market prices for them. These methodologies use information on related goods that do have markets or that is obtained from specially designed surveys applied directly to those from whom we are interested in revealing or determining their valuations. The technique to be used in each case depends on the type of ecosystem good or service we want to valuate and the type of contribution it makes to the wellbeing of individuals or society. Therefore, it is important to appropriately characterize the ecosystem good or service we want to valuate in order to choose the adequate technique. One key characteristic of the good or service to determine is the way it affects the welfare of individuals or society. However, to define precisely the ultimate welfare determinants of the individual and/or social welfare affected by a given ecosystem good or service is not trivial. Moreover, the lack of clarity or the ambiguities still remaining in the definition of the concepts of good and service, as well as of the roles they have in determining the individual and collective welfare, seem to be at the center of the difficulties that natural and social sciences have had to communicate between them. For social sciences in general, and for economics in particular, these are core concepts with generally quite precise definitions from which an important part of their conceptual architectures are built.

In the last decades, natural scientists have made efforts to introduce these concepts in their analyses. Moreover, there have also been attempts from economics, ecological economics and natural sciences to bringing together languages and visions in order to produce a common interdisciplinary approach (Folmer and Johansson-Stenman forthcoming). The Millennium Ecosystem Assessment (MEA 2005) is the most important recent attempt in this line and which has had and will have a significant effect.[5] The MEA relates the ecological functions of ecosystem, the ecosystem processes, the ecosystem services and the ecosystem production of goods and services that have explicit markets, and proposes for their assessment an analytical model with two prominent features. The first is the emphasis it places in what it calls ‘ecosystem services’. The second is the change it introduces to the usual economic meaning of the ‘ecosystem goods and services’ concepts. Regarding the first of these two aspects, the MEA in fact gives great relevance to the usually called ‘environmental services’, ‘ecosystem functions’ or ‘ecosystem services’. In addition, it includes them in three categories: regulating services; supporting services; and, cultural services. This represents a contribution in the sense that it calls attention on the importance that these ecosystem services have, since among them there are some so crucial to human life and wellbeing as the mechanisms that regulate the impacts of stress or sudden shocks –such as disease regulation– and other services related to air

[5] Charles Perrings, for example, believes that “MEA has changed the way we think about the interaction between social systems and ecosystems” (Perrings 2006).

quality regulation, the regulation of hydrologic cycles, of floods, of aquifer recharge, of soil erosion, etc.

With respect to the second aspect, however, by including all goods and services that are produced by the planetary ecosystems in a single category that it calls “services” or “ecosystem services”, the MEA discards the usual differentiation between goods and services defined and employed by economic science. This is a mistake, because, on the one hand, it creates a source for imprecision; and on the other hand, it restricts and diminishes the conceptual richness of the economic nomenclature that employs both terms – goods and services– instead of the last one only. In fact, economic science distinguishes between goods and services to differentiate, among the elements that determine the welfare of individuals or society, between those that are tangible (goods) and those that are intangible (services). Goods, such as bread, fruits and cars provide welfare to people by meeting a particular necessity, such as satisfying hunger or providing mobilization. Services also satisfy personal necessities, such as a haircut or a concert, and for that reason, they also generate welfare to persons and society. Sometimes the term ‘service’ is used to refer to the entire process or activity that generates or produces the ‘element’ that finally affects welfare. Analytically it is important, however, to distinguish that welfare is ultimately affected and determined by that element and not by the entire process or activity that generated such an element. Moreover, when dealing with nature, ecosystems and the goods and services they provide to individuals and society, there are many significant aspects related to their relation with people welfare for which it is analytically useful and meaningful to keep the distinction between good and services.

To classify the goods and services that ecosystems provide to people and society MEA (2005) adopted four categories to classify ‘ecosystem services’[6]: ***1. Provisioning (goods and) services:*** tangible goods (foods, water, fuels, fibers, raw materials, genetic resources, etc.) that are obtained from ecosystems, a large proportion of which has structured markets where they are traded; ***2. Regulating services:*** services (water purification, and regulation of floods, drought, land degradation, and disease, etc.) related to ecosystem processes and their contribution to regulating the natural system; ***3. Cultural services:*** services that humans obtained from ecosystems through spiritual enrichment, cognitive development, inner reflection, recreation and aesthetic enjoyment. They are closely linked to human values, identity and behavior; and, ***4. Supporting (or based) services:*** services (climate regulation and hydrological regulation, etc,) necessary for ecosystem functioning and the adequate production of provisioning goods and services and regulating services. Their effects on welfare show in the long run through impact on the provision of other ecosystems goods and services.

5.1 Overview of wetland literature and one example

Given the importance of wetland ecosystems and due to their characteristic of being public goods, without explicit market values and generally subjected to a common-pool resource type of appropriation and management regimes there is a large and increasing literature on economic valuation of wetland ecosystem goods and services (see for example, Barbier et al. 1997; Bardecki 1998; Kazmierczak 2001).

[6] Goods and services according to economics’ nomenclature.

According to the RAMSAR Convention, wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at

low tide does not exceed six meters. According to the Convention (Article 2.1) wetlands: may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six meters at low tide lying within the wetlands. Moreover, Brander et al (2006) classify wetlands into five types: mangroves, unvegetated sediment, salt/brackish marsh, freshwater marsh, and freshwater woodland.

Wetland ecosystems provide physical ecosystem services such as sediment retention, flood and storm control and other biological and socio-cultural functions, including local and global climate regulation, biodiversity preservation, habitat and shelter and natural amenities. In addition, wetlands allow the extraction of goods and services in the form of natural resources such as water, fish, wood, and energy, and they provide recreational services.

Most of the literature on wetland valuation is on specific wetland sites as Atkins and Burdon (2005) for the Randers Fjord, Denmark; Berrens et al (1996) for minimum instream flows in New Mexico; Bin and Polansky (2005) for wetlands in Carteret county, North Carolina; Birol et al (2005) for Cheimaditita Wetland, Greece; Broadhead (2000) for Garonne River, France; Cardoch and Day (2001) for the Mississippi Delta; Ferguson et al (1989) for Cowishan Estuary, British Columbia; Gren et al (1995) for Danube Floodplains; Kiker et al (1997) for the Gulf of Mexico; Kiker et al (2002) for natural lands in Duval, Clay, St. Johns and Putnam Counties; Lindsey et al (1999) for Crooked Creek Greenway, Indiana; Loomis (1987) for Mono Lake, California; Lynne et al (1981) for Florida´s Gulf Coast; Milon et al (1999) for the Everglades, South Florida; Tkac (2002) for Alfred Bog Wetland, Ontario, Canada; and Whitehead and Groothuis (1992) for the Tar-Palmico River, North Carolina. Others studies are methodological: An (2000); Azevedo et al (2003); Bateman et al (2004); Dalecki et al (1993); Pate and Loomis (1997); Whitehead and Blomquist (1991).

Wetlands valuation studies that use value transfer rather than primary valuation techniques exist, among them: Farber and Costanza (1987), Dahuri (1991), Farber (1992), Gren (1995),Dharmaratne and Strand (2002) and Brenner et al. (2010). In addition, three wetland valuation meta-analyses already exist; Woodward and Wui (2001), Brouwer et al. (1999) and Brander et al. (2006). They collect information on a broad sample of studies of economic valuation of ecosystem services provided by wetlands, and use meta-analysis in order to estimate the economic value of the different services provided by these ecosystems. Basically the technique consists in gathering and analyzing studies of wetlands valuation and to relate the per hectare value with a series of explanatory variables. These explanatory variables vary according to the methodology used, (contingent valuation, hedonic prices, travel cost etc.), the characteristics of wetlands in the study site (area covered, coastal distance, etc.), the measure of valuation estimated (consumer surplus, producer surplus, etc.) and the type of ecosystem services valued (landscape, hunting, fishing, flood control, water purification, etc.). Per hectare values obtained in this fashion are regressed against the explanatory variables. Therefore it is possible to obtain the WTP associated with a particular type of ecosystem service.

For illustrating the use of the income adjustment methodology proposed in the following section we chose the study by Brander et al (2006) rather than the study by Woodward and Wui (2001), owing to the fact that the former includes a greater number of studies (89) against the latter (39), and henceforth it is more representative of the mean value of the ecosystem services. Similarly, the study by Brouwer et al (1999) is not considered here since they estimate WTP using different methodologies without additional controls. Finally, Brander et al. (2006) is the more compressive study so far, and has been applied in different contexts (Anieski and Wilson 2005). The wetland service categories used by Brander et al. (2006) are flood control and storm buffering, water supply, water quality, habitat and nursery service (specifically support for commercial fisheries and hunting), recreational hunting, recreational fishing, amenity and other recreational uses, materials, fuel wood, and biodiversity. The per hectare values of ecosystem services originally estimated by Brander et al (2006) are expressed in dollars of 2000 and are shown in the second column of the Table 1. These are the values that -after adjusting for inflation and PPP- will ought to be adjusted by the income differentials existing between countries involved in the benefit transfer, using an appropriate methodology. In the following section we propose a theoretically sound methodology and present an illustration of its application.

Table 1: Unit values (WTP) of ecosystem services provided by wetlands

Ecosystem service	US$/hectare per year (US$ 2000)*
Flood control	464
Recreational fishing	374
Amenities/recreation	492
Water quality	288
Biodiversity	214
Habitat/nursery	201
Recreational hunting	123
Water supply	45
Materials	45
Fuel wood	14

*Values of Brander et al. (2006)

6. A methodology for adjusting transferred values for income differentials

Differences in willingness to pay (WTP) for an environmental improvement or in willingness to accept (WTA) an environmental worsening may come from income differences between countries. Even though this would not be a problem for benefit transfer within a country or between countries with similar levels of income, it can be an important source of distortions for benefit transfer when the technique is applied with two countries showing significant differences in their income levels.

In order to apply to a policy site those wetland values derived from a study site we adjust the original monetary values from the study site by taking into consideration the differences in the income elasticity of the marginal willingness to pay (MWTP) − which shows the percentage change in the marginal willingness to pay due to a percentage change in income − which in generally will be different for two countries in different stages of development.[7]

When income elasticity is assumed constant, the possible effect of income differences might be controlled for using the income elasticity of marginal willingness to pay (IEMWTP) according to:[8]

$$WTP_{PS} = WTP_{SS}(GDP_{PS}/GDP_{SS})^{\varepsilon} \qquad (1)$$

where ε is the IEMWTP; WTP_{PS} is the WPT in the policy site (the country where the value is going to be applied); WTP_{SS} is the WTP in the study site (the country where the value transferred was originally calculated and is taken from to be transferred); and, GDP_{PS} and GDP_{SS} are the per capita GDP in PPP dollars in the policy site and the study site, respectively.

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7 Throughout this paper we assume that marginal willingness to pay is equal to average willingness to pay. In other words each ecosystem services is provided under the assumption of constant returns to scale (see Brander et al 2006)

8 For presentation in this paper willingness to pay refers to marginal willingness to pay, otherwise it is explicitly stated.

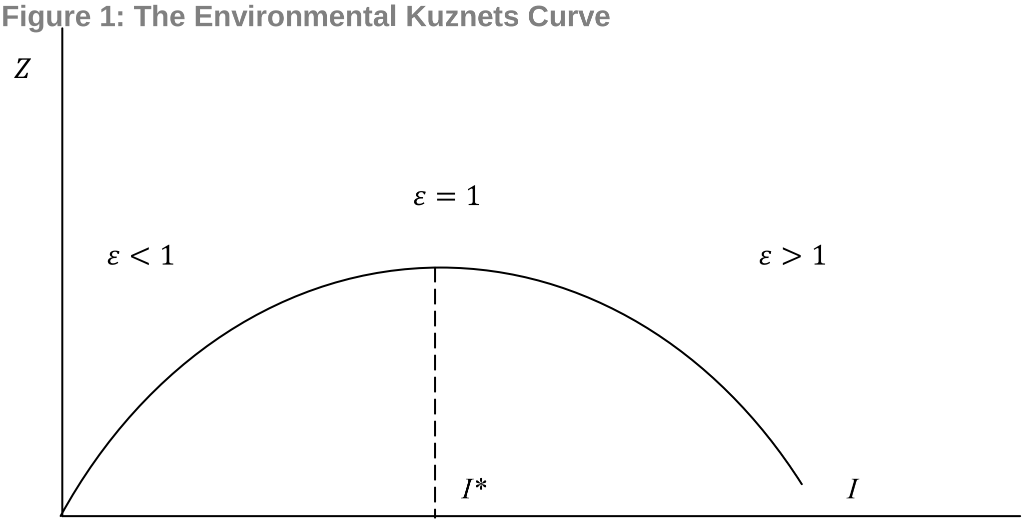
Most studies assume a constant IEMWTP. In general, estimations of this elasticity reported in the literature are between 0 and 1, and in several studies it is assumed equal to one. Regulatory agencies in the US adjust the transferred values estimating the proportional change in the WTP due to a proportional change in income. The EPA uses a distribution of values to account for the uncertainty in the estimation of the value of a statistical life (VSL); with a mode of 0.40 and endpoints at 0.08 and 1.00 based on a 1999 review of the literature (Industrial Economist 1999, EPA 1999).[9] The Department of Transportation, in turn, applies an income elasticity of 0.55 based on Viscusy and Aldy (2003). The Department of Homeland Security (DHS) uses estimates from Robinson (2008), who adjusts estimates from Viscusi (2004) from the time when the data was collected (1997) to the year of the analysis (2007). The estimates from Viscusi and Aldy have a mean of 0.15 to 0.78.

Nevertheless, recent literature has stressed some problems with the assumption of a constant income-elasticity. Willig (1976) for example, shows that constant non-unitary income elasticities of individuals demand –a concept related to IEMWTP- must be equal to each other (also see Varian 1978). Also a constant elasticity is difficult to estimate and there is not agreement in its value (see the previous paragraph). It has some asymptotic problems in dynamic models (Weitzman 1998, Gollier 2009) and it is not theoretically consistent with the empirics of the environmental Kuznets curve.

The Environmental Kuznets curve is an observed empirically regularity showing that pollution increases with income up to a turning point after which pollution starts to decrease. Theoretical models stress the role of a non-constant but increasing elasticity of marginal willingness to pay (Lopez 1994, Coopeland and Taylor 2003, Lieb 2002, Figueroa and Pasten 2010).

Figure 1 shows the Environmental Kuznets Curve with income per capita (I) in the horizontal axis, pollution Z in the vertical axis is the emission level of a given pollutant and I^* indicates the income turning point at which point pollution starts to decrease with economic growth. In the Figure 1, ε is the income elasticity of the marginal willingness to pay (IEMWTP) that shows the percentage change in willingness to pay for an environmental improvement given a percentage change in income. The figure illustrates the fact that, according to theory, ε is less than one in the increasing part of the EKC, equal to one at the turning point, and greater than one in the decreasing part of the EKC. Since it is possible to estimate econometrically the income-pollution relationship as well as the turning point, it is also possible then to estimate ε from the parameters of the EKC as it is shown below.

Figure 1: The Environmental Kuznets Curve



[9] See EPA (2006) for an example of implementation.

An increasing income elasticity of the marginal willingness to pay implies not only the well documented fact that richer countries are willing to pay more than poorer ones for the same marginal improvement in environmental quality, but it also means that as a country becomes richer its marginal utility of income decline faster than in poor countries (mostly given by the fact that basic necessities are already satiated). The estimation of the EKC parameters allows us to estimate ε (IEMWTP) (or the related concept of the elasticity of the marginal utility of income), which makes it possible to adjust accordingly the WTP obtained from a study site to transfer it to a different policy site (country).

GDP per capita in purchasing power parity (PPP) terms and the specific elasticity of the marginal utility of income can be used to estimate the ratio of WTPs between both, the policy site and the study site as is shown below.

6.1 Estimation of the income elasticity of marginal willingness to pay (IEMWTP)

Coopeland and Taylor ((2003, C&T hereafter), develop a model where environmental demand changes with economic growth. One assumption of their model is that government translates the changing society's preferences into efficient environmental regulations under neutral economic growth. In these circumstances, a sufficient condition for an EKC to arise is an increasing in income elasticity of MWTP, ε (Lopez 1994; Coopeland and Taylor, 2003). If ε, marginal willingness to pay grows more rapidly than income and as a consequence pollution decreases with economic growth, otherwise pollution increases with economic growth.

As an example, C&T use the following indirect utility function:

$$V(p, I, Z) = C_1 - C_2 \exp\left(-\frac{I}{\lambda}\right) - dZ \quad (2)$$

where, V(·)is an indirect utility function; λ, C_1, C_2 and d are positive constants; *I* is real income, and *Z* is pollution emissions. Population *N* is indexed to be equal to one. Marginal willingness to pay for pollution reductions (or marginal damage for pollution increases) therefore is given by:

$$|MWPT = \frac{d\lambda}{C_2} \exp\left(\frac{I}{\lambda}\right) \quad (3)$$

According to this result, the income elasticity of the marginal WTP is given by:

$$\varepsilon = \frac{I}{\lambda} \quad (4)$$

From this expression the income elasticity of the marginal WTP, ε, is increasing in income. According to the literature ε = 1 at the income turning point. If we denote the income turning point by δ it is evident by (4) that λ=δ.

Assuming that for a given pollutant the only difference between both countries is given by income and the country-specific turning point δ , the ratio between the WTP in the policy site and WTP in the study site is going to be given by:

$$\frac{WTP_{PS}}{WTP_{SS}} = exp\left(\frac{I}{\delta} - \frac{I^*}{\delta^*}\right) = exp(\varepsilon - \varepsilon^*) \tag{5}$$

In (5), a superscript asterisk denotes parameters of the study site, while no asterisk denotes parameters of the policy site. Total income adjustment is then given by:

$$\theta(\rho, I, I^*) = exp\left(\frac{I}{\delta} - \frac{I^*}{\delta^*}\right) = exp(\varepsilon - \varepsilon^*) \tag{6}$$

Following C&T, if a Cobb-Douglas production function is used, the marginal WTP in (3) must be equal to the marginal product of pollution such that:

$$\frac{d\lambda}{C_2}\exp\left(\frac{I}{\lambda}\right) = \alpha\frac{I}{Z} \tag{7}$$

Resulting in the following closed form of the EKC adapted from C&T,

$$z = \frac{\alpha C_2}{d\lambda} I^{\mu} e^{-\frac{I}{\lambda}} \tag{8}$$

Where, Z are emissions per capita of a given pollutant; α, C_2, γ are parameters of the utility function in (2); α is the coefficient on emission from the Cobb-Douglas production function; I is income per capita; and, $\mu \geq 1$ (C&T assumes $\mu = 1$ which imposes a restriction on the regression (9) below).

Taking logs at both sides of (8) it is possible to estimate the parameters of interest running the following regression:

$$lnz = \beta_0 + \beta_1 lnI + \beta_2 I + v \tag{9}$$

where, v is a random error term,

$$\beta_0 = ln\left(\frac{\alpha C_2}{\gamma\theta}\right); \quad \beta_1 = \mu \quad ; \beta_2 = -\frac{1}{\lambda} \tag{10}$$

and the corresponding turning point, $\delta,$ is given by:

$$\delta = -\frac{\beta_1}{\beta_2} = \lambda\mu \tag{11}$$

finally, the income elasticity of the marginal WTP (IEMWTP) is:[10]

$$\varepsilon = \frac{I}{\delta} \tag{12}$$

In order to estimate (6) for any two pair of countries, a set of 14 OLS regressions is performed considering 12 western European countries plus U.S and Canada. This is the same dataset used by Markandya et al (2006) in their analysis of the EKC for 12 western countries, with sulfur dioxide (SO2) emissions data used as a proxy for (negative) environmental quality and GDP per capita for about 150 years per country. Sulfur dioxide emissions data cover the periods from

[10] In C&T this elasticity is given by I/λ in our setting is given by $1/\delta = 1/\mu\lambda$. Obviously, both are the same as long as $\mu = 1$ but our empirical results show that $\mu > 1$ and hence the adjustment.

1850 to 2000 and have a common unit of kilograms per annum. The source of the time-series data for all 14 countries were compiled by Stern (2005a–c) using a combination of published and reported estimates from several sources. In particular, the data from 1990 to 2000 were obtained from OECD (2004).

Per capita GDP data is measured in 1990 international Geary-Khamis dollars and were compiled by Maddison (2005). Differently to Markandya et al paper we do not fill gaps where GDP is missed and we treat those values as missing values. This particular dataset has been used by Figueroa and Pasten (2009), where specific turning points where estimated that can be compared with the results of the estimation in this paper. Markandya et al (2006) used a quadratic specification for the estimation of the EKC which is different to our semi logarithmic estimation in (9) that has microeconomic theoretical foundations, in contrast to the quadratic specification that it is not grounded in economic theory. Therefore one of the objectives of this paper is to estimate for first time a nonlinear relation (such the one in (9)) that departs from the usual quadratic or cubic estimations and which, at the same time, is grounded in the already empirically tested economic theory of the EKC. However, the main objective of this section is to have an estimate of the income adjustment factor $-\theta(\delta, I, I^{*})-$ to be used for correcting the values being transferred between any two pair of countries involved in a BT exercise. In order to estimate the parameter δ in (6), to deal with the heterogeneity in the sample and as a mean to compare with previous results, longitudinal panel data analyses were performed. Table 2 shows the results obtained.

Table 2: Estimated Coefficients for SO2 in 12 European Countries + USA and Canada

Independent Variables lnz	Fixed Effects	Random Effects	Random Coefficients
Ln (Income) (β_2)	3.91***	3.92***	3.59***
	(0.08)	(0.08)	(0.4)
Income (β_2)	-0.0004***	-0.0004***	-0.0004***
	(0.000006)	(0.000006)	(0.00003)
Trend	-0.008***	-0.008***	-0.004
	(0.001)	(0.001)	(0.006)
Constant (β_0)	-13.3***	-17.21***	-17.4**
	(1.48)	(1.49)	(8.8)
λ	2 500	2 500	2 500
Turning point (δ)	9 775	9 800	8 975

Notes: *, **, ***, denote significance at the 10 percent, 5 percent, and 1 percent levels respectively. Numbers in parentheses are robust standard errors.

As it is shown in Table 2, the relevant coefficients (i.e. coefficients on income and the natural logarithm of income) are highly significant at 1 % critical value. Moreover, the estimated turning points are all plausible since in most of the cases found in the literature the estimated value of the turning point range between $ 8 200 and $ 10 600 in 1990 PPP dollars (see Cole et al 1997, Selden and Song 1994, Stern and Common 2001, Halkos 2001 and Figueroa and Pasten 2009). In particular, Stern and Common (2001) (S&C hereafter) found, with a fixed specification, a turning point of $ 9 239 and, with random effect, a turning point of $ 9 161 which are remarkable closer to the turning point values estimated here, even though we used a different specification and a larger sample (150 years rather than 30 in S&C) and a narrower number of countries (14 against the 23 OECD countries considered by S&C). Figueroa and Pasten (2009) use a random coefficient model and found a turning point of $ 12 776 with different specification and sample of countries. Markandya et al (2006) found a rather larger turning point of about $ 11 900 with the same sample but different specification and two less countries (Canada and US). In order to take account of the larger heterogeneity in the data, we run OLS country level regressions with the specification in (9).

Table 3: Estimated country level regressions for SO2 in 12 European Countries + USA and Canada

Country	Ln (Income) (β_1)	Income (β_2)	Turning point
Austria	3.71***	-0.0004***	9 275
	(0.26)	(0.00003)	
Belgium	3.61***	-0.0004***	9 025
	(0.2)	(0.00002)	
Denmark	5.09***	-0.0004***	12 725
	(0.40)	(0.00002)	
Finland	3.66***	-0.0005***	7 320
	(0.36)	(0.00003)	
France	2.84***	-0.0003***	9 467
	(0.16)	(0.00001)	
Germany	2.73***	0.0004***	6 825
	(0.25)	(0.00002)	
Italy	2.77***	0.0002***	13 850
	(0.47)	(0.00005)	
Netherlands	3.80***	-0.0004***	9 500
	(0.18)	(0.00002)	

Norway	1.30***	-0.0003***	4 333
	(0.28)	(0.00002)	
Sweden	6.05	-0.0005***	12 100
	(15.86)	(0.00002)	
Switzerland	6.59***	-0.0004***	16 475
	(0.56)	(0.00003)	
United Kingdom	3.37***	-0.0003***	11 233
	(0.22)	(0.00001)	
United States	2.30***	-0.0002***	11 500
	(0.28)	(0.00001)	
Canada	2.53***	-0.0004***	6 325
	(0.29)	(0.00002)	

Notes: *, **, *, denote significance at the 10 percent, 5 percent, and 1 percent levels respectively. Numbers in parentheses are robust standard errors.**

As it can be seen from the Table 3, with the only exception of Sweden where the evidence of an EKC is a little weaker, in the rest of the countries, the coefficient are highly significant at 1% critical level giving strong support to the existence of a n EKC based on the specification of C&T. Moreover, all the turning points are attainable given the average per capita GDP of the country group from 1950 to 2000, which is about $ 13 175.

Table 4 shows for each country in the sample the turning point displayed in Table 3 (column 2), the average income between 1950 and 2000 in PPP dollars (column 3) and the income elasticity of the marginal WTP estimated, according to equation (4), as the ratio between the average GDP per capita and the income turning point.

Table 4: Country level estimation of the income elasticity of marginal willingness to pay

Country	Turning point (δ)	Average income 1950-2000	Elasticity of WTP
Austria	9 275	11 602	1.3
Belgium	9 025	12 223	1.4
Denmark	12 725	13 992	1.1
Finland	7 320	11 060	1.5
France	9 467	12 608	1.3
Germany	6 825	11 933	1.7
Italy	13 850	11 049	0.8
Netherlands	9 500	12 961	1.4
Norway	4 333	13 253	3.1
Sweden	12 100	13 391	1.1
Switzerland	16 475	16 697	1.0
United Kingdom	11 233	12 333	1.1
United States	11 500	17 350	1.5
Canada	6 325	14 005	2.2

According to equation (6), it is possible to adjust the WTP obtained in a study site to transfer it as the WTP for a policy site using the information provided in the last column of Table 4. For example, the adjustment factor (AF) for a value obtained in USA to be applied to Germany can be estimated according to (6) by:

$$AF = \exp\left(\varepsilon\_GER - \varepsilon\_USA\right) = \exp\left(1.7 - 1.5\right) = 1.22$$

Therefore, the willingness to pay estimated in the USA has to be adjusted upwards in 22% to transfer it to Germany. These results are consistent with intercountry comparisons of the statistical value of life. Even though it is a different context, mortality and morbidity are the main components in pollution damage evaluation so the VSL is a good indicator of the WTP for air quality improvements. According to Miller (2000) the predicted worldwide VSL is between $ 0.6M and $ 0.9M per capita. It is between $ 1.6M and $ 2.6M for North America and consistently with our results is higher for the European Union $ 2.5M and $ 3.6M in constant PPP 1996 dollars. The procedure highlighted in this section shows that even being the US the country with the highest income per capita in the sample it is not necessarily the country with the highest willingness to pay as the procedure with constant income elasticity would suggest. Overall, willingness to pay is in average larger in Europe than in USA. However, in a country by country basis, for some countries the WTP is larger than in the USA and lower for others.

Table 5 shows the factors of adjustment to be applied between selected countries in Europe and North America. The first column indicates countries as study sites (where the value is being taken) and the first row indicates countries as the policy site (where the value is applied). The cells of the table show the adjustment factor for the pair of countries involved. For example the cell formed by the France in the row and Germany in the column displaying a value of 1.5, indicates that any value taken from France should be adjusted upward 50% to obtaining the corresponding value to be transferred to Germany.

As a final example, Table 6 shows in the second column the estimated values per hectare of ecosystem services provided by wetlands as proposed by Brander et al (2006) and, in the third column, the estimated WTP in Netherlands for similar ecosystem based on the adjustment factors estimated in here.

Table 5: Factor of adjustment between the study site and the policy site

	United States	Canada	Austria	France	Germany	Italy	Netherlands
United States	1.0	2.0	0.8	0.8	1.3	0.5	0.9
Canada	0.5	1.0	0.4	0.4	0.6	0.2	0.4
Austria	1.3	2.6	1.0	1.1	1.6	0.6	1.1
France	1.2	2.4	0.9	1.0	1.5	0.6	1.0
Germany	0.8	1.6	0.6	0.7	1.0	0.4	0.7
Italy	2.0	4.1	1.6	1.7	2.6	1.0	1.8
Netherlands	1.2	2.3	0.9	1.0	1.5	0.6	1.0

Table 6: Unit values (WTP) of ecosystem services provided by wetlands

Ecosystem service	US$/hectare per year (US$ 2000)*	Willingness to Pay in Netherland US$/hectare per year (US$ 2000)*
Flood control	464	418
Recreational fishing	374	337
Amenities/recreation	492	443
Water quality	288	259
Biodiversity	214	193
Habitat/nursery	201	181
Recreational hunting	123	111
Water supply	45	41
Materials	45	41
Fuel wood	14	13

***Values of Brander et al (2006)**

7. Conclusions

Wetlands are among the most productive ecosystems of the planet, and its relevance arises not only from the large amount of biodiversity they maintain but also from their crucial roles in sustaining Earth's balances that make life possible. In spite of that, wetlands are currently one of the most affected planetary ecosystems by the increase in the scale of human activities, the changes in use of vast areas of land and coasts and the mounting polluting discharges from urban as well as agricultural activities. As a result of this, economically valuing wetlands is becoming more urgent and relevant every day, so society and policymakers are capable of performing better and more informed cost-benefit analysis in order to change priorities and improve wetland conservation measures and policies by taking due consideration of the economic value of wetlands' contribution to human survival and well-being.

This paper presented a review of benefit transfer, one of the most used techniques to value ecosystems goods and services in general, and in particular those provided by wetlands, with emphasis on the necessary adjustments that ought to be done to the transferred values from the study site to make them applicable to the policy site.

A novel methodology, theoretically grounded on the preferences structure underlying the empirically tested environmental Kuznets curve phenomenon, is proposed here to perform the necessary adjustment for income differences between the countries involved in a benefit transfer exercise. As it is shown, this methodology for properly adjusting the transferred values can be quite important when benefit transfer is used for valuing wetlands and the countries involved are in different stages of their economic development, so there are large differences in their PPP per capita income levels. Without doubt the methodology proposed here opens a wide avenue to explore new and better specifications of preferences that could allow a better fit between the empirical findings coming from better data sets available and more plausible and convincing theories that conform to the new findings in the areas of economic behavior and experimental economics.

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