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Economic valuation of aquatic ecosystem services in developing countries

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Abstract

An important challenge of integrated water resources management (IWRM) is to balance water allocation between different users. While economically and/or politically powerful users have well developed methods for quantifying and justifying their water needs, this is not the case for ecosystems—the silent water user. A promising way of placing aquatic ecosystems on the water agenda is by economic valuation of services sustained by ecosystems. In developing countries, the livelihoods of rural people often depend directly on the provision of aquatic ecosystem services. In such situations, economic valuation of ecosystem services becomes particularly challenging. This paper reviews recent literature on economic valuation of aquatic ecosystem services in developing countries. "Market price" is the most widespread method used for valuating marketed ecosystem services in developing countries. "Cost based" and "revealed preference" methods are frequently used when ecosystem services are non-marketed. A review of 27 existing valuation studies reveals a considerable range of estimated total economic value of aquatic ecosystem services in developing countries, that is from US$30 to 3,000/ha/year. The paper concludes that economic valuation is vital for bringing ecosystems to decision-making agendas in developing countries and that great effort must be made to bridge the gap between scientists and decision makers.

Keywords: Decision-making; Developing countries; Economic valuation; Ecosystem services; Integrated water resources management

Introduction

The flows of the world's rivers are increasingly being modified through impoundments such as dams and weirs, abstractions for agriculture and urban supply, drainage return flows, maintenance of flows for navigation and structures for flood control. These interventions have caused significant alteration of flow regimes mainly by reducing the total flow and affecting the variability and seasonality of flows. It is estimated that more than 60% of the world's rivers are fragmented by hydrological alterations

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(Ravenga, 2000). This has led to widespread degradation of aquatic ecosystems and the services they provide in support of human well-being (Millenium Ecosystem Assessment, 2005; Forslund *et al*., 2009).

High economic costs, in terms of declining profits, remedial measures, damage repairs and lost opportunities, are associated with this degradation of ecosystems. The highest costs, however, are typically borne by people depending directly on ecosystem services to maintain their livelihoods. These people are generally among the poorest in developing countries. (Emerton & Bos, 2005; Millenium Ecosystem Assessment, 2005; Pearce *et al.*, 2006).

An important challenge of integrated water resources management (IWRM) is to balance water allocation between different users and uses. While economically and/or politically powerful users have relatively well developed methods for quantifying and justifying their water needs, this is not the case for ecosystems—the silent water user.

One of the most promising ways of placing aquatic ecosystems on the water agenda is by economic valuation of services sustained by ecosystems. In this way ecosystem services can be compared to those in other sectors and internalised in decision-making processes. Also, economic valuation of ecosystems serves several other purposes than attempting to internalise externalities and secure efficient decisions. It places ecosystems services on various practical policy-making agendas: the UN Millenium Development Goals (MDGs), poverty reduction, sustainability, equity and so on (Pearce *et al.*, 2006; Forslund *et al*., 2009).

Economic valuation of ecosystems is founded on an anthropocentric, utilitarian ideology with the goal of maximising individual (or societal) utility (Turner *et al.*, 1994; Bockstael *et al.*, 2000; Farber *et al.*, 2002). Some environmentalists adopt a non-anthropocentric, ecocentric ideology and argue that humans are not capable of setting a price on ecosystems (Turner *et al.*, 1994; Costanza, 2003). Thus, they fundamentally reject economic valuation of ecosystem services. Instead, they believe in intrinsic value—a value residing in ecosystems independently of human preferences. Whether or not an intrinsic value exists, it cannot be empirically quantified by humans and, therefore, has no operational value.

While economic valuation of ecosystem services can help identify trade-offs, economic valuation cannot stand alone when it comes to negotiating trade-offs and management options in IWRM. In this case, economic valuation must be placed in a broader decision-making context that may include non-commensurate values and societal priorities. Shared vision planning (SVP) is an example of a collaborative approach to improving such water management decisions (US Army Corps of Engineers, Institute for Water Resources, 2008). SVP combines traditional water resources management with structured public participation and collaborative computer modeling. As such, SVP provides a consensus-based forum for stakeholders to identify trade-offs (commensurate as well as non-commensurate) and management options. Several other multi-criteria decision support systems exist and economic valuation is an important, albeit not exclusive part of these water management trade-off tools.

Nevertheless, for the purpose of informing decision making in an IWRM context, there is an urgent need to identify all the possible services provided by aquatic ecosystems and to estimate their economic value. This need is particularly pressing in developing countries where livelihoods of rural people often depend directly on the provision of aquatic ecosystem services and where economically powerful sectors often set the agenda. However, none of the earlier literature reviews of economic valuation of ecosystem services (Bockstael *et al.*, 2000; Petersen, 2003; Silva & Pagiola, 2003; Turner *et al.*, 2003; Pagiola *et al.*, 2004; Schuyt & Brander, 2004; Emerton & Bos, 2005; Newcome *et al.*, 2005) have been focussed on aquatic ecosystem services in developing countries.

This paper fills that gap by reviewing recent literature on economic valuation of aquatic ecosystem services in developing countries. The aim is to provide an overview of the state of the art and to identify the main challenges. In the next section the most commonly used valuation methods are identified, followed by a review of the recent results of valuation studies relevant to water resource management in developing countries. In the fourth section a number of issues related both to performing valuation studies and to using results from these in policy analysis/decision-making are discussed. Finally, the findings are summarised and conclusions are drawn.

Methods for economic valuation of ecosystem services

When talking about economic valuation studies, most economists associate these with methods employed for valuation of non-marketed goods and services, such as biodiversity protection, provision of recreational goods and non-use values. However, in developing countries, non-marketed goods cover a much broader range. This is because many goods commonly traded on markets in developed countries are not necessarily subject to market transactions or traded on poorly functioning markets in developing countries. Nonetheless, in developing countries, market price (MP) is still the most widespread method used for valuating marketed ecosystem services (Bockstael *et al.*, 2000; Petersen, 2003; Silva & Pagiola, 2003; Turner *et al.*, 2003; Pagiola *et al.*, 2004; Schuyt & Brander, 2004; Emerton & Bos, 2005; Newcome *et al.*, 2005). But when using this method in developing countries, major problems are caused by market distortions, limited access to markets and subsistence use.

Dose–response (DR), factor income (FI) and effect-on-production are all related methods of valuation. The dose–response method assesses the effects of changes in quality/quantity of ecosystem services on the profitability/size of related productions/outputs. When only addressing the effects on marketed (commercial) production, the method is sometimes referred to as “effect on production”. Similarly, when addressing effects on income it is referred to as “factor income”. These relatively simple methods are commonly used. The main challenge remains that of establishing a correct dose–response relationship.

A market price-based approach employed for non-marketed goods is the use of “shadow prices”, which includes replacement cost (RC), avoided cost (AC) and mitigative expenditure (ME). This approach ha been heavily criticised, but is nevertheless widely used. In a review, the National Research Council (2005) concludes that replacement cost methods are “not valid approaches to determining benefits and should not be employed to value aquatic ecosystem services”. But the same authors include mainly cases using replacement cost (7 out of 14) in their review. This reflects a general tendency in the literature to criticise existing methods without providing good alternatives.

Of the methods used for valuation of non-marketed environmental services, travel cost (TC) is often applied to estimate recreational values. The main point of concern, when applying this method in developing countries, is that the value to local people may be underrated. “Stated preference” methods, such as contingent valuation (CV), are the preferred methods for evaluating non-marketed services. However, such methods require people to be familiar with the concept of money. In relation to *ex ante* valuation, it is difficult for people to value trade-offs they have not personally experienced. Furthermore, the budget constraint (ability to pay) of poor people can be inhibitory to any realistic expression of value. Hence the preferences of wealthy people may get a higher weight than that of poor people (Merrett, 2005; Pearce *et al.*, 2006). Consequently, “stated preference” methods are problematic in the context of developing countries and subsidence use.

Benefit transfer (BT) is not a valuation method as such but an easy desktop approach to estimating values based on existing studies. Benefit transfer is extensively used in developing countries (Silva & Pagiola, 2003; Newcome *et al.*, 2005). Unless the transfer is well justified (e.g. the two contexts are comparable) or appropriately adjusted (using context specific data), benefit transfer may produce very poor results. In many applications in developing countries, these prerequisites are not accounted for.

Despite the shortcomings of all economic valuation methods, they have one significant virtue in common: they hold great potential for raising awareness about the roles and values of ecosystem services for human well-being. Some results of valuation studies in developing countries are reviewed in the following section.

Results of ecosystem valuation studies

Recent reviews of the literature on ecosystem valuation find that most studies focus on the valuation of a single ecosystem service at a certain point in time/state of development (Bockstael *et al.*, 2000; Petersen, 2003; Silva & Pagiola, 2003; Turner *et al.*, 2003; Pagiola *et al.*, 2004; Schuyt & Brander, 2004; Emerton & Bos, 2005). Furthermore, they conclude that aquatic and marine ecosystems are strongly underrepresented, particularly so in the context of developing countries.

Table 1 shows the results of a review of recent studies on valuation of aquatic ecosystems in developing countries. Global estimates based on meta-analysis are included for reference. The services provided are based on a comprehensive checklist prepared for integrated water resources management (IWRM) and the services are categorised according to the type of value they represent, for example direct use, indirect use or potentially all (e.g. direct and indirect use, option, bequest and existence). These value categories largely correspond to the function categories suggested by De Groot *et al.* (2002): production function, regulation function and information function. Table 1 is a first attempt to provide pragmatic state-of-the-art information on economic values of ecosystem services with the aim of assisting water resource managers in developing countries.

The values in Table 1 show a considerable scatter. This is to be expected as values are inherently method and context specific. The temporal, socioeconomic and spatial scales of the studies affect the resulting value. Also, different ecosystems provide different services. Therefore, the total values depend on the type of ecosystem being valued. Given all these causes of dissimilarity, it is noteworthy that the "total" values in the literature reviewed only varied by a factor of 100, in the interval from 30 to 3,000 US$/ha/yr.

The unit "net value per hectare per year" was chosen as most studies reported their results in this unit or valued services provided by a known area of wetlands. This unit is useful when the value of an ecosystem service is correlated to the size of an area. In a comprehensive meta-analysis of 39 studies of economic values of wetland services, Woodward & Wui (2001) conclude that "wetland area appears to have little impact on value per acre". Thus it is problematic to use the per hectare unit when other factors than area determine the value of ecosystem services. In the case of recreation, for example, a small but beautiful area may be much more valuable (i.e. attract more visitors) than a large monotonous area. Similarly, while the flood attenuation capacity of a wetland depends on wetland size, the actual value of flood attenuation is determined by the downstream characteristics (affected population and potential damages).

Table 1. Economic values of aquatic ecosystem services reported in recent literature.

Type of value	Service provided	Valuation method	Net economic values reported in literature* Global estimate (US$/ha/year)	Developing countries (US$/ha/year)	Developing countries (US$/capita/year)
Direct use	Water for people	MP	45[10], 2,000–7,500[25]	150[5], 50[14], 400[19]	10[1], 21[5], 1[14]
	Fish/shrimp/crabs (non-recreational)	MP	200[10]	95[1], 6[2], 150[3], 160[4], 17[5], 550[9],50[11], 750[16], 90[17], 80[20], 50[24]	95[1], 25[2], 65[3], 80[4], 40[5], 30[9],50[11], 10[17], 160[20], 50[24]
	Fertile land for flood-recession agriculture and grazing	DR, MP	40–520[25]	110[1], 10[2], 150[3], 2[4], 170[5], 3[8], 180[9], 9[11], 40[14], 10[19], 10[20], 370[24]	70[1], 40[2], 52[3], 10[4], 14[5], 15[8], 9[9], 9[11], 30[14], 30[20], 80[24]
	Wildlife (for food)	MP	40–520[25]	0.02[2], 12[5], 2[20], 320[24]	0.1[2], 50[5], 10[20], 70[24]
	Vegetables and fruits	MP	40–470[25]	1[3], 200[9]	4[3], 11[9]
	Fibre/organic raw material	MP	45[10]	30[1], 1[2], 40[3], 1[4], 3[5], 7[11], 15[19], 20[24]	20[1], 4[2], 25[3], 4[4], 20[11], 5[24]
	Medicine plants	MP		6[3]	2[3]
	Inorganic raw material	MP	25–160[25]	0.1[2]	1[2]
Indirect use	Chemical water quality control (purification capacity)	RC, ME	300[13], 60–6,700[25]	620[1], 20[2], 1,400[7], 40[9], 140[19]	20[1], 50[2], 8[7], 2[9]
	Physical water quality control	RC, ME			
	Flood mitigation	RC, ME, AC	460[10], 15–5,500[25]	1,700[1], 2[2], 30[3], 90[8], 1,400[9], 340[19]	20[1], 2[2], 2[3], 370[8], 75[9]
	Groundwater replenishment	RC, ME, AC		10[2], 90[14], 70[19]	25[2], 30[14]
	Health control	DC, DR			
	Pest control	RC, ME, AC			
	Erosion control	RC, ME, AC		120[3], 20[19]	7[3]
	Salinity control	RC, ME, AC			
	Prevention of acid soil development	RC, ME, AC			
	Carbon “trapping” (sequestration)	AC	130–270[25]	50[2], 2[3], 1,300[8], 2,000[11], 15[19]	120[2], 1[3], 9,000[8], 2,000[11]
	Microclimate stabilization	AC		10[19]	

Continued

Potentially all (direct and indirect use, option, bequest and existence)	Recreation and tourism (incl. fishing and hunting)	TC, CV	990[10], 230–3,000[25]	20[1], 260[11], 30[19]	20[1], 1,100[11]
	Biodiversity conservation	DR, CV, BT		0.6[8], 3,600[11], 30[19]	2[8], 2[11]
	Cultural/religious/historical/symbolic activities	CV, BT	30–1,800[25]	80[19]	
"Total" economic values	120–370[10]	2,600[1], 100[2], 340[3], 30[6], 350[5], 2,400[9], 2,900[11], 500[12], 50[13], 2,000[15], 1,800[16], 110[24], 700[18], 1,100[26], 90[20], 230[21], 1,200[22], 500[23], 760[24]	150[1], 50[2], 30[3], 41[5], 45–76[6], 125[9], 90[11], 70[12], 10[24], 10[18], 200[20], 230[23], 200[24]		

* Note that "per hectare" refers to the area providing the service and "per capita" refers to the people benefiting from that service. Only for some services this corresponds to total area or total population. Total value, therefore, may not be equal to the sum of the individual values.

[1]Emerton & Kekulandala (2002), [2]Turpie *et al.* (1999), [3]Emerton *et al.* (2002), [4]Iftikhar (2002), [5]Emerton (1994), [6]IUCN (2001), [7]Emerton *et al.* (1999), [8]Rosales *et al.* (2003), [9]Gerrard (2004), [10]Schuyt & Brander (2004), [11]Busk (2002), [12]Karanja *et al.* (2001), [13]Seyam *et al.* (2001), [14]Acharya (2000), [15]Woodward & Wui (2001), [16]MRCS (1998), [17]Ringler & Cai (2003), [18]Pyo (2002), [19]Seidl & Moraes (2000), [20]Schuyt & Jansen (1999), [21]Christensen (1982), [22]Sathirathai (1998), [23]Bann (1997), [24]Drew *et al.* (2005) and [25]Costanza *et al.* (1997).

The majority of the reviewed studies provide information on the affected population, allowing a conversion of the per hectare values into per capita values. For most of the ecosystem services, this results in more consistent values and the "total" values are within the range from 10 to 230 US$/capita/year. Woodward & Wui (2001) did not have the information required to include population in their meta-analysis, but they did recognise that "characteristics of the population near a wetland are particularly likely to influence the value placed in the area". The present review strongly supports this statement (see for example "flood mitigation" in Table 1). In general terms, it could be argued that the potential value of an ecosystem service is a function of ecosystem (or biophysical) characteristics (e.g. size) while the actual value (the extent to which the potential value is utilised) is a function of population (or socioeconomic) characteristics. This highlights the importance of addressing "nature's household" (ecology) as well as "humankind's household" (economics) when attempting to value ecosystem services.

It should be noted that the values shown in Table 1 may be affected by a selection bias (Woodward & Wui, 2001). Ecosystems that are considered valuable are more likely to be subjected to valuation than seemingly insignificant ecosystems. Hence highly valued ecosystems are probably over-represented in Table 1.

In addition to the methodological challenges described above, economic valuation of ecosystems is faced with some general challenges. These are discussed in the following section.

Challenges for valuation

While academic/scientific societies hold a widespread recognition of the economic value of ecosystem services, this is generally not the case for decision makers (Russell *et al.*, 2001; Costanza, 2003; Emerton & Bos, 2005; Millenium Ecosystem Assessment, 2005; Pearce *et al.*, 2006). Some argue that the main challenge for ecosystem valuation is bridging the gap between ecologists and economists and their respective perceptions of value (De Groot *et al.*, 2002; Newcome *et al.*, 2005; Pearce *et al.*, 2006). This challenge has, however, been partly offset by the recent focus on ecosystem services as the linkage between ecological functions and economic values as well as the common scientific acceptance of the total economic value (TEV) as reference framework. Nevertheless, several challenges remain.

Acknowledging the assumptions of marginality and substitutability

There are two important assumptions underlying conventional economic valuation: marginality and substitutability. Both assumptions are critical for valuing ecosystem services in developing countries.

The change to be valued must be marginal. Attempting to do economic valuation based on a non-marginal change creates both conceptual and practical problems. Conceptually, the change itself may be meaningless. Practically, the change may alter the unit being used in valuation (e.g. existing market prices may be affected and cannot be used for valuation).

Whether a change can be considered marginal or not, is entirely dependent on the scale of the evaluation study. Furthermore, when dealing with ecosystem services, there is not always a straightforward relationship between impact and resulting change (Limburg *et al.*, 2002). Therefore, it can be difficult to judge the marginality of a change. What is considered a marginal change may, in time, turn out to be a total collapse (Limburg *et al.*, 2002).

The utilitarian principle of substitutability implies that all values (types of capital) are substitutable or replaceable. This is indicative of the so-called weak sustainability approach (Turner *et al.*, 1994). In the case of subsistence use of ecosystem services, this assumption may not be valid. In developing countries, many rural people's livelihoods depend directly on the provision of ecosystem services. Often, these people are poor and should the ecosystems deteriorate, they have few alternatives. In such cases, the value of vital ecosystem services (e.g. fish production) may approach the value of life or the cost of changing livelihood strategy. The notion of critical natural capital (symptomatic of strong sustainability) may be accounted for by assigning extremely high (infinite) values to such natural capital (e.g. ecosystem services).

Using "total" economic values

The "total" economic values reported in Table 1 are not necessarily equal to the common notion of TEV. TEV is the sum of "all" individual ecosystem services whereas the "total" values in Table 1 are sums of selected services, albeit often the most important and valuable services. Therefore, they may considerably underestimate TEV. Also, ecosystem services may be non-additive and simply adding their values may underestimate the "true" TEV of ecosystems (Bockstael *et al.*, 2000). In other words, healthy ecosystems are a prerequisite for the provision of all other services and thus can be said to possess a monetary value. The values reported in the literature largely fail to acknowledge this overarching life-support service of ecosystems. On the other hand, there is a risk of overestimating TEV if individual services overlap and double-counting occurs (De Groot *et al.*, 2002). This may be the case if both biodiversity and recreational values are valued separately.

Furthermore, it is argued that TEV ignores an intrinsic value residing in ecosystems, independently of human preferences (Turner *et al.*, 1994). Whether or not an intrinsic value exists, it cannot be empirically quantified by humans and, therefore, has no operational value. The existence value of TEV may capture parts of the intrinsic value and is sometimes termed the anthropocentric intrinsic value (Turner *et al.*, 2003). Often, however, existence value and intrinsic value are incorrectly used as synonyms (Emerton & Bos, 2005; National Research Council, 2005).

Another important issue emerging from the present review is the confusion regarding the terms total and marginal values. In the reviewed literature, total values are in reality the aggregates of marginal values over some non-critical range. Otherwise the assumption of marginality would be violated (see discussion above). Some argue that marginal value approaches infinity below a certain minimum provision of ecosystem services (Turner *et al.*, 2003) while others reason that the concept of TEV is meaningless below such a minimum provision (Pearce *et al.*, 2006). The latter assertion is supported by the underlying assumptions of marginal change and substitutability as well as the practical/operational notion of budget constraint. In any case, attempting to estimate the total economic value of ecosystems may be meaningless without acknowledging the non-critical range. In the literature, the levels of service provision defining this non-critical range are often not apparent. Also, marginal values are often approximated by average values. This can only be justified if there is a linear relationship between level of service provision and value. In this case, the marginal value will be constant and equal to the average value. Such assumptions are rarely explicitly explained or accounted for.

Defining spatial, socio-economic and temporal scales

The benefits of ecosystems services can be far removed in time and space from the ecosystem that provides them. A clear and explicit definition of the spatial, socioeconomic and temporal scale is, therefore, a critical step in an economic valuation study. The scale is entirely dependent on the question being asked and addressed in the valuation study. Thus, in framing/scoping the valuation study it is important to clearly specify this question.

Spatial scale refers to the geographical extent of the service being valued as well as the location of the stakeholder beneficiaries, local and/or downstream. The *socioeconomic scale* concerns the value of ecosystem services to different groups of people within the same geographical location. Ecosystem services play different roles in people's livelihood strategy. Ecosystem services may produce socioeconomic spin-off effects, such as supporting social structures and employment and preventing pauperisation and conflicts. The extent to which such spin-off effects are included must be defined.

Temporal scale is important as the ecosystem services and their value may change over time. The temporal scale is predominantly an issue when assessing the impact of developments/changes. A considerable "time lag" may elapse before changes in ecosystems (and values) manifest themselves. Depending on the temporal scale, discounting must be considered. Discounting will not be further discussed here; readers are referred to Pearce *et al.* (2006).

The scales of a valuation study are determined by the type of economic analysis in question. Valuation in the context of a financial economic analysis differs significantly in scale from valuation in the context of a welfare economic analysis. Generally, as scale increases, valuation becomes more and more problematic both conceptually and empirically. Conceptual problems may arise if the marginality assumption is violated. Empirical challenges of increasing scale are related to the many uncertainties involved in valuation. This will be discussed in the following.

Dealing with uncertainties

The current knowledge of ecosystems is in many cases insufficient (Millenium Ecosystem Assessment, 2005). When subjected to changing conditions, ecosystems may exert non-linear and/or hysteretic behaviour. A change may cause cascading effects and lead to catastrophic and/or irreversible responses. On the other hand, some ecosystems may show strong resilience. It is, therefore, crucial to identify spatial/temporal thresholds and extrapolations can only be used with great caution (Limburg *et al.*, 2002; National Research Council, 2005).

While existing economic methods have advanced significantly, their inability to generate precise estimates of value persists. It is important to note that threshold effects, described above, can stimulate threshold effects in peoples' preferences. This further complicates valuation (Limburg *et al.*, 2002; National Research Council, 2005).

In the face of lack of knowledge and significant uncertainty, it is necessary to make expert judgements in ecosystem valuation. This suggests that in ecosystem valuation studies there is a strong case for including peer/stakeholder reviews that provide inputs and incorporate quality assurance. Furthermore, as uncertainties can be reduced over time through passive and/or active learning, it may be tempting to postpone a valuation study until more information is available. While this can be justified in relation to some decision-making processes (the "value" of delaying a decision is sometimes referred to as "quasi option value" (QOV) (Pearce *et al.*, 2006)) it may be counterproductive to the urgent need to demonstrate the roles and values of ecosystem services for human well-being.

Notwithstanding this urgent need, it is incumbent on the scientist to communicate assumptions and uncertainties and to test the reasonability of results. Uncertainties must be clearly and explicitly acknowledged, minimised and accounted for. A valuation study must never convey a false sense of precision. Unfortunately, this is often the case and far more significant digits than justified are frequently reported. Failing to accomplish the task of dealing with uncertainties potentially puts at risk the very credibility of ecosystem valuation.

Summary and conclusions

In developing countries, many rural people's livelihoods depend directly on the provision of ecosystem services. Often, these people are poor and they have few alternatives should the ecosystems deteriorate. In such situations, economic valuation of ecosystem services becomes particularly challenging. This paper has reviewed recent literature on economic valuation of aquatic ecosystem services in developing countries and provided an overview of the state of the art and the main challenges.

Most of the existing economic valuation methods can be applied in developing countries, albeit with caution. "Market price" is the most widespread method used for evaluating marketed ecosystem services, also in developing countries. Market distortions and limited access to markets are major problems when using this method in developing countries, where most ecosystem services are non-marketed. "Travel cost" is often applied to estimate recreational values. The main point of concern, when applying this method in developing countries, is that the value to local people is underrated. "Stated preference" methods are the preferred methods for valuating non-marketed services. However, the preferences of wealthy people may get a higher weight than that of poor people and subsistence use may not be accounted for at all. While "cost based" methods have been heavily criticised, they are widely used to estimate indirect use values. Provided that the conditions for using cost-based methods are met, these methods are useful. "Benefit transfer" is the easiest method, but requires contexts to be comparable.

A review of 27 existing valuation studies reveals a total economic value of aquatic ecosystem services in developing countries in the range from 30 to 3,000 US$/ha/year or from 10 to 230 US$/capita/year.

Four main challenges for valuation of ecosystems services are identified: acknowledging the assumptions of marginality and substitutability; using "total" economic value; defining spatial, socio-economic and temporal scale; and dealing with uncertainty. If these challenges are not well appreciated the valuation study may be misleading or meaningless, regardless of the method chosen. This should not lead to rejection of economic valuation of ecosystems, nor should it render scientists paralysed or tempted to convey a false sense of precision. Instead, it should encourage careful and explicit attention to the caveats of economic valuation of ecosystem services.

Also, while economic valuation of ecosystem services can help identify trade-offs, economic valuation cannot stand alone when it comes to negotiating trade-offs and management options in IWRM. In this case, economic valuation must be placed in a broader decision-making context, such as SVP or multi-criteria decision making systems that allow the inclusion of non commensurate values and societal priorities.

Despite the shortcomings of every economic valuation method, they have one significant virtue in common: they hold great potential for raising awareness about the roles and values of ecosystem services for human well-being. There is no excuse for not undertaking valuation studies. As long as decisions are being made that affect ecosystems, ecosystem valuation is being done, whether acknowledged or not. Explicitly assigning values to ecosystems services is better than assigning them a value of zero by default. It ensures higher levels of information and transparency in decision making. Even if the assigned value turns out to be incorrect, the worst thing that can happen is that a wrong decision is being made. Had valuation not been undertaken, there is a high risk that this would have been the case anyway.

It is, therefore, of utmost importance to encourage the wise use of economic valuation of ecosystem services in decision making. This entails ensuring that clear scopes and relevant assumptions and uncertainties are effectively communicated. Otherwise, the credibility of ecosystem valuation and thereby the potential for placing due value on ecosystem services, may be jeopardised. Thus, bridging the gap between scientists and decision makers is a crucial challenge for safeguarding ecosystems and the vital services they provide for humankind.

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