# A meta-analysis of wetland contingent valuation studies

R. Brouwer · I.H. Langford · I.J. Bateman · R.K. Turner

**Abstract** There is growing interest in the potential for producing generally applicable models for valuing non-market environmental services which do not rely upon expensive and time-consuming survey work, but rather extrapolate results from previous studies. This paper presents a meta-analysis for the use and non-use values generated by wetlands across North America and Europe. The study assesses the socio-economic values attributable to the hydrological, biogeochemical and ecological functions provided by such complex environmental assets. The clustering of multiple values derived from single studies is examined through the application of multilevel modelling methods allowing for the hierarchical structure of such data.

**Key words** Meta-analysis · Contingent valuation · Wetlands · Ecosystem functions

## Introduction

This paper addresses the socio-economic values of the various functions performed by wetland ecosystems. Environmental economists have developed a variety of techniques for measuring such values, of which the contingent valuation (CV) method is probably the most widely applied in contemporary research. CV is a survey method where individuals are presented with information about specific environmental changes, and their perception, attitudes and preferences regarding these changes are elicited. In order to measure the effect of the suggested changes on people's welfare, respondents are typically asked for either their willingness to pay (WTP) or their willingness to accept (WTA) compensation for the gains or losses involved (Mitchell and Carson 1989). Of these options the WTP approach has become the most frequently applied and has been given peer review endorsement through a variety of studies (see, for example, Arrow et al. 1993). When aggregated across those who will be affected by the suggested environmental changes, this stated WTP amount is used as a socio-economic indicator of the environmental values involved.

Given the substantial indirect, often off-site, use and non-use values involved, wetlands have been the focus of attention in several CV studies (Crowards and Turner 1996). Many of these studies try to estimate the total economic value of wetlands. Total economic value, not to be confused with total ecosystem value, consists of use and non-use values (Pearce and Turner 1990). CV is the only economic method to date that is able, in principle, to account for possible non-use motivations underlying people's value statements. Whereas use values refer to the values associated with the actual use of the various goods and services wetlands provide, non-use values are unrelated to any actual or potential use of these goods and services.

Wetlands are complex hydro-ecological systems, whose structure provides us with goods or products involving some direct utilisation of one or more wetland characteristics, while wetland ecosystem processes provide us with hydrological and ecological services, supporting or protecting human activities or human properties without being used directly. The stock of wetlands is a multi-functional resource with significant economic value, as also has been suggested by Costanza et al. (1997). Fifteen percent of the value of the world's ecosystem services and natural capital is generated by wetlands (Costanza et al. 1997). However, all over the world countries have experienced severe wetland losses (Tolba and El-Kholy 1992; Turner 1992). Sustainable management of these assets is highly relevant. Since this management process is not costless, they require accurate and meaningful valuation in order to be able to weigh the costs and benefits of their conservation.

In this paper, the main findings of CV studies of wetlands in temperate climate zones in developed economies will be investigated. The main objective is to quantify the socio-economic values associated with wetland ecosystem functioning in a *meta-analysis* of

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wetland CV studies, supplementing qualitative analyses provided, for example, by Gren and Söderqvist (1994) or Crowards and Turner (1996). Natural and social science are brought together by relating the various hydrological, biogeochemical and ecological wetland functions to the societal benefits derived from these functions and the socio-economic values attached to these benefits.

## Approach

The results from 30 different CV studies of wetlands in temperate climate zones in developed economies were compared and synthesised in a meta-analysis. Only very few tropical wetland valuation studies exist (Barbier 1993). Tropical wetland studies are excluded from the meta-analysis presented here because of the enormous differences between population samples in developed and developing countries regarding socio-cultural and demographic-economic characteristics.

Since the beginning of the 1990s, meta-analysis has been playing an increasingly important role in environmental economics research (van den Bergh et al. 1997). Originally a technique used in experimental medical treatment and psychotherapy, meta-analysis is the statistical evaluation of the summary findings of empirical studies, helping to extract information from large masses of data in order to quantify a more comprehensive assessment. It enables researchers to explain differences in outcomes found in single studies on the basis of differences in underlying assumptions, standards of design and/or measurement. As such, meta-analysis is an important extension of quantitative analyses and can be seen as a supplement to qualitative analysis.

Compared to qualitative analysis, important advantages of meta-analysis are that on the 'input' side it does not prejudge research findings on the basis of the original study's quality, while it avoids a differential subjective weighting of studies in the interpretation of a set of findings on the 'output' side (Glass et al. 1981). However, one drawback is that it may be biased towards including significant study results only, since it may well be that insignificant study results will not be published. Furthermore, multiple results from the same study are often treated as individual, independent observations without explicit testing for intra-study correlation (Wolf 1986).

In the field of environmental valuation, meta-analyses have focused on a range of environmental issues from outdoor recreation to urban air pollution, based on single or multiple valuation techniques. The increase in meta-analytical research seems to be triggered principally by (1) increases in the available number of environmental valuation studies, (2) the seemingly large differences in valuation outcomes as a result of the use of different research designs (Carson et al. 1996), and (3) the high costs of carrying out environmental valuation studies which tend to increase policymaker demand for transferable valuation results.

Meta-analysis enables researchers to identify criteria for valid environmental value transfer[1] or to test the convergent validity of value estimates. In the first case the data-set is entirely used to determine the factors that help to significantly explain variances in valuation outcomes. In the second case the data-set can be split, for example into two parts, one of which is used for the first purpose and the other to test whether the value estimates based on the significant factors fall within the confidence interval of the other half's estimates.

Environmental value transfer is commonly defined as the transposition of monetary environmental values estimated at one site (study site) through market-based or non-market-based economic valuation techniques to another site (policy site). The most important reason for using previous research results in new policy contexts is cost effectiveness. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time-consuming original research to quickly inform decision making.

The criteria for selecting studies for environmental value transfer suggested in the literature focus on the environmental goods involved, the sites in which the goods are found, the stakeholders and the study quality (Desvousges et al. 1992). However, very little published evidence exists of studies that test the validity of environmental value transfer. Moreover, in the few studies that have been carried out, the transfer errors are substantial (Brouwer 1998).

As more information about factors influencing environmental valuation outcomes becomes available, for instance through the meta-analysis presented here, transfers across populations and sites become more practicable.

## Dataset and study characteristics

The list of wetland CV studies included in the meta-analysis is presented in Table 1. Most studies have been published in journals. Half of all studies were carried out between 1985 and 1989, with most being published in the first 3 years of the 1990s. One study was carried out in the 1970s, 19 in the 1980s and 10 in the 1990s. Besides the inclusion of published significant results, Table 1 illustrates two other problems in this

[1] The term 'environmental value transfer' is used here instead of the popular term 'benefits transfer', because CV can also measure the benefits foregone, which makes the estimated values costs instead of benefits. WTP is the conventional economic approach to measure *environmental values* in money and hence make them commensurable with other market values – costs and benefits – associated with decisions that have been made, are made or have to be made in the face of limited human and natural resources

**Table 1**
Studies (with year of publication in parentheses) included in the meta-analysis

| | Authors | Type of publication[a] | Study year | $n^b$ |
|---|---|---|---|---|
| 1 | Bateman et al. (1995) | Journal article (EE) | 1991 | 3 |
| 2 | Bergstrom et al. (1990) | Journal article (EE) | 1986 | 1 |
| 3 | Bishop and Boyle (1985) | Consultancy report | 1985 | 2 |
| 4 | Bishop et al. (1987) | Journal article (TAFS) | 1985 | 7 |
| 5 | Brouwer and Slangen (1998) | Journal article (ERAE) | 1994 | 3 |
| 6 | Carson and Mitchell (1993) | Journal article (WRR) | 1983 | 3 |
| 7 | Cooper and Loomis (1991) | Book chapter | 1987 | 3 |
| 8 | Cummings et al. (1994) | Journal article (AJAE) | 1992 | 2 |
| 9 | Desvousges et al. (1987) | Journal article (JEEM) | 1981 | 21 |
| 10 | Farber (1988) | Journal article (JEM) | 1984 | 1 |
| 11 | Garrod and Willis (1996) | Journal article (JEPM) | 1993 | 4 |
| 12 | Green and Tunstall (1991) | Journal article (AE) | 1986 | 1 |
| 13 | Greenley et al. (1981) | Journal article (QJE) | 1976 | 4 |
| 14 | Silvander (1991) | Dissertation | 1989 | 2 |
| 15 | Jordan and Elnagheeb (1993) | Journal article (WRR) | 1991 | 2 |
| 16 | Kaoru (1993) | Journal article (ERE) | 1989 | 1 |
| 17 | Kosz (1996) | Journal article (EE) | 1993 | 1 |
| 18 | Lant and Roberts (1990) | Journal article (EPA) | 1987 | 6 |
| 19 | Loomis et al. (1991) | Book chapter | 1989 | 10 |
| 20 | Loomis (1987) | Journal article (WRR) | 1985 | 1 |
| 21 | Olsen et al. (1991) | Journal article (Rivers) | 1989 | 3 |
| 22 | Phillips et al. (1993) | Journal article (CJAE) | 1991 | 2 |
| 23 | Sanders et al. (1990) | Journal article (WRR) | 1983 | 2 |
| 24 | Schultz and Lindsay (1990) | Journal article (WRR) | 1988 | 1 |
| 25 | Spaninks (1993) | MSc thesis | 1993 | 3 |
| 26 | Spaninks et al. (1996) | Scientific report | 1995 | 2 |
| 27 | Sutherland and Walsh (1985) | Journal article (LE) | 1981 | 2 |
| 28 | Whitehead and Blomquist (1991) | Journal article (WRR) | 1989 | 6 |
| 29 | Willis (1990) | Journal article (AE) | 1986 | 2 |
| 30 | Willis et al. (1995) | Journal article (JEM) | 1992 | 2 |

[a] Abbreviations: *AE* Applied Economics; *AJAE* American Journal of Agricultural Economics; *CJAE* Canadian Journal of Agricultural Economics; *EE* Ecological Economics; *EPA* Environment and Planning A, *ERAE* European Review of Agricultural Economics; *ERE* Environmental and Resource Economics; *JEEM* Journal of Environmental Economics and Management; *JEM* Journal of Environmental Management; *JEPM* Journal of Environmental Planning and Management; *LE* Land Economics; *QJE* Quarterly Journal of Economics; *TAFS* Transactions of the American Fisheries Society; *WRR* Water Resources Research
[b] Number of observations taken from each study

meta-analysis. First, a number of people have been involved in several studies and related publications. This may result in an '*authorship*' effect. Learning from previous studies, authors may use similar, perhaps slightly adapted survey designs in subsequent studies. Secondly, 103 data points (observations) were extracted from 30 studies. This corresponds, on average, to three or four observations per study. More than half of all studies provided one or two observations. Outliers are the studies by Loomis et al. (1991) and Desvousges et al. (1987), providing 10 and 21 observations respectively. Studies provided more than one observation mainly because of the use of split survey samples targeting different wetland user and non-user groups and testing different survey designs. The possibility that results from the same study cluster together, for example as a result of identical survey design or sample population, and that results from some studies may be more variable than others was tested and accounted for in the meta-analysis (see 'The model').

The studies included in the analysis focus primarily on wetlands or wetland-type areas.[2] The specific WTP questions addressed in each study cover a large continuum of activities, actions or projects related to wetlands, but in some cases (approximately a third of all studies) also to water resources in general. These values were kept in the analysis because they referred directly to the hydrological wetland functions distinguished in the analysis and were considered reliable estimates for these functions. The WTP questions range from outdoor recreational activities like birdwatching or fishing, to

[2] Although there is little agreement among scientists on what constitutes a wetland, a workable definition is given by the so-called Ramsar Convention (1975, article 1): '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 6 m'

ground-water protection and complete wildlife habitat preservation. Two-thirds of the studies are carried out in the USA, the rest in Europe. Half of the European studies were carried out in the UK.
Separating the heterogeneous complex of hydrological, biogeochemical and ecological functions performed by the wetlands considered in each study in the meta-analysis was very difficult. Based on the various functions addressed in the reviewed studies, a simple distinction was made therefore first of all between four main wetland ecosystem functions: flood control, water generation, water quality support and wildlife habitat provision (Fig. 1). Secondly, the main function valued in each study was assigned to one of each of these four groups. Hence, each study was categorised as addressing one of these four main wetland functions, unless a study explicitly generated distinct values for different wetland functions (for more details, see Brouwer et al. 1997).
Obviously, wetland ecosystem structures and processes and the functions they provide are highly interrelated, making it very hard, and in some cases impossible, to distinguish between individual functions. They often go hand in hand and attempts to separate them, for example for economic valuation purposes in order to avoid double counting, are liable to be arbitrary. This implies that double counting is a real problem and attempts to aggregate up to system-level values are fraught with difficulties. Also, in the case of the human benefits derived from the wetland functions involved, complete separation of direct and indirect use and non-use benefits is difficult. Only in a third of all studies could a single benefit flow be identified; in all other cases wetland functions provided multiple benefits.
Most studies (70%) asked respondents for the use and non-use values attached to the benefits derived from wetland functions. One study (Schultz and Lindsay 1990), elicited future use value only (option value), while another (Greenley et al. 1981) focused, among others, on the *ex ante* ‘option price’ of being able to make a better informed judgement in the future based on more information becoming available regarding water quality.
In eight studies an attempt was made to break down the stated total economic value *ex post* in the questionnaire into the various components distinguished in the literature, e.g. use, option, philantrophic, bequest, stewardship and existence value. In two water quality studies (Desvousges et al. 1987; Carson and Mitchell 1993), respondents were presented *ex ante* with a ‘value card’ which described the main reasons why water quality might be valued. In another two studies use and non-use values were elicited separately, by the use of either different questionnaires (Bishop et al. 1987) or separate questions for use and non-use values in the same questionnaire (Greenley et al. 1981).
Finally, two study quality indicators were included in the analysis: one for the quality of the studies included in the meta-analysis and one for the quality of the meta-analysis itself. The quality of individual studies is indicated by the study response rate and the quality of the meta-analysis by the so-called scope test. Both indicators are found back in the National Oceanic and Atmospheric

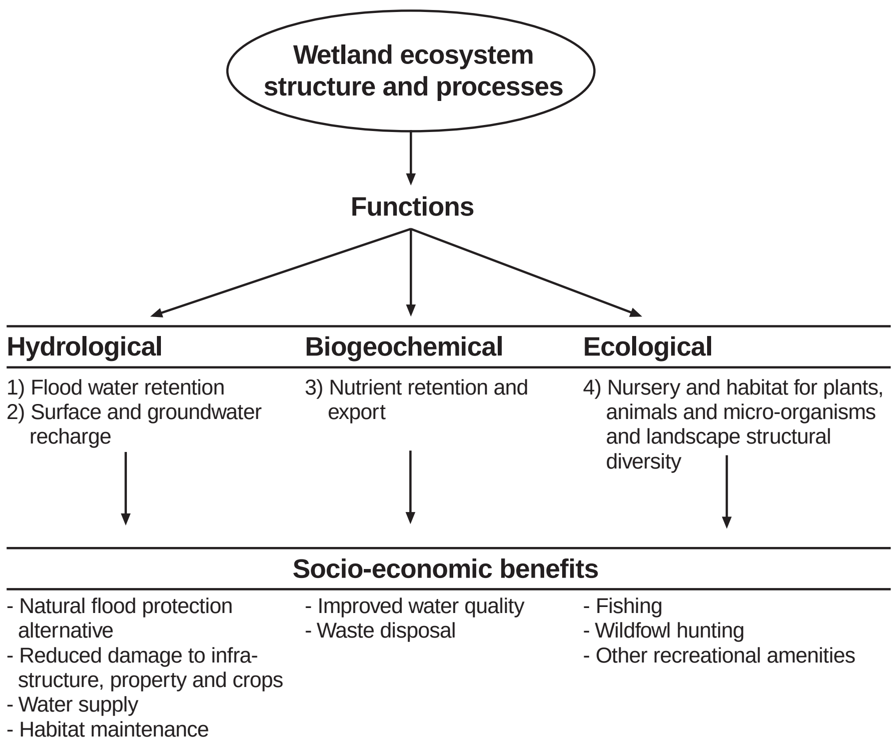


**Fig. 1**
Main wetland ecosystem functions identified in the meta-analysis and their derived socio-economic benefits. The analysis is based on stated WTP (willingness to pay) for goods and services for which no market exists. The value of marketed products such as reed or fish is excluded from the analysis to avoid double counting with the stated use and non-use values

Administration's (NOAA's) 'burden of proof' requirements (Arrow et al. 1993).[3]

A high non-response, either to the entire survey instrument or to the valuation question, raises concern regarding the study's representativeness, and questions the validity of the survey design employed and the extent to which the valuation scenario in the questionnaire was comprehensible and credible. The scope test refers to the sensitivity of WTP measures to changes in the provision level of the goods and services being valued, i.e. the difference between reference and target provision levels. Conforming to the strong monotonicity assumption in neo-classical consumer theory, responses should reveal a smaller WTP for smaller amounts of an environmental commodity provided by an environmental programme.

Very few studies reported the extent of protest bids and other questionable responses in the survey. Although most studies mention the survey response rates, it is in many cases not clear what these response rates actually represent or which criteria have been used to exclude responses from further analysis. Where such information was available, protest bids and questionable responses were excluded from the response rates.

In order to carry out a scope test, the size of the affected study site and the difference between the reference and the target levels of environmental service provision in the CV scenarios should ideally be considered. However, in two-thirds of all studies no information is provided about the size of the area involved. In about one-third of all studies, the study site size was estimated using geographical maps. Problems accumulate when aiming to also include the difference between the reference and target levels of the various wetland functions distinguished in the environmental scenarios in each study. The multi-dimensional nature of these functions makes a comparison between studies impossible. Hence, instead a 'relative size' variable was compiled, referring to the share of each study site in the country's total stock of wetlands.

## The model

The structure of the data used in the meta-analysis is complex. WTP values are generated by different studies, carried out in different geographical locations using different valuation formats. Using the summary statistics of these different studies in a pooled sample, the usual conditions required for ordinary least squares (OLS) regression are likely to be violated. In order to account for heteroscedasticity, a generalised least squares (GLS) regression technique called multilevel modelling was used (Langford 1994; Langford et al. 1998). [The GLS regression was carried out using the package MLN (Rasbash and Woodhouse 1995).]

Given the shape of the distribution of the WTP amounts, a logarithmic transformation was used. Hence, for the log(WTP) amounts of the studies Y, the GLS model is:

$$\mathrm{Y} = X\beta + \mathrm{Z}\Theta \qquad (1)$$

where $X\beta$ consists of the design matrix $X$ and associated parameters $\beta$ represent the mean or fixed effects of the explanatory variables on the dependent variable $Y$. However, whereas in OLS regression there is a single vector of error terms or residuals, here a more complex variance structure may be modelled where the values of residuals are dependent on explanatory variables included in the design matrix $Z$ for the random part of the model. For example, using one explanatory variable for simplicity, Eq. (1) can be written as:

$$\mathrm{y_i} = \beta_0 + \beta_1 x_{1i} + u_i + v_{1i}, \quad \begin{bmatrix} u_i \\ v_i \end{bmatrix} \sim N\left(\begin{bmatrix} 0 \\ 0 \end{bmatrix}, \begin{bmatrix} \sigma_u^2 & \sigma_{uv} \\ \sigma_{uv} & \sigma_v^2 \end{bmatrix}\right) \qquad (2)$$

where $u_i$ is the residual associated with the intercept $\beta_0$, and $v_i$ is the residual associated with the slope parameter $\beta_1$ of $x_1$. While the variance of the responses in OLS is determined by a single residual term, in the basic GLS model the variance is dependent on the explanatory variable:

$$\mathrm{var}(\mathrm{y_i}) = \sigma_\mathrm{u}^2 + 2\sigma_{\mathrm{u}v}\mathrm{x}_1 + \sigma_v^2\mathrm{x}_1^2 \qquad (3)$$

This can be done for any number of variables, hence making the variance of the responses a complex function of the explanatory variables, accounting for heteroscedasticity. This turns out to be highly relevant, as there are significant differences, for example, in the variance of responses within different studies (intra-study effects). Using the subscript $j$ to label different studies, the basic previous GLS model can be rewritten as:

$$\begin{aligned} \mathrm{y_{ij}} &= \beta_0 + \beta_1 x_{1ij} + u_{ij} + v_{1ij}x_{1ij} \\ &\quad + s_j, \quad \begin{bmatrix} u_i \\ v_i \end{bmatrix} \sim N\left(\begin{bmatrix} 0 \\ 0 \end{bmatrix}, \begin{bmatrix} \sigma_u^2 & \sigma_{uv} \\ \sigma_{uv} & \sigma_v^2 \end{bmatrix}\right), \\ s_j &= N(0, \sigma_s^2) \end{aligned} \qquad (4)$$

where $\sigma_s^2$ is the variance parameter that describes the differing variability of estimates within different studies. This latter model will be referred to as the 'extended' model in the results section.

## Results

### Summary statistics

A first step in the meta-analysis was to make stated average WTP amounts in each study comparable. The response

[3] In 1992, the NOAA commissioned a prestigious 'Blue-Ribbon Panel' of economists and survey specialists, co-chaired by Nobel laureates Kenneth Arrow and Robert Solow, to investigate the CV method. After carefully considering a wide range of issues, the panel's report gave the method a qualified bill of health, but only if studies were conducted to a rigorous set of guidelines. The panel identified a subset of issues which it called 'burden of proof' requirements

variable is average WTP per household per year for the preservation of specific wetland aspects. After expressing WTP in national currencies in terms of their 1990 purchasing power, these national currencies were converted in the International Monetary Fund's (IMF's) Special Drawing Rights (SDRs), which is the Fund's official monetary unit of account (IMF 1996). Average WTP for wetland function preservation found in all studies taken together is 62 SDRs (by the end of 1995, 1 SDR approximately equalled 1.5 US$). The median is considerably lower, namely 34 SDRs.

The breakdown of WTP values according to a number of possible explanatory factors is presented in Table 2. Mean WTP values have been calculated for (1) wetland types as identified by Dugan (1990); (2) main wetland functions; (3) relative wetland size; (4) the different value types elicited in the studies (use and/or non-use values); (5) the continent where the wetland sites are found; (6) the way people were asked to pay for wetland function preservation in the CV survey as part of the institutional setting of the wetland conservation programmes (e.g. through general income taxation or otherwise); and (7) the way the WTP question was elicited in the CV survey (e.g. in an open-ended question or otherwise).

The calculated differences in mean WTP for each of these categories are statistically significant at the 5% significance level or stronger (see the outcomes of the Chi-square test statistic in the last column of Table 2). The range of values (minimum and maximum) found for the factor levels across studies is considerable. Mean WTP per household is more or less the same for salt- and fresh-water wetlands. However, the number of observations for salt-water wetlands is very low. Almost all observations refer to fresh-water wetlands. Within fresh-water wetlands, the value of wetlands fed by rivers (riverine) is twice as high as the value of lakes and ponds (lacustrine) or marshes and swamps (palustrine). Ground-water is valued highest, although the number of observations is again low.

The wetland function flood control generates the highest mean WTP, followed by wildlife habitat provision and landscape structural diversity (labelled biodiversity in Table 2). Surface and groundwater recharge (labelled water generation in Table 2) has the lowest value. As expected, larger sites result in higher WTP. An inconsistency is found between the categories 'small' and 'very small'.

Use values associated with wetland functions are almost twice as high as non-use values. However, a combination of the two is not equal to their sum, suggesting some non-linear relationship between the two. Socio-psychological and related factors underpinning so-called embedding effects, where the sum of the valuations placed on the parts of a commodity exceeds that for the whole (Bateman et al. 1997), may be one important reason.

North Americans are willing to pay, on average, more than Europeans. Since average income data for the survey samples are missing in most of the studies reviewed, the country in which the wetland sites are found and the CV surveys were carried out is used as an indicator for income differences to account for people's capacity to pay. Income taxation as a payment vehicle generates the highest average WTP value, followed by stated WTP over and above actual trip expenditures to visit a wetland site (use value). The latter is part of a set of payment mechanisms that present wetland functions to respondents as a private good, i.e. to be 'consumed' by the individual who is being interviewed by asking him or her to pay, for example, an entrance fee. On the other hand, general income taxation is expected to prompt responses that consider the implications of wetland preservation for society at large, not just for the individual (Mitchell and Carson 1989; Blamey 1995). Hence, the higher WTP value elicited through this payment mechanism is expected to reflect more than private use values only. (Obviously, 'users' of specific wetland functions, for instance people who visit a wetland site for recreational fishing or boating, may hold values related to their non-use, e.g. preservation for future generations, as well.) The high value for non-specified payment modes is due to outliers and the very low number of observations. Calculating through the value of wetland function preservation in existing product prices yields a significantly higher mean WTP than the establishment of a private fund or raising entrance fees.

Finally, corresponding to previous research results (e.g. McFadden 1994; Bateman et al. 1995; Willis et al. 1995), the open-ended (OE) elicitation format yields a significantly lower WTP than other formats. The dichotomous choice format (yes or no to a given bid amount) yields the highest average WTP, followed by the iterative bidding procedure (yes or no to a sequence of bid amounts). Possible explanations are the larger numbers of non-responses or protest responses OE elicitation tends to produce (Desvousges et al. 1983) or the uncertainty experienced in answering the unfamiliar WTP question for non-market goods and services in an OE format (Bateman et al. 1995).

### Regression results

The findings for the basic and extended GLS model in which we account for study level effects are presented in Table 3. Only those variables are included that are statistically significant at the 0.1 level. The fixed part of the model represents the fixed or mean effects of each variable, as for an OLS regression model, while the random part displays the variance and covariance parameters that model heteroscedasticity.

The estimates for the regression results are obtained through Maximum Likelihood techniques (e.g. Maddala 1983). The outcome of the likelihood ratio test ($\chi^2_{14} = 96.51$; $P < 0.01$) rejects the null hypothesis of zero effects for all explanatory variables. A pseudo $R$-squared was calculated from the log likelihood (LL) function. The outcome corresponds with the goodness of fit measures usually found in CV studies. Since the pseudo $R$-squared lacks the straightforward explained variance interpretation of $R$-squared in OLS regression (Hamilton 1993), it is used here as a rough indicator for the model's goodness-of-fit.

**Table 2**
Summary statistics (*WTP* willingness to pay; *SDR* Special Drawing Rights)

| | Mean WTP (SDRs) | Standard error | Min (SDRs) | Max (SDRs) | $n^a$ | $\chi^2$ ($p$<)[b] |
|---|---|---|---|---|---|---|
| Wetland type | | | | | | 15.2 (0.05) |
| *Saltwater* | 56.2 | 27.2 | 19 | 137 | 4 | |
| Marine | 22.7 | 3.7 | 19 | 26 | 2 | |
| Lagoonal | 136.6 | – | – | – | 1 | |
| Lake | 42.8 | – | – | – | 1 | |
| *Freshwater* | 58.9 | 6.1 | 1 | 267 | 97 | |
| Riverine | 71.7 | 13.7 | 1 | 267 | 38 | |
| Lacustrine | 36.8 | 9.4 | 12 | 88 | 9 | |
| Palustrine | 36.9 | 4.3 | 9 | 117 | 31 | |
| Groundwater | 125.7 | 24.3 | 99 | 174 | 3 | |
| *Fresh- and saltwater* | 237.5 | 106.2 | 131 | 344 | 2 | |
| Wetland function | | | | | | 7.8 (0.05) |
| Flood control | 92.6 | 24.4 | 24 | 177 | 5 | |
| Water generation | 21.5 | 6.8 | 3 | 59 | 9 | |
| Water quality | 52.5 | 5.9 | 9 | 174 | 43 | |
| Biodiversity | 76.1 | 12.8 | 1 | 344 | 46 | |
| Relative wetland size | | | | | | 13.1 (0.01) |
| Very large | 86.9 | 17.6 | 19 | 177 | 8 | |
| Large | 70.3 | 21.6 | 12 | 344 | 16 | |
| Medium | 67.0 | 8.9 | 3 | 267 | 58 | |
| Small | 29.5 | 13.2 | 1 | 137 | 13 | |
| Very small | 53.4 | 13.8 | 24 | 105 | 6 | |
| Value type | | | | | | 6.1 (0.05) |
| Use value | 68.1 | 8.4 | 9 | 344 | 50 | |
| Non-use value | 35.5 | 4.8 | 12 | 78 | 13 | |
| Use and non-use values | 63.8 | 12.9 | 1 | 267 | 40 | |
| Country | | | | | | −3.0 (0.003)[c] |
| USA and Canada | 70.8 | 7.8 | 3 | 344 | 80 | |
| Europe | 32.8 | 8.4 | 1 | 177 | 23 | |
| Payment mode | | | | | | 27.4 (0.001) |
| Income tax (1) | 121.3 | 18.1 | 2 | 267 | 22 | |
| Entrance fee/private fund (2) | 28.6 | 5.7 | 1 | 137 | 28 | |
| Product prices (3) | 47.8 | 8.9 | 3 | 174 | 22 | |
| Combination of (1) and (3) | 42.8 | 6.3 | 9 | 117 | 26 | |
| Trip expenditures | 102.9 | 6.8 | 89 | 112 | 3 | |
| Not specified | 237.5 | 106.2 | 131 | 344 | 2 | |
| Elicitation format | | | | | | 10.1 (0.01) |
| Open-ended | 37.4 | 6.5 | 1 | 137 | 35 | |
| Dichotomous choice | 91.2 | 17.1 | 3 | 344 | 29 | |
| Iterative bidding | 78.5 | 14.9 | 9 | 244 | 20 | |
| Payment card | 47.1 | 8.4 | 10 | 174 | 19 | |

[a] Number of observations does not sum up to 103 in all cases as a result of missing values
[b] Outcome of the non-parametric Kruskal–Wallis test statistic which has approximately a Chi-squared distribution under the null hypothesis of equal average WTP in all groups
[c] Outcome of the non-parametric Mann–Whitney test statistic for two independent samples which has approximately a standard normal distribution under the same null hypothesis

The estimated models account for approximately 37% of the observed variability in the mean WTP values found in individual studies.
For the fixed part of the basic and extended model, the estimated coefficients in the semi-log function represent the constant proportional rate of change in the dependent variable per unit change in the independent variables (Johnston 1984). Hence, the coefficient estimated for the dummy variable 'Payment vehicle' in the basic model reflects, *ceteris paribus*, an almost twice as higher average WTP for an increase in income tax than for any other payment vehicle. Compared with other elicitation formats, WTP is reduced, on average, by 41% (*ceteris paribus*) when using studies in a value transfer exercise which are based on an open-ended WTP question.
The basic model also indicates that study location has a significant impact on average WTP. The dummy variable has a value of 1 if the research took place in North America and zero if in Europe. As shown before, average WTP is substantially higher in North America than in Europe. The parameter estimates for the four main wetland functions are particularly interesting. These functions are found to have a statistically significant role in explaining variance in average WTP. The size of the

**Table 3**
Generalised least squares results for the basic and extended model. As a result of missing values for explanatory variables, number of observations is reduced from 103 to 92

| Parameter | Parameter definition | Basic model | | Extended model | |
|---|---|---|---|---|---|
| | | Estimate | Standard error | Estimate | Standard error |
| Fixed effects | | | | | |
| Constant | Intercept | 3.356*** | 0.100 | 3.311*** | 0.247 |
| Payment vehicle | Dummy: 1 = income tax; 0 = other | 1.880*** | 0.265 | 1.576*** | 0.362 |
| Elicitation format | Dummy: 1 = open-ended; 0 = other | −0.411** | 0.130 | −0.376* | 0.183 |
| Country | Dummy: 1 = North America; 0 = other | 1.861*** | 0.217 | 1.629*** | 0.363 |
| Response rate (1) | Dummy: 1 = 30–50%; 0 = other | −2.253*** | 0.326 | −1.722*** | 0.451 |
| Response rate (2) | Dummy: 1 = >50%; 0 = other | −1.904*** | 0.333 | −1.461** | 0.450 |
| Flood control | Dummy: 1 = flood control; 0 = other | 1.477*** | 0.240 | 1.134* | 0.456 |
| Water generation | Dummy: 1 = water generation; 0 = other | 0.691* | 0.342 | 0.441 | 0.479 |
| Water quality | Dummy: 1 = water quality; 0 = other | 0.545† | 0.282 | 0.659* | 0.327 |
| Random effects | | | | | |
| Between studies | | | | | |
| $\sigma^2_{\text{constant}}$ | Variance | – | – | 0.160* | 0.071 |
| Between average *WTP* | | | | | |
| $\sigma^2_{\text{constant}}$ | Variance | 0.059* | 0.029 | 0.045 | 0.028 |
| $\sigma_{\text{payment vehicle, constant}}$ | Covariance | 0.020 | 0.043 | 0.001 | 0.036 |
| $\sigma_{\text{country, constant}}$ | Covariance | 0.689** | 0.222 | 0.351** | 0.129 |
| $\sigma_{\text{country, payment vehicle}}$ | Covariance | −0.707** | 0.226 | −0.345** | 0.134 |
| $\sigma_{\text{flood control, constant}}$ | Covariance | −0.013 | 0.050 | 0.027 | 0.060 |
| $\sigma_{\text{water generation, constant}}$ | Covariance | −0.637** | 0.227 | −0.266† | 0.153 |
| $\sigma_{\text{water quality, constant}}$ | Covariance | −0.424† | 0.231 | −0.188 | 0.135 |
| $LL_{\text{unconstrained}}$ | | −83.907 | | −81.874 | |
| Pseudo *R*-squared | | 0.365 | | 0.380 | |
| *n* | | 92 | | 92 | |

† Significant at 0.10; * significant at 0.05; ** significant at 0.01; *** significant at 0.001

estimated parameters indicates that average WTP is, as before, highest for flood control, but this time, whilst controlling for other explanatory factors, followed by water generation and water quality and lowest for the wetland function biodiversity supply. The latter is used as the baseline category in the regression analysis in order to avoid multicolinearity. The positive parameter estimates for the three other wetland functions indicate that these functions generate higher values than the baseline function biodiversity supply.

This suggests the prominence of use over non-use motivations underpinning stated WTP amounts. The distinction between use and/or non-use values does not have a significant impact on average WTP, probably because the corresponding variance is already accounted for by the distinction between wetland functions. Also, relative wetland size is statistically not significant. Higher response rates, a rough indicator of better overall study quality, appear to result in significantly lower average WTP than low response rates. A practical explanation may be that low response rates are sometimes biased towards including a relatively large number of sample respondents with a greater interest than average in environmental protection and corresponding WTP.

The model's random effects can be used to (1) model heteroscedasticity and (2) investigate the suitability of using specific CV results in a value transfer exercise. This will be explained below. The interpretation of the random effects is as follows: the variance of the constant in the basic and extended model is the variance associated with the baseline case, i.e. where the value of all the explanatory variables is zero. The variance of, for example, North American studies in the basic model is $(0.059 + 2 \times 0.689) = 1.437$. Hence, North American studies are more variable than European ones. In this way heteroscedasticity can be modelled in the basic model. Another example is North American studies using income taxation as a payment vehicle:

$$\sigma^2_{\text{country, payment vehicle}} = 0.059 + 2 \times 0.689 + 2 \times 0.020 - 2 \times 0.707 = 0.023.$$

The last term in this equation is the covariance between payment vehicle and country. So, wetland CV studies based on income taxation in the US appear to have a particularly low variance.

Accounting for study-level effects in the basic model significantly reduces the sample variance or standard deviation of average WTP in the extended model ($\chi^2_1 = 4.06$; $P < 0.05$). The extended, multilevel model accounting for the random effects between studies hence provides a significant improvement over conventional meta-analysis by allowing for the hierarchical structure of data implicit in clustering of multiple results from single studies. As expected, having explained some of the variance in the model by study-level effects, the random effects between log(WTP) amounts have decreased, except for the covariance between flood control and the intercept. However, in the extended model the fixed effects have slightly decreased, while the significance level of half of the explanatory variables is lower. Only the significance of the wetland function water quality has increased. The wetland function water generation has become statistically insignificant and has switched place with the function water quality supply in the order of size, suggesting that the results must be viewed with some caution when properly accounting for intra-study variability.

If low variance is considered an estimate of quality in the sense that study results are better suited for comparison and hence can be more readily put together in a value transfer exercise, then it can be concluded that, on the whole, studies using income taxation as a payment vehicle are better suited than other payment vehicles, and that studies valuing wetland biodiversity tend to be less variable than studies valuing wetlands in their capacity of generating water or maintaining water quality.

## Discussion and conclusions

In this paper, estimates for socio-economic use and non-use values attached to different hydro-ecological and biogeochemical wetland functions were compared and synthesised in a meta-analysis of wetland CV studies. The meta-analysis provides insights into the factors that have to be considered when attempting to transfer environmental values on the basis of CV studies. A statistically significant breakdown of WTP values for four main wetland functions has been presented. Although single ecosystem characteristics or functions are given meaning and value within existing ecosystem structures, the distinction between functions is essential for a valid transfer of the economic values generated within an ecosystem's primary self-organising capacity. It reduces the risk of double counting when attempting to assess a natural resource's total economic value on the basis of different valuation studies.

From an anthropocentric point of view, the size of the estimated parameters in the estimated basic model is as expected. Average WTP is highest for flood control, because of the possible risks to life and livelihood as a result of flooding and the capacity of wetlands to reduce this risk, followed by water supply and water quality and finally the provision and maintenance of biodiversity. However, these results have to be handled with care for a number of reasons.

Ecosystem structures and processes provide a heterogeneous complex of highly interrelated socio-economic functions. The analysis presents a simple and arbitrary breakdown of these functions into independent components. The distinction between four main wetland functions does also not necessarily correspond with people's perception of the various functions wetlands perform, which depends upon their own knowledge and experience with the resource. It is people's perception of a good's characteristics or functions that influence their attitudes and behaviour, not necessarily the good's 'objective' characteristics (e.g. see Adamowicz et al. 1997). Finally, accounting for intra-study variability, the statistical analysis produces slightly different results regarding the significance and size of the effect of the main functions on the WTP values. The low number of observations also has to be taken into consideration.

On the other hand, the study progressed meta-analytical research in environmental economics by providing a statistical multilevel model which accounts for the clustering of results from the same studies, for example as a result of identical survey design or sample population, and owing to the fact that results from some studies may be more variable than others. In the GLS models used, the variance and covariance estimators not only enabled us to model heteroscedasticity, but also provided important background information for environmental value transfer.

Finally, although considerable effort has been put into specifying the characteristics of the environmental functions and correspondingly the environmental goods and services involved, other important aspects that may have helped to explain differences in valuation outcomes remain undefined. This is a common problem in meta-analyses in the field of environmental valuation as a result of insufficient and inadequate information provided in published valuation studies. Relevant information about the samples' socio-economic values is missing in many studies, let alone respondents' socio-psychological and cultural characteristics.

In meta-analysis, inferences are made on the basis of information on global statistics, such as the mean and standard deviations of parameter estimates. These may or may not describe individual behaviour adequately. In order to overcome this potential problem and to increase the study's validity and reliability, a logical next step would be to gather more information about sample population characteristics by complementing the analysis with the underlying individual responses. This will provide an important test of the appropriateness of meta-analysis as an instrument to synthesise CV outcomes for the purpose of value transfer.

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