D20 - Identification key methodological issues

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SIXTH FRAMEWORK PROGRAMME

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Content

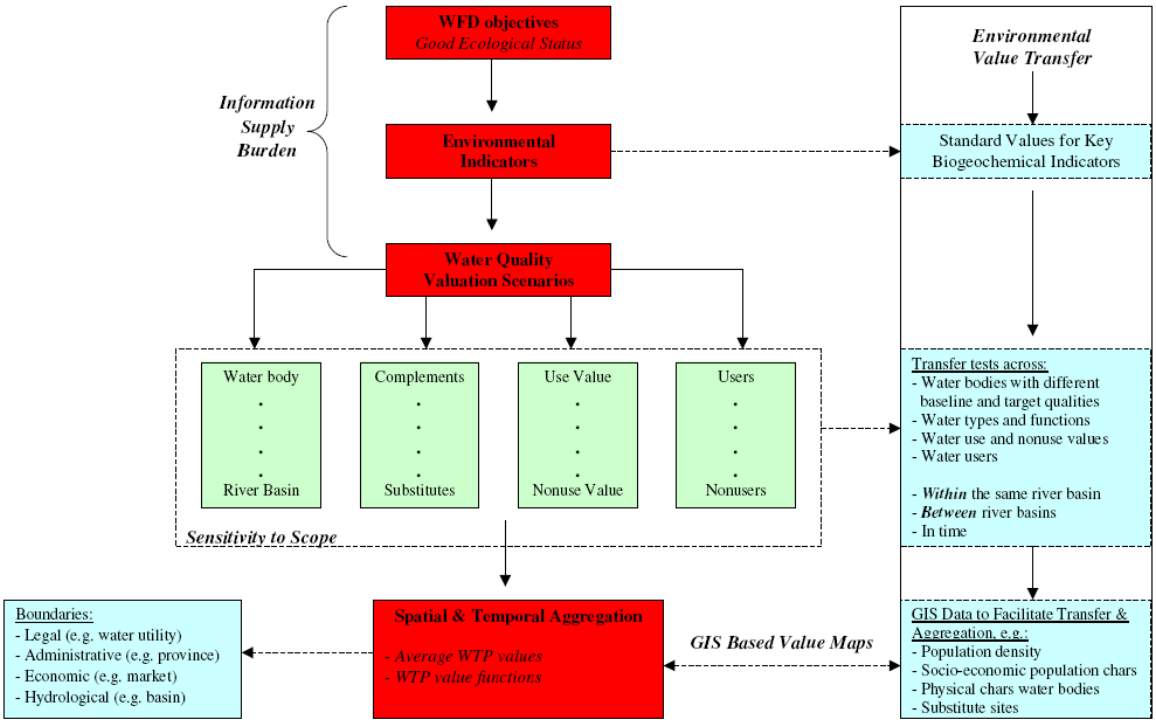
1. Introduction

The objective of this short report is to identify and briefly describe the key methodological issues in water resources valuation. Based on discussions with the AquaMoney team, the following key issues were identified (Figure 1):

1) Identification and definition of the goods and services to be valued.
2) Scale at which the goods and services are provided and valued.
3) Sensitivity of existing valuation methods to changes in the size of good and service provision.
4) Transferability of the estimated values for goods and services.

These key issues will be addressed and further elaborated in the different AquaMoney case studies.

Figure 1: Key methodological issues identified in AquaMoney



2. Good Ecological Status

Any economic valuation exercise for a specific environmental change in water is based on a scientific understanding of the underlying environmental dose-response or damage functions. Ideally, reaching *Good Ecological Status* (GES) for natural water bodies and *Good Ecological Potential* (GEP) for heavily modified and artificial water bodies is defined in clear physical *state* (or impact) parameter values based on available knowledge and information about changes in (the pressures exerted on) the water system on chemical and ecological target values (e.g. concentration levels of priority substances or abundance of a specific species in the water column). In practice, even if these state parameter values can be determined based on ecological references, they are often surrounded by a lot of uncertainty as a result of limited knowledge about complex water systems. This uncertainty relates to the uncertainty in dose-pathway-effect relationships of reaching GES or GEP with the help of the proposed means or measures and the uncertainty about the environmental characteristics of GES or GEP and the stability of such an environmental system given ecosystem dynamics.

The relationship between economic values and these underlying physical environmental changes (via outcomes, functions, goods and services) is not always straightforward and has a number of important implications for the design of the valuation study and the analysis, interpretation and presentation of the results. Researchers applying stated preference methods like contingent valuation (CV) or choice experiments (CE) face an important ‘information supply burden’. This means that they have to translate often complex and complicated scientific knowledge and information in lay public terms. Public participating in a stated preference survey should understand and be able to relate to the proposed changes in the water system in order to be able to derive meaningful answers and values. Scientific knowledge and information and the uncertainties surrounding this knowledge and information have to be translated into easy understandable terms in the valuation scenarios. The quality and quantity of information supplied in social survey research has important consequences for the outcome. Information effects have been tested extensively in the stated preference literature over the past decades. However, no specific, straightforward guidelines exist towards the optimal level and quality of information.

3. Scale

The importance of space in economic valuation is nowhere better illustrated than in travel cost or hedonic price studies where the economic value is determined by the cost price of travelling to a specific water site or encapsulated in property prices. For instance, the further away someone lives, the higher the travel cost and hence – so it is assumed – the value attached to the site involved (measured through WTP). However, space may play various roles in economic valuation studies. In principle, the spatial dimensions underlying economic valuation of water resources and their functions relate to:

1) the spatial distribution of individual water bodies across the watershed or river basin and their physical characteristics relevant for economic valuation (e.g. shape, size, function, quality, accessibility, distance etc.);

2) the relationship between spatially interconnected (e.g. upstream-downstream) water bodies in terms of their functional complementarity or substitutability;

3) the spatial distribution of the population of beneficiaries or polluters inside and outside the river basin and their specific characteristics relevant for economic valuation (e.g. income levels, perception of quality, distance and accessibility etc.).

Space has traditionally been interpreted in terms of distance between objects. For example, the distance between an angler's place of residence and the place where (s)he fishes. Public perception of space and distance may differ from the traditional 'objective' measures. Travel cost studies have shown that public perception of distance may differ significantly from the 'objectively' measured travel distance and provide a stronger explanatory factor of travel behaviour. Subjective location specific factors (e.g. place of birth) may play an important role in someone's value attached to a specific place, including the presence of water in a landscape. The spatial distribution of water bodies and their users and non-users is expected to affect the economic value attached to the different water bodies and the river basin as a whole in a number of ways.

4. Sensitivity to scope

For most non-market goods there can be no criterion validity test (Mitchell and Carson, 1989), i.e. there is no external objectively 'correct' measure (akin to a market price) against which we can assess study results. Given this, the broad approach to validity is to examine whether or not findings conform to prior expectations, with those expectations being principally drawn from economic theory. It was this reasoning that led the so called NOAA Blue-Ribbon Panel on contingent valuation (CV) to recommend that the principal validity test for CV studies should be the 'scope test' (Arrow et al., 1993). This is based upon the expectation that (Arrow et al., 1993, p.4604):

"[u]sually, though not always, it is reasonable to suppose that more of something regarded as good is better so long as an individual is not satiated. This is in general translated into a willingness to pay somewhat more for more of a good, as judged by the individual".

The NOAA Panel recommended a number of conditions for survey validity, of which "responsiveness to the scope of the environmental insult" (Arrow et al., 1993, p. 4614) has come to be "regarded by many as an acid test" (Carson et al., 1996, p. 3) of survey-derived values. However, as Banerjee and Murphy (2005) point out, sensitivity to scope is neither necessary nor sufficient for preference consistency. There are very few non-market goods for which we have prior expectations regarding what degree of increase in WTP might be reasonable. Indeed, given that individuals may become satiated with environmental goods at low levels of provision (e.g. it would be reasonable for a respondent to think that once they had access to one nearby clean river they were not willing to pay anything for a second), then the only expectation that economic theory provides for us is that WTP should not be negative for an increase in provision of a good. This is hardly an adequate test of study validity. The claim to validity arising from studies that merely report a statistically significant increase in WTP as scope increases seem insufficient (Bateman and Brouwer, 2006).

5. Value transfer

Value transfer exercises typically involve estimating the value of a given change in provision of an environmental good at some target 'policy site' from previous analysis undertaken at one or more 'study sites'. Analysts have for many years sought methods which will reduce decision costs and the extrapolation of assessments from one case to another is clearly attractive. Given the significant costs of valuing preferences for non-market goods it is not surprising that this area has now generated a considerable literature concerning the transfer of benefit and cost estimates, most particularly in the area of environmental valuation (Brouwer, 2000).

To date the most common approach for such study site analyses is some variant of stated preference (SP) technique such as contingent valuation (CV) or choice experiments (CE) (Bateman et al., 2002). Although CE are considered superior to CV for the purpose of value transfer (Morrison et al., 2002), the most fundamental problem for value transfers is not regarding which technique to apply, but rather in assessing whether a given transfer is correct or not when the 'true' value of the policy site is a-priori unknown. Because of this problem, the literature has placed great emphasis upon the development and testing of value transfer methods (e.g. Bergland et al., 1995; Downing and Ozuna, 1996; Brouwer and Spaninks, 1999; Ready et al., 2004). Here researchers typically undertake studies at two or more sites and test the effectiveness of a transfer method by dropping data from say one site (i.e. designating it as the policy site) and using information from remaining (study) sites to estimate values for the policy site. Errors are then assessed by comparing transferred value estimates with those obtained from the study conducted at the policy site itself.

The methods used for transferral can be broadly categorised into two types. The simplest approach is to attempt to find study sites which appear similar to the policy site and transfer mean values from the former to the latter (e.g. Muthke and Holm-Mueller, 2004). Such 'univariate' transfers are frequently used in practical decision making, but are crucially dependent upon the pertinence of differences between transfer sites. Clearly all sites are to some degree dissimilar (e.g. unique ecosystem habitats or the spatial pattern of substitutes around a site is unique); it is the degree to which this dissimilarity affects values which will determine the appropriateness of unconditional mean value transfers. It is because of such concerns that value function or 'multivariate' transfer approaches have been developed. Here statistical techniques are used to estimate value functions from study site data. These are then used to predict new values for policy sites. This is achieved by assuming that the underlying utility relationship embodied in the parameters of the estimated model applies not only to individuals at the study sites but also to those at policy sites. Usually, these parameters are kept constant, while the values of the explanatory variables to which they apply are allowed to vary in line with the conditions at the policy site. In those cases where results are used from studies carried out years ago, an important question obviously is to what extent preferences and parameters have changed (Brouwer, 2006).

Pearce et al. (1994) argue that because value function transfers allow the analyst greater control over differences across sites, they should in principle yield lower transfer errors than simple mean value transfers. However, empirical evidence regarding this assertion is mixed with sometimes the opposite result being observed (e.g. Bergland et al., 1995; Barton, 2002; Ready et al., 2004). This is partly due to the lack of a systematic assessment of a set of (theoretically driven) baseline conditions needed to be in place for valid and robust value transfer. In this paper, we argue that the reason for the perverse results reported in the literature is in part because of misspecification of empirical value functions.

In a previous cross-country comparison, Brouwer and Bateman (2005) show that a simple unadjusted unit value transfer works best for similar case study sites, while errors generated by simple mean value transfer are considerably larger than those arising from function transfer across dissimilar case study sites. As expected, if conditions are not the same across study and policy sites, some degree of adjustment helps reducing the error. In an international water quality transfer study based on unadjusted average values between Norway and Germany using the RFF water quality ladder (Muthke and Holm-Mueller, 2004), significant differences are found between WTP values exceeding error margins of 60% due to differences in population and site characteristics and baseline conditions that could not be controlled for. Similar large errors (87-130%) are found in Barton and Mourato (2003) when transferring WTP for coastal bathing water quality in Portugal and Costa Rica and reductions in associated health risks. Lower errors based on international transfer of unadjusted unit values of around 40% are found, on average, in Ready et al. (2004) in the context of ill health episodes across five European countries, and 25% in Rozan for improved air quality and related health impacts in France and Germany. Average transfer errors are around 30% for unadjusted value transfer and 10% for function transfer for dissimilar case study sites in Europe and New Zealand facing global health risks in Brouwer and Bateman (2005). Obviously, the accuracy of value transfer is conditioned on the measurement errors contained in original studies, and part of the measurement error transferred from the original value estimation may be amplified in a value transfer if care is not taken to minimize such effects (Wilson and Hoehn, 2006).

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