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Discussion Paper 68-2013

CONSIDERING HOUSEHOLD SIZE IN CONTINGENT VALUATION STUDIES

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ISSN 1867-934X (Printausgabe)
ISSN 1868-0720 (Internetausgabe)

Considering Household Size in Contingent Valuation Studies

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Abstract:

In many empirical Contingent Valuation studies one finds that household size, i. e. the number auf household members, is negatively correlated with stated household willingness to pay for the realization of environmental projects. This observation is rather puzzling because in larger households more people can benefit from an environmental improvement than in small households. Therefore, the overall benefit should be greater for larger households. A plausible explanation could be that household budgets are tighter for large families than for smaller families with the same overall family income. The fact that larger families can afford only smaller willingness to pay statements in Contingent Valuation surveys than smaller families with the same income and the same preferences might have consequences for the allocation of public funds whenever the realization of an environmental project is made dependent on the outcome of a Contingent Valuation study. In this paper we show how the use of household equivalence scales for the assessment of environmental projects with the Contingent Valuation Method can serve to reduce the discrimination of members of large families.

JEL-Class.: D61, H43, Q51

1. Introduction

The Contingent Valuation method (CVM) is one of the most popular methods for the economic appraisal of environmental projects. It aims at the assessment of the change in social welfare generated by public projects in monetary terms in order to decide if a project is worthwhile from a social welfare point of view or not. If two or more alternative public projects are under discussion the CVM can help to decide which of these projects should be realized and which should be dropped.[1] Since the benefits accruing from different public projects affect different groups of the population differently the outcome of a valuation study has not only consequences for the efficiency of public spending but also for the distribution of the ensuing benefits. This paper deals with the equity aspect of environmental valuation.

The CVM is an interview-based direct valuation technique which aims at the assessment of people's Hicksian Compensating Variation (HCV) for a public project. The HCV is positive if an individual's utility increases as a consequence of the project in question, otherwise it is negative. For a utility increasing project it can be interpreted as a person's maximum

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1 Of course, there are other uses of environmental valuation studies, e. g. the assessment of damages to nature after environmental accidents [cf. Carson / Hanemann (2005, p. 827 ff.)] or the appraisal of the non-market production of the agricultural sector as an assessment basis subsidies in the EU [cf. Ahlheim / Frör (2003)].

willingness to pay (WTP) for the realization of this project. For a utility decreasing public project it can be interpreted as her willingness to accept compensation (WTA) for the utility loss she expects from the project. Much has been written about the theoretical deficiencies of the Contingent Valuation method [cf. e. g. Harrison (2007), Mathews et al. (2004), Carson / Hanemann (2005, p. 906 ff.), Bockstael / Freeman (2005), Diamond (1996a, b), Diamond / Hausman (1994) or Hausman (2012)]. But this has not impaired the dominant position of the CVM among environmental valuation techniques and it still holds that it "is hard to overestimate the central importance of contingent valuation to modern environmental economics" as Carson and Hanemann (2005, p. 826) put it, an opinion that is basically shared by Kling et al. (2012) and confirmed by Carson (2012).

While the theoretical concept of individual welfare measurement aims at the assessment of changes in the wellbeing of single persons, practical CVM surveys deviate from this path of virtue in two decisive ways: (1) instead of assessing individual welfare changes in terms of people's willingness to pay (for $HCV > 0$) or willingness to accept compensation (for $HCV < 0$) for a public project, in practical surveys typically the WTP or WTA of whole households instead of the single household members is assessed and (2) the amounts of willingness to pay or willingness to accept stated by the different households are aggregated over all households. This contradicts the rules of ordinal utility theory where a utility function is defined only up to a continuous and monotonically increasing transformation, so that no intensities of utility or of utility changes can be assessed and, therefore, no aggregation of utility levels or utility changes is allowed.

The household perspective adopted in practical CVM surveys raises two kinds of aggregation problems, an intra-household and a trans-household aggregation problem. The *intra*-household aggregation problem refers to the question of how the individual preference orderings and, accordingly, the WTPs of the different members of a household can be aggregated to obtain one common household preference ordering or one common household WTP. The *trans*-household aggregation problem, on the other hand, refers to the question of how the WTPs of different households should be aggregated to attain the overall "social WTP" for a public project or its social value. This trans-household aggregation problem is solved in most practical valuation studies simply by multiplying the mean WTP assessed from a representative household sample by the number of all households affected by the project in question. Ideally (i. e. if the selected household sample is really representative) this procedure is equivalent to adding up the HCVs over all households. As a decision criterion regarding the social desirability of a certain public project the Hicks (1939) - Kaldor (1939) criterion, which was originally defined for the sum of *individual* HCVs, is applied to the sum of *household* WTPs in practical cost-benefit analyses. In this context it postulates that a positive sum of household HCVs signals a potential Pareto improvement, so that the project in question can be recommended for realization. If the sum of all HCVs is negative the project should be rejected.[2] This shows that practical cost-benefit analysis based on the Hicks-Kaldor criterion violates the rules of ordinal utility theory by aggregating utility on two levels: the level of the people living in the same household, and the level of different households which are all affected by the same public project.

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[2] Since not only the CVM but also most of the other environmental valuation methods use this kind of household-based aggregation the resulting plausibility problems are, of course, not confined to the CVM only.

Our main concern here is, of course, not the fact that practical project appraisal techniques like the CVM contradict the pure doctrine of neoclassical welfare theory but, instead, we are worried that these techniques when applied in practical valuation studies often lead to counterintuitive results. When projects causing environmental improvements are valued one would expect that the WTP for such projects increases with household size since in many-person households more people will benefit from that project than in single-person households. Contrary to this hypothesis many empirical studies yield results where stated WTP decreases with household size [see e. g. Whitehead (1991), Whittington / Smith. (1992), McDaniels et al. (1992), Garrod and Willis (1994), Johannesson et al. (1996), Chambers et al. (1998), Roschewitz (1999), Hammit et al. (2001), Liu et al. (2003), Ahlheim et al. (2004), Dong et al. (2004), Aprahamian et al. (2007)]. Chambers et al. (1998, p.149) give the following explanation for their result of a negative influence of family size on WTP: "This result might be related to ability to pay; as family size increases, budgets tighten, and WTP falls". Our analysis here will be focussed on this aspect of trans-household aggregation since it has consequences for the distributional effects of public spending in the environmental sector.

If aggregation of household preferences for a public project is done by simply adding up the WTPs and WTAs of all households affected by this project and if the stated WTP of the members of large families as compared to the members of small families with the same household income and the same preferences of household members has to be smaller because the same budget constraint is more restrictive for large households than for small households this procedure implies a discrimination against the members of large households. It means that their "vote" in terms of stated WTP has less weight in the aggregation procedure than the vote of smaller households, all other things being equal. The fact that households with equal preferences and equal household income are treated differently in such valuation studies violates the principle of horizontal equity according to which equals have to be treated equally [cf. e. g. Slemrod / Yitzhaki (2002, p. 445)]. In order to reduce this household-size bias of Contingent Valuation results we suggest considering household size explicitly in valuation studies by using household equivalence scales for the aggregation of household WTPs.

The rest of the paper is organized as follows. In the second section we highlight the theoretical background of aggregation in environmental valuation analyses and show that WTP has to decrease with freely disposable income. This is the source of implausible results if household WTPs instead of individual WTPs are aggregated. In section 3 we discuss the theoretical concept of equivalence scales and their possible integration in environmental valuation studies. In section 4 we illustrate the practical application of equivalence scales empirically using a practical example of a Contingent Valuation study. Section 5 offers some concluding remarks.

2. Project appraisal

In this section we contrast the theoretical background of environmental valuation with the practice of project appraisal to show where the common aggregation procedure leads to a deviation from what theory demands.

2.1 Individual welfare measurement

The assessment of individual welfare effects of an environmental project aims at the identification of the individual utility changes caused by this project

(1) $$\Delta U_j = U_j^1 - U_j^0 = u_j(x_j^1, z^1) - u_j(x_j^0, z^0)$$

where the index j denotes individuals $j \in \{1, 2, ..., J\}$ and U_j^0 and U_j^1 denote the utility levels attained by an individual j before (situation 0) and after (situation 1) the project has been accomplished. The function $u_j(\cdot)$ is the individual's direct utility function, $x_j \in \Re^N$ is the vector of market commodities consumed by individual j and $z \in \Re^M$ is a vector of parameters describing the state of the environment, e. g. water or air quality, the number of different species in a certain area, the number of services provided by a certain ecosystem etc.

Since utility changes cannot be observed directly ΔU_j is typically measured by the Hicksian Compensating Variation (HCV). If we focus on the environmental effects of a public project and assume prices and income to be constant, the Hicksian Compensating Variation for an individual j can be described indirectly by the identity

(2) $$v_j(p, z^1, I_j - HCV_j) \equiv v_j(p, z^0, I_j) = U_j^0$$

where $v_j(\cdot)$ is the j^{th} individual's indirect utility function. Alternatively the Hicksian Compensating Variation can be defined directly by the identity

(3) $$HCV_j(p, z^1, U_j^0, U_j^1) \equiv e_j(p, z^1, U_j^1) - e_j(p, z^1, U_j^0)$$

where $e_j(\cdot)$ is the j^{th} individual's expenditure function. From the strict monotonicity of the expenditure function in utility U it follows that HCV is a reliable welfare indicator in the sense that it is positive for utility increases and negative for utility decreases. Within the world of ordinal utility the absolute value of the HCV is meaningless, it is only its sign that matters. In this pure interpretation an aggregation of the HCVs of different individuals does not make sense.

2.2 Aggregation

Nevertheless, in practical cost-benefit analyses the HCV is typically aggregated over all individuals affected by a project. For the political decision if this project should be carried out or not it is often not enough to assess the individual utility effects of that project according to (3), but it is also necessary or at least desired by politicians that these individual HCVs are aggregated. We know that an "objective", i. e. a non-normative aggregation of individual preferences is not possible under fairly plausible conditions. As a convention the Hicks-Kaldor criterion, according to which a public project should be accepted as socially beneficial if the sum of the individual Hicksian Compensating Variations according to

(4) $$\sum_{j=1}^{J} HCV_j = \sum_{j=1}^{J} \left(e_j(p, z^1, U_j^1) - e_j(p, z^1, U_j^0) \right)$$

is positive, is widely accepted in cost-benefit analysis. This kind of aggregation is compatible with the postulation of a utilitarian welfare function since for constant p and constant z the expenditure function is a (money-metric) utility function.

Of course, (4) does not represent some objective form of utility aggregation but corresponds to a specific (and in a way arbitrary) distributional norm. In the cardinal world of utility aggregation according to (4) a positive HCV (WTP) of one individual can be overcompensated by the negative HCV (WTA) of another individual. Further, if it is to be decided which of two alternative projects in two different regions should be realized it might be decisive in which region people with higher incomes live because higher incomes lead to higher WTPs and, therewith, to a higher overall social benefit in this region according to the aggregation mode (4).

The reason for this (mostly unwanted) distributional effect of government spending is that the HCV increases with income. This can be seen if we substitute the household's indirect utility function $v_j\left(p,z,I_j\right)$ for the utility levels U_j in (3) so that we obtain the "indirect" version of the Hicksian CV as a function of p, z and I according to

(5) $$HCV_j^{indir}(p,z^0,z^1,I_j) \equiv e_j\left(p,z^1,v_j\left(p,z^1,I_j\right)\right) - e_j\left(p,z^1,v_j\left(p,z^0,I_j\right)\right).$$

Taking into account the duality identity $e_j\left(p,z,v_j\left(p,z,I_j\right)\right) \equiv I_j$ we obtain the first-order derivative of (5) with respect to income as

(6) $$\frac{\partial\, HCV_j^{indir}}{\partial\, I_j}(p,z^0,z^1,I_j) \equiv 1 - \frac{\partial e_j\left(p,z^1,U_j^0\right)}{\partial U_j}\cdot\frac{\partial v_j\left(p,z^0,I_j\right)}{\partial\, I_j}\ .$$

From $e_j\left(p,z,v_j\left(p,z,I_j\right)\right) \equiv I_j$ it follows that the derivative of the expenditure function with respect to utility at some point [p, z, v(p, z, I)] is reciprocal to the derivative of the indirect utility function w. r. t. income, so that (6) can be expressed as

(7) $$\frac{\partial\, HCV_j^{indir}}{\partial\, I_j}(p,z^0,z^1,I_j) \equiv 1 - \frac{\dfrac{\partial e_j}{\partial U_j}\left(p,z^1,U_j^0\right)}{\dfrac{\partial e_j}{\partial U_j}\left(p,z^0,U_j^0\right)} > 0\ .$$

Since the expenditure function is strictly monotonically decreasing in environmental quality, for an environmental improvement it holds that under the assumption of a decreasing marginal utility of income its reciprocal, i. e. the "marginal cost of utility" $\partial e_j/\partial U_j$, must be greater for $e_j\left(p,z^0,U_j^0\right)$ than for $e_j\left(p,z^1,U_j^0\right)$. From (7) it follows then that under the conventional assumption of a decreasing marginal utility of income the Hicksian Compensating Variation increases with disposable income.

In an ordinal world where only the sign and not the absolute value of the HCV is considered this relation between HCV and income does not matter, but as soon as we start to aggregate the individual HCVs according to (4), the interests of persons with higher disposable incomes are more effectively represented in cost-benefit analyses than the interests of individuals with low incomes and the same preference ordering.

2.3 Project appraisal on a household basis

The Contingent Valuation method is an interview-based valuation technique. In the centre of a CVM interview is the elicitation question where respondents' WTP for the project is assessed. In a typical CVM survey a representative random sample of all households potentially affected by a public project is drawn for the interviews. Then a randomly chosen member of each of the selected households is interviewed and she or he is asked (among other things) the whole household's WTP for the project in question. This implies that the final decision if a certain environmental project should be implemented or not, is made after a two-stage aggregation process. On the first stage the individual preferences of the members of a single household are aggregated (at least implicitly) by the household member who is asked the household's WTP in a CVM interview, and then on the second stage the WTPs of all households are aggregated by the researchers in order to obtain a social WTP for the public project in question.

This procedure leads to two different aggregation problems arising in the context of a Contingent Valuation study. One problem is the problem of intra-household utility aggregation mentioned above. This refers to the question of how the preferences of different household members should be aggregated so that a single WTP for a public project can be elicited. Munro (2009, p. 5) or Lindhjem and Navrud (2009, p. 11) find that most papers reporting the results of empirical CVM studies are rather hazy regarding the kind of elicitation question that should be asked: should respondents answer the WTP question as individuals or as households? In most studies, however, it is at least implicitly assumed that the randomly chosen household member who is interviewed should state the WTP of the whole household for the environmental project in question. The theoretical fundament of this practice is the unitary household model according to which a household can be treated as if there were a single agent in each household maximizing a single (unitary) household utility function. This model presupposes income pooling which implies that for a household's consumption decisions only the amount of aggregate household income matters while the source of income is irrelevant for the household decisions. In this case it does not matter which household member is interviewed in a CVM survey because the stated WTP for a public project will always be the same. Ion Strand (2007) shows in a theoretical household bargaining model that the WTP for a public good stated by an arbitrary household member on behalf of the whole household is the same as the sum of the individual WTPs of all household members if the marginal valuation of that public good is the same for all household members. Ebert and Moyes (2009) analyse household decision making also on the basis of a game-theoretical household model and show that the outcome of an intra-household decision process depends decisively on the degree of cooperation between the different household members.

Empirical evidence does not support the unitary household model. Empirical studies show that in practice individual consumption or valuation decisions made separately by household members on behalf of the whole household depend on the person who makes the decision. Bateman and Munro (2005) find in an experimental study with couples that risky choices for the couple have different outcomes depending on who makes the choice: the husband, the wife or both together. This result is confirmed by another empirical study by Bateman and Munro (2009) where they find in a series of choice experiments that the health risks accruing from pesticides and fat in food are valued differently depending on which household member is asked. In a split-sample CVM study Lindhjelm and Navrud (2009) ask individual household members their valuation of a biodiversity preservation project in Norway from their own individual perspective and from the household perspective. They find that individual valuation is not much different from household valuation if the respective answers are compared between the two samples. But they also find that in the same sample stated household WTP is much higher than individual WTP if respondents are first asked their individual WTP and then their WTP on behalf of the whole household.

These empirical results show that the common practice of aggregating household WTPs instead of individual WTPs leads to biased results and needs some additional considerations: firstly, we cannot be sure that respondents in CVM interviews aggregate the preferences of the different household members correctly and, secondly, even if the respondent aggregates the WTP of the individual household members correctly, their WTP is restricted by the common household budget constraint that is the tighter the more people live in the household, other things being equal.

2.4 Distributional consequences

In practical cost-benefit analyses in the environmental sector only the benefits accruing from a public project are assessed in household interviews since households typically benefit from environmental projects, while they do not see the project costs. Therefore project costs, which are typically based on market prices (like the cost of capital, materials and labour), are assessed separately from the benefits which are measured in terms of households' willingness to pay for the realization of such a project.

Practical project assessment studies, therefore, follow the decision rule that a public project should be realized if the benefits as measured by the sum of the affected households' willingness to pay for the project WTP_h exceed the project costs C:

$$(8) \quad \left(\sum_{h=1}^{H} WTP_h - C \right) \begin{Bmatrix} > 0 & \Rightarrow \text{accept} \\ = 0 & \Rightarrow \text{either accept or reject} \\ < 0 & \Rightarrow \text{reject} \end{Bmatrix} \text{project for implementation}$$

where h is the index denoting households h = 1, 2, …, H. As mentioned before empirical studies like e. g. Whitehead (1991), Whittington / Smith (1992), McDaniels et al. (1992), Garrod and Willis (1994), Johannesson et al. (1996), Chambers et al. (1998), Roschewitz (1999), Hammit et al. (2001), Liu et al. (2003), Ahlheim et al. (2004), Dong et al. (2004), Aprahamian et al. (2007) show that in practice stated household WTP_h is often negatively correlated with household size. This implies that the chances for the realization of a project are the worse the more large-size households are among the supporters of the project. If the decision is between two alternative projects the project with a higher number of small-size households (other things being equal) stands a better chance of being selected for implementation even if a smaller number of people might be concerned by this project.

Therefore, with the implementation rule (8) which is typically used in environmental valuation studies large households have more problems to get projects realized that are in their interest than small-size households with the same income and preferences.

Since one cannot reasonably expect that the aggregate appreciation of an environmental project is lower in many-person households than in households with less members the only plausible explanation for this negative correlation is that household WTP of large-size households is more restricted by the household budget than the WTP of small-size households other things being equal. This would not pose a problem from a theoretical as well as a political point of view if everybody could freely choose in which kind of household he wants to live and if he could correct this choice at any time. Then the choice of the household a person lives in (and its size) could be regarded as reflecting her preferences like any other consumption decision does. In reality this is typically not the case, not for the children and often also not for the adults. Therefore, family size for many people is a kind of "fate" and this fate decides on the effectiveness with which people's preferences for some public good are represented and considered in a household-based valuation study.

The question whether this constitutes a problem or not, is a political as well as an ethical question. Environmental valuation studies provide the scientific basis for political decisions regarding the allocation of public funds. Since these allocation decisions are also decisions with respect to the distribution of the benefits accruing from these public funds questions of equity and distributional justice cannot be neglected here. For consistency as well as distributional reasons we suggest including household equity considerations explicitly in decisions on the realization of public projects, especially in the environmental sector. Since environmental decisions are oriented towards the future and the wellbeing of future generations, especially in this context a discrimination of families with many children seems to be rather unfortunate. Therefore we suggest using a weighted sum of household WTPs instead of (8) for the cost-benefit comparison, where household equivalence scales serve as welfare weights. Such a weighted aggregation of household WTP would lead to a more equal treatment of different households with different household size.

3. Enhancing the validity of CVM studies by using equivalence scales

Equivalence scales are well-known from the literature on poverty lines and the assessment of welfare payments [cf. e.g. Takeda (2010), Dagum / Ferrari (2004), Browning (1992)]. Equivalence scales are also considered in the context of taxation because they are regarded suitable to meet the requirements of horizontal equity there [cf. e. g. Lambert (2004)]. Horizontal equity in taxation refers to "…the idea that equals should be treated equally by the tax system, or that tax liability should not depend on any of a set of irrelevant characteristics" [Slemrod / Yitzhaki (2002, p. 445)]. Lambert (2004, p. 76) states: "It has become conventional to apply an equivalence scale to determine the equals at the family level". This is exactly what is needed for an adequate aggregation of household WTPs: identifying "equals" and giving them equal opportunities to feed their preferences for a public project into the decision rule (8). "Irrelevant characteristics" [Slemrod / Yitzhaki (2002, p. 445)] like the size of the family they happen to be born into should not be an obstacle to their influence on that decision. Equity in the context of environmental valuation, therefore, refers to people's influence on the decision if a specific environmental project should be realized or not, or which of several alternative projects should be carried out. Our suggestion is to give households with equal preferences and equal household income but different size the opportunity to state equal WTPs for a specific environmental project. For this purpose the households' stated WTPs should be weighted by suitable equivalence scales in order to make their stated WTPs compatible with each other. This aggregation procedure would lead to a modified decision rule

$$(9) \qquad \left(\sum_{h=1}^{H} S_h \cdot WTP_h - C \right) \begin{cases} > 0 & \Rightarrow \text{accept} \\ = 0 & \Rightarrow \text{either accept or reject} \\ < 0 & \Rightarrow \text{reject} \end{cases}$$

where S_h is the equivalence scale of a household h.

Studies on the different "welfare potential" of the same income for different households with different socioeconomic characteristics have a long tradition going back as far as the end of the nineteenth century to the work of Engel (1883 and 1895) whose equivalence scale concept serves as a reference for more modern concepts even today. Engel used the food-expenditure shares of different household groups as welfare weights in order to make households of different size and composition comparable with each other. While Engel's equivalence scales follow mere statistical concept later approaches to the equivalence scale problem were more sophisticated and typically based on neoclassical household theory. Especially, the papers of Prais and Houthakker (1955), Barten (1964), Kapteyn and van Praag (1976), Lewbel (1989), Blundell and Lewbel (1991), Muellbauer (1974, 1977, 1980) set further landmarks in this field of research. Here the question what incomes would be needed to achieve a certain level of household utility with different household sizes and compositions stands in the center of interest. Several concepts of equivalence scales have emerged over the years [for an overview see e. g. Dagum / Ferrari (2004)]. The extensive literature on the empirical measurement of equivalence scales covers a variety of approaches (cf. Ray 1986). Well-known concepts stem from, among others, van Praag (1968 and 1991), Kapteyn (1994) or Steward (2009). For equivalence scales based on Russian data see Takeda (2010), for Dutch data see Melenberg / van Soest (1995) and for an application to German data Charlier (2002).

The general idea which is common to the various concepts of household equivalence scales is that the material needs of a household depend, among other things, on its demographic characteristics, especially on its size and composition. Households with different demographic characteristics need different amounts of income to attain a given utility level or standard of living, even if preferences are equal. Analogously, the same income generates different degrees of satisfaction for households with different demographic characteristics and equal preferences. A household equivalence scale is "... a budget deflator which reflects household needs", as Muellbauer (1980, p. 154) puts it. Or, more precisely: "Equivalence scales are to welfare comparisons across households with different characteristics what cost of living indices are to welfare comparisons for a given household facing different prices" (Muellbauer, 1980, p. 155).

In analogy to the individual expenditure function used in (3) we may define a household conditional expenditure function [see e. g. Pollak / Wales (1979), p. 217] as

(10) $$e(p, z, W, \delta) \equiv \min p \cdot x \quad , \quad \text{s. t. } w(x, z, \delta) \geq W$$

where W is the household utility level defined by the (unitary) household utility function $W = w(x, z, \delta)$ $\left(\equiv \tilde{w}\left(u_1(x_1, z), u_2(x_2, z), ..., u_F(x_F, z), \delta\right)\right)$ with $x = \sum_{f=1}^{F} x_f$. The number of family members is F and δ is a vector of demographic household parameters like the number of adults, number of children etc. Since this unitary kind of household model is assumed in most practical valuation studies at least implicitly and since we do not want to interfere with the discussion on intra-household aggregation in this paper we will build our further argumentation here on this model.

From (10) we can derive the definition of conditional equivalence scales.[3] A standard or reference household with two adults and no kids is defined and all other households are regarded in relation to this reference household. Then an equivalence scale S_h for a demographic household group h expresses the ratio between the minimum expenditure a household from this group has to make in order to realize a certain level of satisfaction on the one hand and the respective expenditures of the reference household on the other [cf. also Deaton / Muellbauer (1980, p. 205) or Takeda (2010, p. 352)):

(11) $$S_h = S(p, z, W, \delta^h, \delta^r) \equiv \frac{e(p, z, W, \delta^h)}{e(p, z, W, \delta^r)} \qquad \left(h \in \{1, 2, ..., H\}\right)$$

where the functional forms of the expenditure functions for household h and for the reference household r are the same since, typically, all households are assumed to have the same preference ordering [4] and differ only in the demographic parameters δ. Obviously, for the reference household r it holds that $S_r = S(p, z, W, \delta^r, \delta^r) \equiv 1$.

Since the household utility level W cannot be assessed directly it is typically expressed by the indirect household utility function v(p,z,I) so that the equivalence scale S_h becomes

(12) $$S_h = \hat{S}\left(p, z, I, \delta^h, \delta^r\right) \equiv S\left(p, z, v\left(p, z, I\right), \delta^h, \delta^r\right) \quad , \quad \left(h \in \{1, 2, ..., H\}\right)$$

The dependence of the equivalence scale on prices and income is usually interpreted as a dependence on real income $\hat{I} = I/\hat{p}$ where $\hat{p}$ is some cost-of–living index.

Assuming that the WTP stated by the reference household r is a reliable indicator for its utility gain resulting from an environmental project (i. e. neglecting the intra-household aggregation problem here) we suggest correcting the WTP stated by other households with different

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3 As mentioned before there are many different definitions and concepts of equivalence scales in the literature. For an overview see e. g. Dagum / Ferrari (2004).

4 The implications of the assumption of identical preferences across all households have been extensively analyzed by Fisher (1987).

household sizes using an appropriate equivalence scale S_h in order to assess their "true" benefits B_h received from the project in question:

(13) $$B_h = S_h \cdot WTP_h \qquad \left(h \in \{1, 2, ..., H\}\right)$$

This up- or down-scaling of the WTP stated by a household h corresponds with endowing this household with a virtual income that would enable this household to state the same WTP as the reference household r [cf. Ahlheim (1998)]. Correcting stated WTP through equivalence scales which leads to scale-corrected household benefits B_h is a pragmatic way of avoiding the problems associated with the empirical assessment of virtual incomes.

The revised social value of some environmental project as needed in (9) would then be

(14) $$B^{social} = \sum_{h=1}^{H} B_h = \sum_{h=1}^{H} S_h \cdot WTP_h$$

instead of the unweighted sum of household WTPs according to (8). In practice, however, not every household potentially affected by a public project can be interviewed. As explained before, typically a representative random sample of households is chosen for the CVM interviews and the mean household WTP calculated from this sample is then extrapolated to the set of all households concerned [for the problems arising from this aggregation procedure see e. g. Bateman and Munro (2005, 2009)]. The weighted aggregation mode (14) has, therefore, to be applied to the different socio-demographic groups contained in the chosen household sample.

Since the questionnaire of a CVM survey typically contains questions with respect to the demographic characteristics of the households, it can be seen how many one-person, two-person, three-person etc. households are in a household sample. If we define K different demographic groups k (k = 1, 2, ..., K) in a sample of $\overline{H}$ households we can assign suitable equivalence scales S_k to each household group k, so that all households in a group k obtain the same equivalence scale S_k. If the number of households in a demographic group k is N_k, the mean WTP of this group is denoted by $\overline{WTP_k}$ and the number of all households potentially affected by a project is H we can calculate the "sample"-version of the social benefits accruing from this project as

(15) $$B^{social}_{sample} = \frac{H}{\overline{H}} \cdot \sum_{k=1}^{K} \overline{WTP_k} \cdot S_k \cdot N_k \quad .$$

This aggregation procedure means a break with the simplifying "a dollar is a dollar"-principle prevailing in traditional cost-benefit analysis and a step towards a more sophisticated valuation of environmental changes where the different demographic characteristics of households are considered explicitly and the discrimination of the members of large households is reduced. Scale correction of household WTP in environmental valuation does not mean the solution to all our problems, but it constitutes an important step in the right direction.

Correcting household WTP for household size directly by multiplying stated WTP by a suitable equivalence scale is, of course, only one possibility to deal with the problem of different household sizes. Another possibility would be to first correct household income using equivalence scales and then feed this corrected virtual income into the equation for the estimation of WTP as a function of various household data one of which being income.[5]

One problem when applying the equivalence scale approach in practice is the choice of the equivalence scale to be used in a concrete CVM study. Obviously, there is no unique "scientifically correct" way of computing equivalence scales S_k. Since aggregation of individual preference orderings is not possible on an objective, purely scientific basis all kinds of aggregation are normative and somehow arbitrary in the end. Ideally, an equivalence scale would take on the general form (11) but we know that it is difficult to assess empirically the values of the expenditure function needed for the computation of S_k according to (11). The main problem here is to assess a value for the household utility level W in (11). One class of studies relies on self-reported utility or wellbeing which is then used to estimate the equivalence scales econometrically based on household data (see e. g. Steward (2009), Takeda (2010) or Balli and Tiezzi (2010)). There are other, more pragmatic versions of equivalence scales proposed in the literature, some of which will be discussed in the next section. It is also clear that the absolute value of equivalence scales depends among other things on the choice of the reference household. For most equivalence scales a two-adult household is chosen as a reference household for which the respective equivalence scale is set equal to 1. Nevertheless, other choices might appear plausible as well.

4. The Effect of the use of equivalence scales on CVM results

In this section we demonstrate the effect of household equivalence scales on the results of practical CVM surveys in an empirical example (for details of the underlying valuation project cf. Ahlheim et al. (2004)).[6] We apply this scale correction procedure to stated household WTP in a contingent valuation study carried out in Eastern Germany (cf. Ahlheim et al. (2004)) and show how the results of the study respond to the use of different forms of equivalence scales. The aim of the study was to assess the social benefits accruing from the reclamation of a former open-pit mining area close to the city of Cottbus, which lies 120 kilometers south-east of Berlin. In this lignite pit near Cottbus mining activities will end by the year 2015. At this time the mining company will stop pumping off the groundwater so that it will rise to its original level and the former pit will be turned into a lake with a recreation area around it and also a small nature reserve.

For the assessment of the benefits accruing from this reclamation project a contingent valuation study was carried out in 2003. In this study more than 1,000 households were interviewed. Their willingness to pay for the realization of this rehabilitation project was

[5] This approach was followed by Carlsson et al. (2004, p. 156)

[6] This numerical example is taken from an unpublished discussion paper (Ahlheim / Lehr 2008). We are grateful to Ulrike Lehr who made the calculations cited in this section.

elicited using the double-bounded dichotomous choice question format.[7] The results are shown in Table 1. Based on a logit model, the average household willingness to pay for the Cottbus Lake turned out to be 4.39 Euro per month (with a 95% confidence interval between 3.45 and 5.23 Euro per month).

The city of Cottbus and the surrounding communities have roughly 100,000 people that live in 50,899 households. This yields an aggregate willingness to pay for the population affected by the rehabilitation project of ca. 223,000 Euro per month or 2.68 mill. Euro per year as shown in table 1.

Table 1: Average household WTP and aggregate WTP for the "Cottbus Lake"

	Logit Model
Avg. household WTP	4.39 Euro / month
Aggregate WTP	2.68 mill. Euro / year

Source: Ahlheim/Lehr (2008).

Table 2 shows the sign of the coefficients for some potential determinants of willingness to pay with household size being one of them (for the details of the underlying Contingent Valuation study s. Ahlheim et al. (2004)). While the education level and household income have a positive effect on stated willingness to pay, age and the distance from people's homes to the lake have a negative effect. As expected from our earlier discussion, household size has a negative effect on willingness to pay. Most of these effects are plausible, only the fact that household size should have a negative effect on households' willingness to pay for the proposed rehabilitation project does not make sense. In larger households typically more children are living than in smaller households, and it is especially the younger people who will be able to enjoy the benefits accruing from the new lake, since these benefits will be available only in the far future (as viewed from 2003 when the survey was conducted). Therefore large households should have a higher household willingness to pay for the rehabilitation project than small household because more household members will receive benefits from the project and part of them will also enjoy these benefits longer than the members of small households since they are children now and will live longer after the rehabilitation process will have been accomplished and the lake will be ready for utilization.

7 This rise in the cost of living was explained to the respondents as a consequence of the fact that the project would have to be financed by the communities in this area who in turn will raise their communal taxes, fees etc. These increases in costs will be passed on by shop owners or house owners to their customers or tenants giving rise to a general increase in prices and the cost of living in this area.

Table 2: The effects of some determinants on willingness to pay

	sign	p-value
Age	-	0.028
Monthly income	+	0.000
Education level	+	0.195
Household size	-	0.025
Distance to the lake (in travel time classes)	-	0.006
Gender (1=female, 0=male)	+	0.041
Constant	+	0.029
Bid	-	0.000

Source: Ahlheim/Lehr (2008).

In this basic version of the model the aggregate willingness to pay for the realization of the proposed reclamation project was calculated without consideration of household size or household composition. In a next step we shall analyze the effect of an explicit consideration of different versions of household equivalence scales on our results. The distribution of different family sizes in the research area is stated in table 3. It can be seen that in more than 36% of all households in the Cottbus area more than two persons were living at the time when the survey was conducted.

Table 3: Distribution of family size in Cottbus (2004, in percent)

Number of household members	1	2	3	4	5
	18.44%	46.25%	20.32%	12.33%	4.25%

Source: Ahlheim/Lehr (2008).

In our section 3 the concept of equivalence scales based on neoclassical household theory was discussed in detail. The empirical assessment of equivalence scales is still under debate among economists [cf. e.g. Schulte (2007), but also Steward (2009), Takeda (2010) or Balli and Tiezzi (2010)]. For practical applications the literature often suggests the use of expert scales or survey approaches [for the latter see Charlier (2002) or Schwarze (2003)]. Survey-based approaches which rely on slef-reportetd household satisfaction levels were already popular in the 1970s with the so-called Leyden School (cf. Kapteyn / van Praag 1976) but are still under discussion today (see e. g. Steward (2009) or Takeda (2010)).

Expert scales are the most widely used scales in practical applications, and the OECD scales are the most prominent examples (s. also Biewen (2000)). Table 4 gives an overview over the range of equivalence scales for Germany and compares them to the OECD scales. As can be seen from table 4 the amount of equivalence scales for a one-adult household is typically not just half the amount of the equivalence scales for a two-adult household but higher. This reflects the fact that two adults living together can realize economies of scale regarding their cost of living, so that with the same household income they can realize a higher standard of living than a one-adult household with half that income.

Table 4: Equivalence scales for Germany

Number of household members	1	3	4	5	6
Charlier	0.7	1.2	1.3	1.42	1.54
Missong	0.6	1.28	1.43	1.54	
OECD (modif.)	0.67	1.2	1.4	1.6	1.8
OECD (old)	0.59	1.29	1.59	1.88	2.18
Praag	0.83	1.12	1.21	1.29	1.35
Schröder	0.67	1.15	1.28	1.41	
Schwarze	0.79	1.15	1.26	1.37	
Social Assistance	0.56	1.36	1.72	2.08	2.44

A childless couple is the reference household with an equivalence scale of 1.00. Source: Schulte 2007

Table 4 does not account for the age of household members, especially of children, but we considered this in our calculations presented in table 5. In table 5 the equivalence scales from table 4 were used to correct the stated WTP of the different household groups for household size. It shows that the use of equivalence scales increased the total benefits from the reclamation project under consideration from 2.68 million Euro up to over 3 million Euro per year. This increase of the social value of the project by more than 300,000 Euro per year (or nearly 12%) as compared to the unscaled result is quite impressive. It gives us an indication of the considerable effect the use of equivalence scales could have on valuation studies carried out in developing countries where large families are far more common than in Germany.

These results show that the use of equivalence scales changed the original non-scaled results substantially. This was to be expected since in our research site many households with children and tight budgets could be found, like also in the rest of Eastern Germany at that time. Therefore, the adjustment of WTP to household size as shown in table 5 lead to considerable changes in the total social value of the proposed reclamation project. One can also see from table 5 that the choice of the equivalence scale has a considerable influence on the survey results. Unfortunately, there are no clear-cut rules for a suitable choice of equivalence scales for such studies which would make things easier. Therefore, also the explicit consideration of household size and composition in CVM studies leaves us with a rest of uncertainty which kind of adjustment of our survey results would lead us to the "correct" results. This has to be considered especially when interpreting CVM results or explaining their importance to politicians or government officials.

Table 5: Mean willingness to pay adjusted with equivalence scales in €

Scale	WTP/ average household	Total WTP
Charlier	4.58	2,800,614
Missong	4.61	2,816,776
OECD (modif.)	4.64	2,835,498
OECD (old)	4.80	2,934,793
Praag	4.55	2,777,772
Schröder	4.45	2,721,712
Schwarze	4.53	2,771,584
Social Assistance	4.94	3,020,152

Source: Ahlheim/Lehr (2008).

5. Concluding remarks

Our main concern in this paper is the fact that in standard cost-benefit analyses based on Contingent Valuation surveys a systematic discrimination of households with many household members can be observed. The reason for this "household size bias" of CVM results is that the absolute value of the Hicksian Compensating Variation (which is equivalent to the willingness to pay for utility-enhancing public projects) decreases as the freely disposable income of a household decreases. Since the political decision if a certain project should be realized or not depends on the sum of the WTPs of all households affected by this project, the influence of many-person households on this decision is smaller than the influence of small households with the same gross income and the same household preferences, since a smaller part of their income is freely disposable and their budget constraint is more "biting". Especially in the context of environmental projects this is very unfortunate since large households are typically large because they have many children who might enjoy improved environmental quality in the future. Their preferences are not adequately considered in the decision rule used in conventional cost-benefit analyses. Therefore, we recommend using household equivalence scales as welfare weights in the Hicks-Kaldor criterion to adjust Contingent Valuation results to household size and to protect especially the interests of children in the political decision process based on environmental cost-benefit analyses.

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