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The Assumption of Equal Marginal Utility of Income: How Much Does it Matter?

Abstract:
In most applied cost-benefit analyses, individual willingness to pay is aggregated without using explicit welfare weights. This can be justified by postulating a utilitarian social welfare function, along with the assumption of equal marginal utility of income for all individuals. However, since marginal utility is a cardinal concept, there is no generally accepted way to verify the plausibility of this latter assumption, nor its empirical importance. In this paper we use data from seven contingent valuation studies to illustrate that if one instead assumes equal marginal utility of the public good for all individuals, aggregate monetary benefit estimates change dramatically.

Keywords: Utility comparisons, environmental valuation, cost-benefit analysis, choice of numeraire.

JEL classification: D61, D62, D63, H41, Q2

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1. Introduction

Undergraduate students of economics usually spend considerable time and energy grappling with the concept of *ordinal* utility. This is not surprising given that ordinal utility is not comparable between persons and does not tell us anything about intensities being simply a tool for describing an individual's binary choices. As such it is a concept very much deprived of normative content [Sen, (1977)], although it is extremely useful for purely descriptive analyses.

When faced with the task of conducting an applied cost-benefit analysis after graduation, however, many economics need to go through the reverse process, since applied cost-benefit analysis requires a *cardinal*, interpersonally comparable utility concept [Arrow, (1951)]. After spending so much time getting accustomed to ordinal utility, the economist now has to grasp the implicit consequences of assuming, instead, that utility does indeed tell us something about intensities, and that one *can* compare benefits between persons.

The most common way to proceed in applied cost-benefit analysis is to implicitly or explicitly postulate a utilitarian social welfare function, so that social welfare is defined as the sum of all individuals' utilities. In addition, one needs a way to make utility cardinal and interpersonally comparable. The standard way of doing this is to assume that everybody's marginal utility of income is equal. If commenting upon this at all, most textbooks admit that this is questionable.[1] The problem is, what else can one do? There is still no generally accepted way to measure cardinal and interpersonally comparable utility, although some economists certainly have made attempts in that direction [see, e.g., van Praag (1991)]; hence, the very simplest assumption seems as good as any other.

Since the assumption of equal marginal utility of income does not rest on any empirical evidence, and is usually chosen more out of convenience than for any other reason, it introduces a certain kind of arbitrariness into the analysis. Unlike the choice of social welfare function, which can be discussed on ethical grounds, assumptions about the cardinality and comparability aspects of utility functions may be regarded as positive rather than normative; but the problem is that, in the absence of measurement methods, they cannot be empirically verified. One simply does not know whether the assumption of equal marginal utility of income is reasonable.

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[1] For a critical discussion, see, for example, Hammond (1990).

However, although we cannot directly test the plausibility of this assumption, we may still perform sensitivity analyses regarding the robustness of results to alternative ways of comparing cardinal utility between persons. This paper is an attempt to do precisely that. Based on data from seven contingent valuation studies of environmental changes, we calculate aggregate monetary benefits using an alternative operationalization of cardinal and interpersonally comparable utility; namely, that individuals may have different marginal utility of income, but are assumed to have an equal marginal utility of the environmental good in question.

This latter assumption corresponds to using units of the environmental good as the numeraire when aggregating individual benefits, instead of the usual approach of using money for this purpose. Until recently, it was a common belief that the choice of numeraire does not matter in cost-benefit analysis. However, Brekke (1997) demonstrated that the unit of measurement does indeed matter [see also Dréze (1998), and Johansson (1998)]. Brekke pointed out that, when it comes to public goods, different individuals generally have different marginal rates of substitution, since the amount of the public good is necessarily equal for all; implying that individuals have different marginal conversion rates between those goods that may alternatively be chosen as the numeraire. Consequently, when individual benefit estimates are aggregated, the interests of different individuals are given a different emphasis depending on which measurement unit is used.

Brekke’s result may be dismissed as irrelevant by some, arguing that it is not practicable to use environmental units: For example, one cannot in practice pay compensations in environmental units, and survey questions using environmental units may be very difficult for respondents to understand. However, we believe that the importance of Brekke’s result lies elsewhere: As already mentioned, using environmental units as the numeraire corresponds to an alternative operationalization of cardinal, interpersonally comparable utility; namely, that everybody has the same marginal utility of the environmental good. As such, it provides a means to check the empirical importance of the seemingly innocuous, but admittedly arbitrary assumption of equal marginal utility of income used in most applied cost-benefit analyses. Brekke presents one empirical example in his paper: Using data from a survey by Strand (1985), he found that the maximum per person cost that would make the project’s net benefits positive were *22 times higher* if money were used as a numeraire than if one used environmental units. In other words, exchanging the assumption of equal marginal utility of income for an assumption of equal marginal utility of the public good changed the result dramatically in this particular case. It is hard to see that the former assumption is more plausible than the latter from a theoretical point of view: Arguing in favour of one or the other requires reasoning about cardinality and interpersonal comparisons of utility, which is rarely found in economic theory.

If alternative and seemingly equally *a priori* plausible ways to operationalize interpersonally comparable cardinal utility yield dramatically different results, this should be of concern for all practitioners and users of cost-benefit analysis. In such a case, the traditional assumption cannot be defended by convenience alone, and one needs to take the issue of interpersonal comparisons of utility more seriously. We therefore believe it important to investigate whether Brekkes' finding of dramatically different aggregate benefit estimates holds more generally.

We should stress at the outset, however, that our aim is neither to argue that using environmental units is necessarily more relevant than using money as a numeraire, nor to identify the "best" way to aggregate individual welfare. Given that we do not know how to measure interpersonally comparable cardinal utility, we are simply examining the implications of replacing current practice with an alternative approach that is, in theory, equally valid. Our result is, in brief, that the two methods yield very different results.

2. Some central concepts

Below, we present a simple model explaining the main concepts which will be used in our calculations. The reader is referred to Brekke (1993, 1997) and Medin (1999) for further details.

Assume that there are *n* heterogeneous consumers with utility functions

$$U_i = u_i(Y_i, E) \tag{1}$$

where Y_i is individual *i*'s income, for all $i = \{1,...,n\}$. E is a pure public good, which we will think of as being provided by the environment. E will be measured in physical units, for example the estimated number of fish in a lake, or km^2 of wilderness. Utility is assumed to be increasing in income and the public good. Social welfare is given by the utilitarian welfare function

$$W = \sum_{i=1}^{n} U_i \tag{2}$$

This welfare function is chosen for simplicity, since our focus in the present paper is on utility measurement rather than the social objective function itself. Also, it corresponds to the most common practice in applied cost-benefit analysis.[2] We want to evaluate a project where the environmental good is increased by $dE > 0$, at a total cost ΣC_i, where C_i is the amount of money person i has to pay if the project is implemented. To avoid complicating matters unnecessarily, we will assume that $C_i = C$ for every $i = \{1,...,n\}$, i.e. every individual faces the same cost. Consequently, if the project is marginal, the project's effects on social welfare is

$$dW = \sum_{i=1}^{n} (-u'_{iY}\, C + u'_{iE}\, dE) \tag{3}$$

where u'_{ij} denotes the partial derivative of the utility function of person i for good j, $j = Y_i, E$.

We will take as a starting point that C and dE are known, but that u'_{ij}, which are cardinal properties of utility functions, are not directly observable. However, individuals' marginal rates of substitution u'_{iE} / u'_{iY} can in principle be observed by asking their willingness to pay for a one unit change in the environmental good. Here, we will abstract from all practical problems of actually eliciting true willingness to pay, and simply assume that it can be observed. Assume further that the project is marginal for all individuals, in the sense that any changes in individuals' marginal rates of substitution due to the project's implementation are small enough to be disregarded. A money measure of the project's net effect on i's utility, dU^Y_i, can be derived by differentiating (1) and dividing by u'_{iY}:

$$dU_i^Y = \frac{u'_{iE}}{u'_{iY}} dE - C \tag{4}$$

This is i's *net willingness to pay* for the project (i.e. her maximum willingness to pay to ensure the environmental change, minus the costs she has to pay, C).[3] However, this benefit measure can only be

[2] For a critical discussion of utilitarianism, see Sen and Williams (1982). Given that the social welfare function is kept unchanged throughout the analysis, and that the welfare function requires not only interpersonal comparison of utility but also cardinal utility, the results are relevant for any choice of social welfare function, although the actual formulae for aggregate welfare indicators would look different with another welfare function. Welfare functions of the minimax- or maximin-type (i.e. putting all emphasis on the interests of the worst off or the best off individual), require comparability, but (as long as the identity of the worst or best off individual is unchanged) not cardinality, in which case the problem we discuss does not arise.

[3] In this case, dU_i^Y is i's compensating variation. If we were to evaluate a project in which $dE < 0$, dU_i^Y would correspond to the equivalent variation. However, as long as the project is marginal, the difference does not matter under standard neoclassical assumptions. For a critical discussion of the latter with respect to such welfare measures, see Bateman et al. (1997a).

aggregated to yield a money measure of the project's total effect on social welfare if we know how to compare individuals' cardinal utilities. The usual way to do this is to assume that u'_{iY} is equal for all *i*. With this assumption, dW^Y, corresponding to the net benefit estimate from a standard unweighted cost-benefit analysis, is a monetary measure of the project' aggregate net welfare effect, such that the project is welfare-improving if $dW^Y > 0$:

$$dW^Y = \sum_{i=1}^{n} dU_i^Y = (\sum_{i=1}^{n} \frac{u'_{iE}}{u'_{iY}}) dE - nC \tag{5}$$

Alternatively, we could measure net individual utility changes in units of the environmental goods, dU^E. Differentiation of (1) and dividing by u'_{iE} yields

$$dU_i^E = -\frac{u'_{iY}}{u'_{iE}} C + dE \tag{6}$$

which tells us how large an increase in the environmental good individual *i* demands in order to be willing to pay *C*. We can call this benefit measure *i*'s *public good requirement.* Just as in the case of the money measure above, we must make sure that individual benefit estimates are comparable in order to aggregate them. If we assume that u'_{iE} is equal for everyone, we can simply add the individual public good requirements, yielding an aggregate benefit estimate dW^E. If this indicator is positive, the increase in the environmental good is large enough to justify the costs nC:

$$dW^E = \sum_{i=1}^{n} dU_i^E = ndE - (\sum_{i=1}^{n} \frac{u'_{iY}}{u'_{iE}}) C \tag{7}$$

While the *individual* net benefit estimators dU^Y_i and dU^E_i will always have the same sign as the individual's utility change, regardless of the chosen numeraire, the two *aggregate* benefit estimators dW^Y and dW^E may have different signs [Brekke (1997)]. This is caused by the different assumptions about interpersonal comparability of cardinal utility underlying these indicators: When there are conflicts of interest between individuals, the way one operationalizes interpersonal comparisons of utility determines the emphasis placed on each person's interests.

Apart from the possibility of different signs, it is difficult to use dW^Y and dW^E directly to judge the empirical importance of a particular choice of aggregation method. They are measured in different units, and since each person may have a different "exchange rate" between units (i.e. different marginal rates of substitution), it is not obvious which conversion rate one should use in an attempt to make them directly comparable. However, an interesting comparison can be made by looking at the *per person costs which would leave the project with exactly zero net benefits* using the two methods. If we denote by C^* the per person costs that implies $dW^Y = 0$, i.e. the maximum acceptable per person cost when equal marginal utility of income is assumed, we have from (5) that

$$C^* = \frac{1}{n}(\sum_{i=1}^{n} \frac{u'_{iE}}{u'_{iY}})dE \tag{8}$$

Similarly, we can denote by C** the maximum allowable per person cost when equal marginal utility of the environmental good is assumed. C** can be defined by

$$C^{**} = \frac{n}{(\sum_{i=1}^{n} \frac{u'_{iY}}{u'_{iE}})} dE \tag{9}$$

which is the per person cost implying exactly $dW^E = 0$. C^* and C^{**} can both be regarded as measures of aggregate benefits from increasing the public good supply. Both are measured in monetary units, but they are based on different assumptions regarding cardinal utilities.[4]

An interesting indicator for the empirical importance of the choice of numeraire is C^*/C^{**}. This will be the central indicator in our empirical results. We will denote this the *MAC* ratio, since it describes the ratio of *Maximum Acceptable Cost.* Mathematically, the *MAC* ratio is given by

$$MAC = \frac{1}{n^2}(\sum_{i=1}^{n} \frac{u'_{iE}}{u'_{iY}})(\sum_{i=1}^{n} \frac{u'_{iY}}{u'_{iE}}) \tag{10}$$

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[4] Note the similarity to the uniform variation measures proposed by Hammond (1994). The uniform compensating variation is defined as the total amount that society is willing to pay, in the form of a uniform poll tax on all individuals, in order to be allowed to move from the status quo to an alternative social state.

It can easily be seen from (10) that if all individuals have the same marginal rate of substitution between income and the environmental good, the *MAC* ratio = 1. Hence, if we are concerned only with ordinary market goods in a perfectly competitive market (assuming no corner solutions), the maximum acceptable per person costs will be the same using both measurement methods. However, whenever marginal rates of substitution differ between individuals, the numeraire problem will arise. This will be the case in a number of circumstances, e.g. when some goods are rationed; but for simplicity, we will concentrate on the case where the good is a public (environmental) good.

For later reference, we note that (10) could alternatively be expressed as

$$MAC = \frac{1}{n^2}\left(\sum_{i=1}^{n} WTP_i\right)\left(\sum_{i=1}^{n} \frac{1}{WTP_i}\right) = (\overline{WTP})\,(\overline{WTP^{-1}}) \qquad (11)$$

where ($\overline{\mathrm{WTP}}$) is the average of all respondents' marginal willingness to pay, while ($\overline{\mathrm{WTP}^{-1}}$) is the average of all respondents' *inverse* willingness to pay. Thus, even if we do not have direct information on people's public good requirements, MAC ratios can be calculated using individual willingness to pay data only, assuming that the project is marginal.[5] The MAC ratios reported in the next section were calculated using equation (11).

Generally, a given numeraire will favour the interest of a person if the numeraire is of relatively *low* value to that person [Brekke (1997)]. If, for example, a person cares little about money, this person's net benefits expressed in money terms must become a large number. On the other hand, if the same person cares a lot about the environment, her net benefits expressed in environmental units may be a quite small number. Consequently, using money as the numeraire will favour those with a relatively high valuation of the environment, compared to using environmental units as the numeraire.

The *MAC* ratio presupposes that costs are shared equally between individuals. Under this assumption, unless the project is a Pareto improvement, those who have the lowest valuation of the environmental good will always be project's opponents, because the cost they have to pay always exceeds their willingness to pay. Thus, the benefit measure *C** will systematically give less weight to the interests

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[5] For a discussion of non-marginal projects, see Appendix 1.

of the project's opponents than C^{**}, implying that the *MAC* ratio ≥ 1 will always hold.[6] In other words, given the assumptions employed here, assuming that everybody has the same marginal utility of income will *always favour the project*, compared to the alternative assumption of equal marginal utility of the public good.

The indicators presented above presuppose that the project can be regarded as marginal. If the project is non-marginal in the sense that individuals' marginal rates of substitution change significantly due to the project's implementation, the above indicators provide only approximations. However, regarding the *MAC* ratio, errors caused by changes being non-marginal will generally go in both directions, because the public good requirement is overestimated for those who have positive net benefit from the project and underestimated for those who have negative net benefit from the project. We thus cannot know a priori whether the *MAC* ratio is over- or under estimated in the case of a non-marginal public good change. In the special case of quasi-linear utility, however, the *MAC* ratio will be correct even if the public good change is non-marginal. See Appendix 1 for more details on this issue.

3. Empirical results

The disturbing part of the theoretical results discussed above is that one way of operationalizing cardinal and interpersonally comparable utility *systematically* favours certain interest groups, compared to another, equally simple method. However, if this bias were of a small empirical magnitude, it might still not be of much practical importance. To examine the empirical significance for applied cost-benefit analysis of the choice of assumption regarding cardinal utilities, we have calculated the *MAC* ratio from seven contingent valuation studies, using individual willingness to pay (*WTP*) data. Unfortunately, our results indicate that the choice of numeraire (corresponding to a certain choice of assumption on cardinal utilities) may be extremely important.

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[6] It is possible to calculate similar indicators with other assumptions concerning the distribution of costs. A more generally applicable indicator may be denoted the *TMAC* ratio, the ratio of maximum allowable *total* costs, where $TMAC = MAC$ if $C_i = C$ for every i. One will generally have a *TMAC* ratio > 1 if costs are shared "under-proportionally" with individual marginal willingness to pay for the environmental good, i.e. $C_i = K(u'_{iE} / u'_{iY})^{\alpha}$, where K and α are positive constants, and $\alpha<1$. Equal distribution of the costs is a special case of under-proportionally cost distribution. If costs are "over-proportionally" distributed ($\alpha>1$), we get a *TMAC* ratio < 1. However, overproportional cost sharing seems to be a fairly peculiar sharing rule, leading to severe incentive compatibility problems. Proportional cost distribution will give a *TMAC* ratio $= 1$. See Brekke (1993) and Medin (1999) for details on this issue.

The studies we have used are those of Loomis (1987), Navrud (1993), Bateman et al. (1995), Bateman and Langford (1997), Bateman et al. (1997b), Magnussen et al. (1997), and Strand and Wahl (1997). All the studies examine willingness to pay to avoid reductions in certain specified recreational services, except the Magnussen et al (1997) and the Bateman et al (1997) surveys, which measure willingness to pay for an *increase* in recreational services.[7] Several of the studies varied the survey design between subsamples; for those studies, results are reported for each subsample. The table below reports results for a total of 18 subsamples. A brief summary of each study is provided in Appendix 2. All calculations are based on open-ended *WTP* data.[8]

A well-known problem in contingent valuation research is that average *WTP* estimates, and thus C^*, can be sensitive to extremely high reported individual values. A common approach to such "outliers" is simply to omit them from the data, assuming that they are caused by errors, misunderstandings, strategic responses, or protest reactions on respondents' part. On the other hand, if these "extreme" observations do reflect respondents' valuations, one may obviously understate average (and aggregate) willingness to pay by omitting them.

When environmental units are used as numeraire in the aggregation of individual welfare effects, one encounters a similar problem regarding extremely *low* observations of individual willingness to pay (low u'_{iE} / u'_{iY}, or correspondingly, high u'_{iY} / u'_{iE}). Taken literally, a zero willingness to pay for a public good implies that an *infinite* amount of the public good is required to compensate for the cost this particular person has to pay. Correspondingly, the social welfare loss measured in environmental units caused by forcing such persons to pay a positive cost is also counted as infinite, and the project will not be socially desirable, regardless of how much other persons are willing to pay.

Just as contingent valuation practitioners have to think carefully about how to treat extremely high and "infinite" willingness to pay-bids, we have to consider how to treat zero willingness to pay-bids when using environmental units in the aggregation of individual values. These zero bids may be given different interpretations. One possible interpretation is that zero bidders have a positive but very small willingness to pay. The most extreme assumption to make in our context, however, is to take the zero

[7] If changes are non-marginal, WTP to *avoid loss* measures the individual's equivalent variation, while WTP for a *gain* measures the compensating variation.

[8] In this elicitation procedure, respondents are asked questions about how much they are willing to pay, and are free to state whatever amount they want. An alternative procedure is that of dichotomous choice, in which respondents are asked yes/no-questions such as "would you be willing to pay X?", where X is varied across the sample. See Appendix 2 for a description of the method used in each survey.

bids literally, since this implies that some respondents have *infinite* public good requirements. Correspondingly, the presence of a zero bid will always imply that $C^{**} = 0$ (in which case the MAC ratio is not well defined).

In all studies except those by Loomis (1987) and Strand and Wahl (1997), *before* the willingness to pay question, respondents were faced with a "payment principle" question concerning whether they were, in principle, willing to pay anything at all for the environmental change. Those who responded negatively to this question were *not* asked to state their *WTP*. It seems reasonable to assume that some of these "no-bidders" did so to protest against the very idea of valuing the environment in monetary terms, or against accepting a personal responsibility for the problem at hand. The environmental good may still be important to such respondents' welfare. Others might have responded "no" because their marginal valuation was indeed zero. Finally, some respondents may reply "no" to the payment principle question because they actually have a negative WTP. However, it is very difficult to judge which respondents belong to which group, and what level of "true" *WTP*, if any, such "no" responses might correspond to.

Since the correct treatment of "zero"-bids is far from obvious, and the same is true for "no"-bids, we have calculated two different versions of the MAC ratios. The first version is based on the assumption that *all* "zero"- and "no"- bids reflect very small, but positive WTPs. *How* small is determined, somewhat arbitrarily, as 5 percent of the *lowest strictly positive bid* reported in that survey. This assumption probably implies an underestimation of some respondents' public good requirements, because a WTP of zero, taken literally, would imply an infinite public good requirement and thus an infinite MAC ratio. Since our assumption here does not allow anyone to have a lower WTP than 5 percent of the lowest strictly positive observation, MAC ratios will not go to infinity.[9] However, for "no"-bidders whose "true", but unobserved valuations are significantly higher than zero, our assumption may imply that MAC ratios are overestimated: Very small observations tend to yield large MAC ratios, while medium-sized observations have much less dramatic impact on MAC ratios.[10]
The second version of the MAC ratio is calculated after omitting all "no"- and "zero"-bids from the datasets. This amounts to an assumption that the "true" valuations underlying such observations are

[9] Using, for example, 1 percent of the strictly positive bid yields dramatically higher MAC ratios.

[10] The assumption that utility is increasing in both income and the public good implies that no respondent has a negative WTP. If in fact some of the no or zero bids reflect negative "true" WTPs, some individuals must have negative marginal utility of either money or the public good. The former is inconsistent with the assumption of equal marginal utility of money, and thus employment of C* as a welfare estimate, while the latter is inconsistent with equal marginal utility of the public good, and thus implies that C** is not a correct welfare estimate.

distributed in exactly the same way as the set of strictly positive observations. This yields conservative estimates of MAC ratios compared to the previous version, since values close to zero tend to imply high MAC ratios.

It turns out that the assumptions one makes on “no”- and “zero”-bids is quite essential for the magnitude of the MAC ratios. Ideally, then, we should have more information about these observations; but since the surveys were designed with elicitation of monetary values in mind, such information is not available. Below, we will focus on the second, most conservative version.

Table 1 reports our empirical results. Column (1) shows the number of observations in each study, while column (2) reports the percentage no- and zero-bids (they are lumped together because we could not separate no-bids from zero-bids in some of the surveys). Column (3) reports the first version of the MAC ratios, assuming that no- and zero-bids reflect a very small, but positive *WTP* (5 percent of the lowest bid in that survey). This yields extremely high *MAC* ratios, ranging from 23 [subsample 1, Strand and Wahl, (1997)] to 22,434 (!) [(subsample 2, Bateman et al, (1995)].[11] Thus, if one accepts the assumptions underlying the first version of MAC, the estimated aggregate monetary benefit indicator is reduced by a factor of up to about 22,000 by replacing the conventional assumption of equal marginal utility of income by an assumption of equal marginal utility of the environmental good.

Column (4) reports the more conservative version of *MAC* ratios, i.e. after *all* zero- and no-bids are omitted from the dataset. This approach yields considerably less extreme *MAC* ratios, varying between approximately 2 [the four subsamples in Strand and Wahl (1997) and two subsamples from Magnussen et al. (1997)] and 307 [subsample 2 in Bateman et al. (1997)]. However, even the smallest *MAC* ratios of approximately 2 are, in one sense, large, since they imply that using environmental units as numeraire instead of money almost *halves* the maximum acceptable per person cost which leaves the project socially desirable. In the study with highest MAC ratio, the maximum acceptable per person cost varies with a factor of up to 307, depending on whether one employs an assumption of, respectively, equal marginal utility of income or of the environmental good.

[11] To understand this seemingly bizarre result, recall that if zero bids are taken literally, the *MAC* ratio goes to infinity, since this implies infinite public good requirements.

Table 1. MAC ratios (ratio of maximum acceptable costs) under different assumptions on zero- and no-bids and the single lowest bid. (If choice of aggregation unit does not matter, the MAC ratio = 1.)[12]

Survey	*MAC-ratio*				
	(1) N	*(2) Zero- and no bids, per cent of N*	*(3) Each zero- and no bid = 5 percent of smallest positive bid*	*(4) All zero- and no bids removed*	*(5) Zero-and no- bids and smallest bid removed*
Bateman et al 1995					
Subsample 1	846	15	20,202	38	27
Subsample 2	2051	15	22,434	11	5.9
Bateman and Langford 1997					
Subsample 1	93	37	8,647	70	40
Subsample 2	90	63	378	6.7	4.5
Subsample 3	88	6.8	93	5.2	4.5
Subsample 4	80	16	5,894	83	53
Bateman et al 1997					
Subsample 1	143	18	11,598	169	138
Subsample 2	126	10	18,003	307	226
Loomis 1987	78	17	82	3.2	2.8
Magnussen et al 1997					
Subsample M 1	143	60	101	3.1	2.8
Subsample M 2	139	59	34	2.1	2.1
Subsample S 1	139	47	97	2.9	2.7
Subsample S 2	132	49	87	2.3	2.1
Navrud 1993	161	32	806	4.2	2.6
Strand and Wahl 1997					
Subsample 1	140	14	23	1.8	1.8
Subsample 2	140	13	30	1.8	1.8
Subsample 3	138	28	60	1.8	1.7
Subsample 4	145	21	69	2.3	2.2

N = Number of respondents
Zero-bids = respondents reporting a zero willingness to pay
No-bids = respondents responding “no” to the payment principle question

[12] Most surveys also include a number of respondents who say they don’t know, or who do not answer the WTP question. These respondents have been omitted from the data in all studies, except from the Magnussen et al. (1997) survey, where we could not distinguish such respondents from zero bidders.

Table 1 shows that the MAC ratios emerging from the data of Bateman et al. (1995), Bateman et al. (1997b) and subsample 1 and 4 in Bateman and Langford (1997) are considerably higher than those of the other surveys. One reason for this might be that these studies contain several very small WTP bids, in the sense that the ratio between the smallest and the highest bid reported is large.[13]

Column (5) reports *MAC* ratios when the *single* smallest *WTP* bid is removed from the data, *in addition to* removing the zero- and no-bidders. It turns out that the *MAC* ratio can be surprisingly sensitive to such removal of one single observation. Particularly interesting is subsample 2 from Bateman et al. (1995), where despite the unusually large sample size of about 1,800 observations (after the removal of all zero- and no-bids), the *MAC*-ratio is almost *halved* by removing the single smallest strictly positive *WTP* bid from the data.

To understand this phenomenon, recall the expression for the *MAC* ratio used in equation (11): $MAC = (\overline{WTP})\,(\overline{WTP^{-1}})$. Somewhat imprecisely, one might say that the effect of omitting one observation from the dataset depends on whether this observation's relative impact on these two averages is very different, or rather, asymmetric. Removing a very small bid may have a large impact on $\overline{WTP^{-1}}$, while $\overline{WTP}$ may be quite unaffected.[14] Removing a high bid, on the other hand, is likely to affect $\overline{WTP}$ much more than $\overline{WTP^{-1}}$. In the surveys examined here, removing the single highest bid in addition to all the zero and no bids did not affect *MAC* ratios nearly as much as removing the smallest strictly positive bid.

Some of the studies also estimated the costs of the project. For example, in Magnussen et al. (1997) (subsamples S1 and S2), the annual total costs were estimated at between \$ 0.38 million and \$ 0.51 million[15]. These costs were to be divided between approximately 8,800 households, implying annual costs per household of \$ 43 - \$ 65. The average monetary benefit per household (assuming equal marginal utility of income) was estimated at between \$ 111 and \$ 132 per annum. Thus, benefits appeared to substantially exceed costs, and the project was deemed to be socially desirable. Would

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[13] In subsample 2 from Bateman et al. (1997), for example, the highest bid reported was £ 1000, while the lowest strictly positive bid was £ 0.005. Thus, in this survey, using money as the numeraire implies that the net benefits of the person with the highest WTP is weighted 200,000 times more than the net benefits of the person with the lowest WTP, as compared to the procedure of using environmental units as the numeraire.

[14] For example, imagine a survey where N = 10. Say that 9 respondents report a WTP of \$ 10, while one reports \$ 0.05. Omitting the latter observation would change $\overline{WTP}$ from 9.005 to 10, while $\overline{WTP^{-1}}$ would change from 2.09 to 0.1. Correspondingly, the *MAC* ratio would change from 18.8 to 1.

the policy recommendation of this study be changed if one had, instead, assumed equal marginal utility of the environmental good? If all zero- and no-bids are removed from the dataset, corresponding to the "conservative" MAC ratios reported in column (4), maximum acceptable per household costs (C**) will be between $ 85 and $ 97, which is still below the estimated costs; and the project still yields positive social benefits.[16] Thus, in this particular case, the conclusion of the analysis seems robust.[17]

The *MAC* ratios reported in Table 1 indicate that the way one compares utility between persons may be extremely important in applied cost-benefit analysis. However, the generality of our results depends, of course, on the extent to which the studies we have used are representative of the "typical" response pattern in CVM studies. Also, some of the simplifications employed in our theoretical model may not hold in practice. One objection is concerned with the fact that our theoretical model assumes only *marginal* changes in the public good supply, in the sense that any changes in individuals' marginal rates of substitution between the public good and income due to the project can be disregarded. In practice, for many environmental projects, this will not hold for at least some individuals. Regarding the studies mentioned in Table 1, this seems particularly questionable for subsample 1 in Bateman et al. (1997), and for both subsamples in Bateman et al. (1995) (see Appendix 2 for details). However, as mentioned above, one cannot know *a priori* whether the calculated *MAC* ratios will be too high or too low if the public good change is in fact non-marginal, as the errors will generally go in both directions; and in the special case of quasi-linear utility functions, equation (11) can be used to calculate *MAC* ratios correctly, even if willingness to pay data does not represent marginal changes (see Appendix 1).

It is also somewhat difficult intuitively to understand what measuring in "environmental units" really means. Some may dismiss our results on the grounds that the environmental unit has not been well-defined enough in some or all of the studies we have used. All the surveys do consider measurement problems and problems related to giving a precise definition of the public good, and all authors appear to have given serious consideration to this problem in their survey design. It is certainly often difficult

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[15] We assume the exchange rate between NKR and USD to be 7.828 (31. August 1998).

[16] "Person" is here used interchangeably with "households", thus, we disregard intra-household conflicts of interest in this example.

[17] Note, however, that since all zero-bids are omitted, many respondents who would most likely get a negative net benefit have been excluded from the analysis. Thus, *both* C* and C** may overestimate the projects' net benefits. If all no- and zero bids are included, and interpreted as 5 per cent of the lowest strictly positive bid, the C** estimate is reduced to $ 1.3, which is far less than the estimated annual per household costs; and the conclusion of the cost-benefit analysis is changed.

to specify the environmental good in a precise enough way, and this is a problem which all contingent valuation studies have to grapple with. Since the surveys we have used were designed with the income numeraire in mind, questionnaires cannot be expected to have focused on aspects which are considerably more important in our context than in the traditional context. However, since MAC ratios can be expressed using only monetary valuations (see equation 11), respondents do not in practice need to express their net valuations using environmental units (i.e. their public good requirements). The issue of defining and understanding what “an environmental unit” means only represents a problem in our context to the extent that misunderstandings regarding this prevented respondents from actually reporting their true WTPs (in monetary terms). If the overall pattern of responses in the surveys we have used are typical for CVM surveys, our results on the *MAC-ratios* will also be typical.

4. Conclusions

The results reported above indicate that aggregate social benefit estimates may be extremely sensitive to alternative ways of comparing different individuals’ utility changes. Making non-verifiable assumptions on cardinal and interpersonally comparable aspects of individuals’ utility functions introduces a non-negligible element of arbitrariness into cost-benefit analysis. Our results show that if one assumes that everybody has an equal marginal utility of the public good, instead of the usual assumption of equal marginal utility of income, aggregate monetary benefit estimates are reduced by a factor of between 2 and 307, using our most conservative estimates.

We wish to stress that our aim has not been to argue in favour of one or the other method of making utility interpersonally comparable. We also recognise that money, being a much more generally exchangeable numeraire than environmental units, is the most convenient measurement unit in many contexts. Under certain conditions it may also be argued that assuming equal marginal utility of money is more reasonable than assuming equal marginal utility of the public good: For example, if there are respondents with negative WTPs, the latter assumption would imply that some people have a negative marginal utility of money. However, rejection of the assumption of equal marginal utility of the public good does not imply that the assumption of equal marginal utility of income is correct: Maybe neither assumption is correct.

Our results illustrate that operationalization of interpersonal utility comparisons is extremely important for empirical cost-benefit analysis. In the light of this, we believe that cost-benefit practitioners should take on a much more active attitude towards this issue. Either, considerably more care is required in interpreting aggregate social benefit estimates, or welfare economists must face the question of utility comparisons explicitly, and address the issue of which methods are actually defensible.

Appendix 1

Implications of assuming that a non-marginal project is marginal

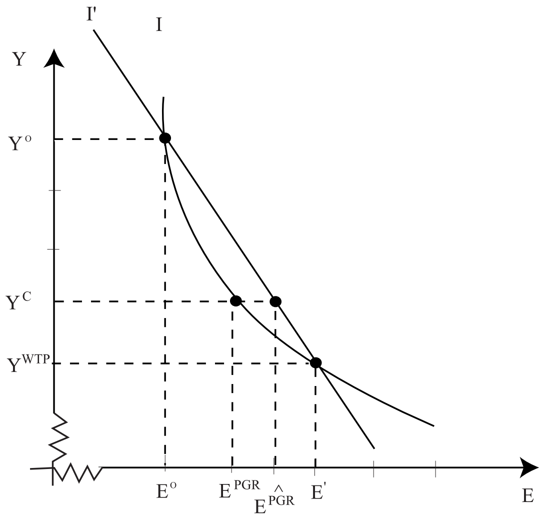
We have assumed that the change in the public good is marginal in the sense that the individuals' marginal rates of substitution between income and the public good are not changed due to the implementation of the project. If this does not hold, individual public good requirements *(PGR),* i.e. the amount of the public good a person requires in order to be willing to pay the per person cost *C*, may be over- or under estimated. This may lead to an over or under estimation of the *MAC* ratio.

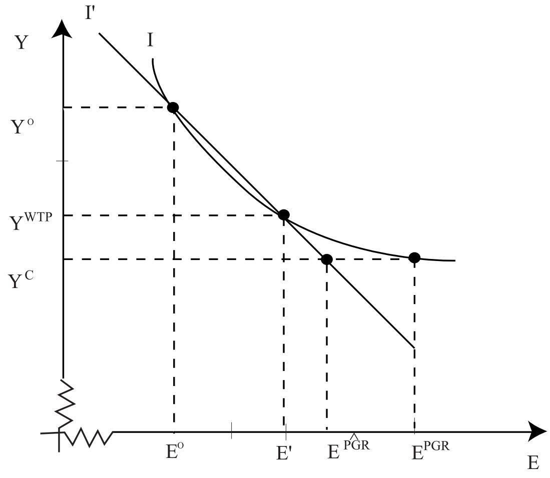
Figure 1 depicts the indifference curves (*I*) for income and a public good for two different persons A and B. Assume a project which implies a non-marginal increase in a public good. Let Y^c - Y^0 = C, i.e. the difference between the person's initial income level, Y^0, and her income after the project Y^c is given by the per person cost C (equal for A and B). The person's willingness to pay for the project is shown as the difference between Y^0 and Y^{WTP} and her monetary net benefit can be expressed as $Y^c - Y^{WTP}$. Figure 1 A shows the situation for person A, who gets a positive net benefit from the project, and figure 1 B shows the situation for person B, whose net benefit from the same project is negative.

In both Figure 1 A and 1 B, *I* is the person's real indifference curve, while *I'* is the estimated linear indifference curve we implicitly assume when the project is taken to be marginal. E^0 is the initial amount of the public good, and is by definition equal for A and B. $E^{'}$ is the amount of the public good after the project, thus dE is given by $E^{'} - E^0$. By moving along *I*, we find the amount of the public good the person requires *(PGR)* to be as well off as before if she must pay the cost, C. *PGR* is given by $E^{PGR} - E^0$ and the net benefit, expressed in environmental terms, is equal to $E^{'} - E^{PGR}$.

By assuming that the project is marginal, we implicitly use *I'* instead of the correct indifference curve I. The estimated PGR corresponding to I' is the difference between $E^{P\hat{G}R}$ and E^0. The figures show that estimated PGR will differ from real PGR. In A's case, where net benefit is positive, *PGR* is overestimated, while in B's case, where net benefit is negative, *PGR* is underestimated.

Figure 1A. **Figure 1B.**





If *PGR* for *all* persons is overestimated, the net benefits expressed in environmental units ($E^{'} - E^{PGR}$), are underestimated, and hence dW^{E} is underestimated. This leads to an underestimation of *C***, and thus an overestimation of the *MAC* ratio. However, in this case the project must be a Pareto improvement, and would thus hardly be too controversial anyway.

Generally, a project will yield positive net benefit for some individuals and negative net benefit for others. Thus *PGR*s will be both over- and underestimated in the same survey; but we cannot in general know the net effect of this. An additional individual with a negative net benefit will contribute to an underestimation of MAC, while additional individuals with positive net benefits will contribute to an overestimation. However, whether the MAC ratio is over- or underestimated depends not only of the *number* of positive versus negative errors, but also of course on the magnitude of each error.

If we make some more specific assumptions about the utility functions, however, we can be more conclusive. In particular, with quasi-linear utility functions, the MAC ratio can be calculated as usual, even if the project is not marginal. To see this, recall first equation (11):

$$\text{MAC ratio} = \overline{WTP}\;\overline{WTP^{-1}} \tag{1A}$$

but let us now assume that $\overline{WTP}$ is willingness to pay for a *discrete* public good change, averaged over all respondents, while $\overline{WTP^{-1}}$ is average inverse willingness to pay for the same discrete change. Now, assume that all utility functions are quasi-linear in income, such that

$$U_i(Y_i, E) = \lambda_i Y_i + v_i(E) \tag{2A}$$

for all *i*, where λ_i is i's marginal utility of income. In this case, equivalent and compensating variation measures coincide [see e.g. Johansson (1993)]; so we do not have to consider which is most relevant here. The change in social welfare (given the utilitarian social welfare function) due to a non-marginal project is then given by the sum of all individuals' discrete utility changes, which gives

$$\Delta W = \Sigma_i[-\lambda_i C + (v_i(E^1) - v_i(E^0)] \tag{3A}$$

where superscript 0 and 1 denote a variable's value before and after the project, respectively. Assuming that everybody has an equal marginal utility of income, we can define C*, as before, as the per person cost which makes society exactly indifferent to the project:

$$C^* = (1/n)\, \Sigma_i[\{v_i(E^1) - v_i(E^0)\} / \lambda] \tag{4A}$$

The expression on the right-hand side of in (4A) now corresponds to $\overline{WTP}$.

Now, in the analysis above, C** has been defined as the maximum acceptable per person cost, given that the marginal utility of the public good is equal for all. Here, the marginal utility of the public good cannot be assumed to be constant. However, we may instead assume that the utility change due to the change in the public good is equal for all *i*, i.e.

$$v_i(E^1) - v_i(E^0) = \Delta v \text{ for all } i. \tag{5A}$$

We may now define C** as the maximum acceptable per person cost assuming that (5A) holds. (Note that (5A) does not necessarily imply that the *marginal* utility of the public good, before or after the project, is equal for all.) With this assumption, the aggregate welfare change induced by the project, measured in environmental units, can be written as

$$\Delta W / \Delta v = -C\, \Sigma_i \lambda_i + n\Delta v \qquad (6A)$$

C** is then found by requiring that $\Delta W = 0$, which yields

$$C^{**} = (n\Delta v)/(\Sigma_i \lambda_i) = 1/[(\Sigma_i \lambda_i / n\Delta v] = 1/[1/n(\Sigma_i(\lambda_i/\Delta v] = (\overline{WTP^{-1}})^{-1} \qquad (7A)$$

We can now calculate the MAC ratio as

$$MAC = C^{*}/C^{**} = \overline{WTP}\ \overline{WTP^{-1}} \qquad (8A)$$

which is exactly the formula used to calculate the MAC ratio in the case of marginal changes.

Appendix 2

Summary of the contingent valuation studies

Bateman et al. (1995)

The public good considered in this study is the Norfolk Broads in East Anglia, the largest wetland area in Britain. When the survey was carried out, the wetlands were under salt-water intrusion from the North Sea. The majority of the area under threat of permanent loss and a variety of different flood alleviation schemes were under consideration. Annual *WTP* to prevent flooding and loss of this area was examined.

The survey was carried out on the site during August and September of 1991. Two subsamples, each consisting of a cross section of visitors and local residents, were collected. Subsample 1 consisted of 846 completed questionnaires in which respondents were asked an open-ended willingness to pay question. Subsample 2 consisted of 2051 completed interviews using a dichotomous choice question followed by an iterative bidding game culminating in an open ended WTP question, responses to which are used in this analysis. However, it should be noted that these responses were found to be strongly affected by starting point bias caused by the initial dichotomous choice bid amount. The number of protest zero bids was assumed to be very low (approximately 30 of both samples). The payment vehicle was increased taxes.

The project can hardly be regarded as marginal for all respondents, as it implies an avoidance of the permanent loss of a public good which is recognised as internationally unique, and hence does not have adequate substitutes. Nevertheless, respondents' *WTP* for the project amounted to, on average, approximately 16 % of their total annual recreational budget. If the public good is regarded as consisting of all recreational services rather than just the ones derived from the Norfolk Broads then it may still be reasonable to assume that marginal rates of substitution between money and recreation services remain approximately the same.

Bateman and Langford (1997)

The survey was carried out on the site in March and April 1993 and examined a number of CV design issues, including ordering and budget constraint effects. Respondents were asked questions concerning their *WTP* for conservation of the recreational facilities at Lynford Stag, a woodland site within the Thetford Forest in East Anglia, England. Lynford Stag has many substitutes, which may indicate that the project can reasonably be regarded as marginal.

351 of 475 approached parties of visitors agreed to be interviewed. The value elicitation procedure was open-ended. All respondents were asked both their annual *WTP* and their *WTP* per visit, the payment vehicle being increased taxes for the former and an entrance fee for the latter. The sample was divided into four subsamples. Subsample 3 and 4 were asked to state their annual recreational budget prior to the WTP question, while subsample 1 and 2 were not asked this budget question. Further, subsamples 1 and 3 were asked about their annual *WTP* prior to their *WTP* per visit, while subsamples 2 and 4 were asked about their *WTP* per visit first. Our calculations of the *MAC* ratios are based on the annual *WTP* data from the four subsamples.

Bateman et al. (1997b)

The survey was carried out in August and September in 1997 and examined *WTP* for a beach protecting scheme at Caister-on-Sea, a village in East Anglia, England. The scheme would result in a considerable extension of the beach at Caister. *WTP* was examined both among the residents of Caister (subsample 1) and the visitors (subsample 2). Subsample 1 consisted of 245 respondents, 143 of which stated a *WTP*. Subsample 2 consisted of 198 respondents, and 126 of these stated their *WTP*. The value elicitation procedure was open-ended, and the payment vehicle was increased taxes (both direct and indirect). No pure protest answers were recorded. As respondents in subsample 1 live adjacent to the beach and see it as a vital part of the sea defences protecting their homes it seems very likely that this good would not be seen as marginal for those respondents. Regarding subsample 2, however, it seems more reasonable to assume that the change in the environmetal good is marginal.

Loomis (1987)

In this survey the public good in question is Mono Lake, one of California's largest lakes, which is located in the eastern part of the state. Water diversions that would normally flow into the lake provides water for the residents of Los Angeles. The diversion rate at the time the survey was carried out was 100.000 acre feet. The water diversion rate had already caused some reduction of the environmental quality of the lake, and a further reduction would take place if the diversion rate didn't decrease. *WTP* for a reduction in the diversion rate was examined. The respondents were explained the impact a maintenance of the diversion rate would have on issues like recreational access to the lake, species diversity and scenic visibility of the lake. We find it reasonable to regard the project as marginal.

The survey was conducted as a mail survey in 1985 and the sample was drawn randomly from California's phone directories. The response rate was 44% and 78 out of 108 respondents answered the WTP question. The value elicitation method was dichotomous choice, with a following open-ended question. Our calculations are based upon the open-ended *WTP* responses, which may be affected by starting point bias caused by the amount of the dichotomous choice bid. The payment vehicle was an increase in households' monthly water bills or an increase in rent for households whose water was included in their rent.

Magnussen et al. (1997)

Two surveys examine annual *WTP* for increased environmental quality of two polluted watercourses in two different local councils in Norway: Langenvassdraget in Ski county (the S sample) and Gaustadvannet/Ånøyavassdraget in Melhus county (the M sample). The environmental quality was defined through a careful description of the water quality, recreational facilities and species diversity of the watercourse in question. It is reasonable to look upon the projects as marginal because they both imply a small increase in the environmental quality. There are also many substitutes to the two watercourses.

The surveys were carried out in June 1991. The samples were drawn randomly from their respective counties and divided into two subsamples: 1 and 2. In subsamples M 1 and S 1 the respondents were asked to state their *WTP* for water quality and their *WTP* for recreational facilities and species diversity separately. Their *WTP* for the project was calculated as the sum of these two *WTP* responses. In subsamples M 2 and S 2 the respondents were asked about their *WTP* for the whole project directly. Some respondents were removed from the subsamples due to their age, lack of information or protest answers. In subsample M1, 8 respondents out of a sample size of 151 were omitted and 10 out of 149 were omitted from subsample M 2. Subsamples S 1 and S 2 consisted of 150 respondents each, whereof 11 were removed from subsample S 1 and 18 from subsample S 2. The value elicitation procedure was open-ended, but payment cards were used to help the respondents to state their *WTP*. Nevertheless the respondents were also free to state other amounts. The payment vehicle was increased taxes.

The annual total costs of the project in Ski county were estimated to be \$ 0.38 - \$ 0.51 million. With number of households of approximately 8800, annual costs per household would be somewhere between \$ 43 and \$ 58. The costs for the project in Melhus county were not estimated.

Navrud (1993)
In this survey the public good in question is the fish stock in Audna, a watercourse in the south of Norway. The fish stock in Audna had been heavily reduced due to acidic water. When the survey was carried out, the watercourse had been limed and stocked with fry for some years. The liming maintained the fish stock and the survey examined annual *WTP* for continuing the liming. If the liming were stopped, the fish stock would decrease heavily. Despite this significant reduction of the public good, we find it reasonable to assume that the marginal rates of substitutions can be regarded as approximately constant, as there are many substitutes to the public good.

The survey was carried out through telephone interviews in 1990 and aimed to examine the non-user values of the fish stock. Other surveys were conducted to examine the use-values. The sample was randomly drawn from the Vest-Agder county's phone directories. The sample size was 200 of which 161 answered the *WTP* question. The value elicitation process was open-ended, and the payment vehicle was payment to a special fund to be used only for liming of Audna. 31 of the zero bids were assumed to be protest bids.

Strand and Wahl (1997)
The survey was carried out in April and May 1997 and aimed to examine *WTP* for a marginal reduction of the municipal parks in Oslo, the capital of Norway. The sample was divided in to four subsamples where two were asked for their *WTP* for a 5% reduction of the parks (subsamples 1 and 2) and the others were asked for their *WTP* for a 10% reduction (subsample 3 and 4). Further, the value elicitation process was open-ended, but an initial bid, which varied between the subsamples, was given. The responses were found to be affected by starting point bias caused by the amount of this initial bid. The payment vehicle was increased municipal taxes.

In subsample 1 the initial bid was \$ 64, and 16 out of 156 respondents did not answer the *WTP* question. In subsamples 2 and 3 the initial bid was \$ 128, and 11 and 14 respondents out of sample sizes of respectively 151 in subsample 2 and 152 in subsample 3, were omitted, because they did not answer the *WTP* question. In subsample 4 the initial bid was $ 256 and 14 out of 159 respondents did not answer the *WTP* question.

The project treats a relatively small reduction of the public good in question. However, no significant difference was found between average *WTP* in subsamples 2 and 3. Hence, according to these subsamples, the respondents, on average, have approximately the same *WTP* for a 5% and a 10 % reduction of the public good. This appears to be an embedding effect, perhaps caused by respondents not being capable to of distinguishing between the two proposed changes. However, this result is, strictly speaking, inconsistent with the assumption of marginality, since a "marginal" project must be understood here as a project which leaves marginal rates of substitution unchanged. As WTP is not significantly affected by increasing the environmental change from a five percent to a ten percent 10 improvement, respondents' marginal valuation of the last 5 percent appears to be much lower than their valuation of the first five percent.

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