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Energy-related External Costs due to Land Use Changes, Acidification and Eutrophication, Visual Intrusion and Climate Change.

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Table of Content

Table of Content 3

1. Introduction 5
  1.1 Land use change and biodiversity 5
  1.2 Acidification 6
  1.3 Eutrophication 6
  1.4 Visual Intrusion 6
  1.5 Climate Change 7

2. Economic valuation of land use change and biodiversity loss 8
  2.1 Introduction 8
  2.2 The NEEDS approach 8
    2.2.1 The object to be valued 8
    2.2.2 The physical impact pathway 10
    2.2.3 Preference elicitation 10
    2.2.4 Discussion 14
  2.3 Meta-analytic value function for land use change and biodiversity loss 15
    2.3.1 Economic valuation literature on land use change and biodiversity loss 15
    2.3.2 Conversion of land use change to Ecosystem Damage Potential (EDP) 17
    2.3.3 Standardisation of value estimates 17
    2.3.4 Biodiversity value data descriptives 18
    2.3.5 Value function for biodiversity 21
  2.4 A comparison of the meta-analytic value function and restoration costs 22
  2.5 Conclusion 23

3. Estimates of impacts of acidification, eutrophication and visual impacts of wind and hydro 25
  3.1 Introduction 25
  3.2 Monetary estimates of acidification impacts on fresh water fish 26
    3.2.1 Summary of European studies 26
    3.2.2 Discussion of acidification studies 32
    3.2.3 Linking physical and economic indicators in acidification 33
    3.2.4 Linking changes in emissions and exceedance of critical loads 34
    3.2.5 Dose-response functions for exceedance of critical loads and fish damage 35
    3.2.6 Economic valuation of fish damages 35
    3.2.7 Contingent Valuation of reduced fish damage 37
    3.2.8 The potential for benefit transfers 37
    3.2.9 Conclusion 38
  3.3 Monetary estimates of impacts of eutrophication on use and non-use values 39
    3.3.1 Summary of European studies 39
    3.3.2 European Contingent Valuation and Travel Cost studies 45
    3.3.3 European Travel Cost studies 46
    3.3.4 European Benefit Transfer studies 47

3.3.5 European Meta-analysis 49
3.3.6 Discussion of eutrophication studies 51
3.4 Monetary estimates of landscape aesthetic impacts of renewable energy - Wind 52
3.4.1 Summary of European studies 52
3.4.2 Discussion of aesthetic effects of wind studies 61
3.5 Monetary estimates of landscape aesthetic effect of hydropower 62
3.5.1 Summary of studies 62
3.5.2 Discussion of aesthetic effects of hydro power studies 70
3.5.3 Example of benefit transfer of estimates of hydro power environmental effects 71
3.5.4 Aesthetic effects of overhead power transmission lines 72
3.5.5 Possibilities for benefit transfer of estimates for aesthetic effects of power transmission lines 75
3.6 Benefit transfer methods and validity tests of benefit transfer 76
3.6.1 Potential for benefit transfer 76
3.6.2 Practical Benefit Transfer Guidelines for Cost-Benefit Analysis 77
3.6.3 Validity of benefit transfer and acceptable transfer errors 84

4. Marginal damage cost of greenhouse gas emissions 97
4.1 Introduction 97
4.2 Marginal Damage Costs 97
4.2.1 Integrated Assessment Models 97
4.2.2 Discounting and ‘deep’ uncertainty 99
4.2.3 Equity weighting 99
4.2.4 Illustrative marginal damage values for the CASES project 101
4.2.5 Marginal damage costs of other greenhouse gases 103
4.3 Marginal Abatement Costs 104
4.3.1 Stabilization targets 107
4.3.2 Research approach: meta-analysis 108
4.3.3 Description of the database 108
4.3.4 Results 113
4.3.5 Conclusions and discussion 118
Annex I Database of GHG stabilization studies 119

5. Note on discounting for CASES 124
5.1 Discounting around the world 124
5.2 Social rate of time preference 125
5.3 The opportunity cost of capital 127
5.4 Step-declining discount rate 128
5.5 Recommmendations and conclusions 128

1. Introduction

The external costs of energy production include a number of items that have hitherto proved hard to value in monetary terms. These cost items include the external value of land use change, acidification of the aquatic environment, eutrophication, visual intrusion, and the cost of damage resulting from climate change. In relation to climate change, but also to other dynamic elements in the power sector, the rate of discount or the discount factor to compute the present value of future costs and benefits has attracted much attention. This report screens recent literature in these areas for information regarding these costs that can be used at the European level. Information on these costs is commonly found only in a limited number of studies, covering a limited geographical region, and considering a limited number of ecosystem types and functions. Therefore, this report also discusses the technique of 'benefit transfer' to transfer values or functions that were assessed in 'study sites' to other 'policy sites' across Europe. The primary objective of the research is to find data and functions that can be built in the EcoSense model, the computer model that is used by the CASES project to assess the external costs of power generation in scenarios over the period 2005-2030.

The research was conducted in three steps. In the first step, a database of relevant literature was built and briefly described (see Kuik et al., 2007). In the second step, the data was analysed in order to find unit values or value functions that could potentially be included in the EcoSense model in order to improve and extend its coverage. In the third step, the preliminary results of the research were presented to and discussed by the CASES consortium at the mid-term project meeting in Brussels.

In the following we will briefly describe our main research findings for each item that we addressed.

1.1 Land use change and biodiversity

We reviewed a new approach to the assessment and valuation of land use change and biodiversity that was developed for the NEEDS project. Key to this approach is a relative measure of species' abundance – the "Potentially Disappeared Fraction" (PDF) – that is associated with land use, and, in a rather complicated way, with the deposition of acidifying pollutants. We identified a number of key assumptions in this approach that warrant further research. Most of these assumptions relate to the physical impact pathway – from deposition to species' abundance. We have flagged the critical assumptions, leaving elaboration to further ecological research.

One of the assumptions, however, relates to the economic valuation of these changes in species' abundance. As will be explained in the next chapter, the critical assumption here is that restoration costs are a good proxy to the social benefits of (or willingness to pay for) land use changes. To examine this assumption, we carried out a meta-analysis of economic studies related to the valuation of land use change and biodiversity loss. The purpose of this meta-analysis is to provide a basis for the transfer of existing value estimates for biodiversity loss to the impacts of air pollution on biodiversity. In order to provide values for changes in biodiversity that can be adjusted to reflect important local characteristics, we estimated a value function.

We propose that future research examines the possibility of using this function in EcoSense, possibly as an alternative to the current restoration cost approach. In principle, location specific biodiversity values can be calculated using data on each explanatory variable in the locality of the biodiversity under consideration. Before implementing our value function in EcoSense, we might want to consider *a priori* restrictions on certain coefficients.

1.2 Acidification

Based on an extensive review of literature on the acidification of the aquatic environment, we conclude that there are at least two options in selecting a *unit of transfer* for this valuation estimate: i) per tonne of reduced sulphur deposition, measured as critical load exceedance, and ii) per square kilometre of reduced land area with critical load exceedance. Both these options combine the weak and strong sustainability indicators of economic value and critical loads.

1.3 Eutrophication

There are several studies available on this topic, especially from the northern part of Europe. There are also several examples of benefit transfer studies and meta-analysis. However, most studies are rather old (early 1990s) and the linking of physical indicators and economic valuation is a major challenge for benefit transfer for this topic. Therefore we do not propose to include the monetary assessment of eutrophication damage in EcoSense at this moment.

1.4 Visual Intrusion

European studies that value aesthetic effects of wind parks are largely from the Nordic countries (Norway, Denmark and Sweden) and from southern Europe (France and Spain). Damage costs are highly project- and site-dependent, and are also very sensitive to the alternative or reference scenario (“how would electricity be generated without the wind mill?”). We certainly believe that aesthetic effects of wind parks can in principle be monetized with a sufficient degree of certainty to be used in cost-benefit analyses. But before we implement functions and values in EcoSesnse we need more primary valuation studies.

The relatively few studies available, and the “bundle” of environmental goods valued (purposely or not) in valuation studies of aesthetic effects of hydropower, makes it difficult to foresee accurate benefit transfer. More original studies of good quality, and with a clear understanding of which (bundle of) environmental effects to value, are needed first.

More studies are also needed for a robust valuation of the external costs of transmission lines.

1.5 Climate Change

There is large uncertainty on the damage costs of greenhouse gas emissions. Specific estimates differ because of alternative assumptions on discounting the future, on aggregating damages across poor and rich countries and on the treatment of 'deep' or 'structural' uncertainty on the possibility of catastrophic events. The Social Cost of Carbon project of the UK government suggests a central value that increases from €23/tCO_2 in 2000 to €41/tCO_2 in 2030. For sensitivity analysis an upper value is suggested of €53/tCO_2 in 2000 to €110/tCO_2 in 2030. We compared these damage cost estimates with estimates of abatement costs to reach long-term stabilization targets for CO_2 in the atmosphere, from 550 to 350 ppmv CO_2. On average, across the entire range of stabilization targets, marginal abatement costs in 2025 could be between € 13 and € 119 per tonne of CO_2. For the strictest target (350 ppmv), marginal abatement costs in 2025 could range between € 74 and € 227/tCO_2.

References:

Kuik, O. Brander, L., Nikitina, N., Navrud, S., Magnussen, K., Fall, E.H. (2007). A Database of Studies on Energy-related External Costs due to Land Use Changes, Acidification and Eutrophication, Visual Intrusion and Climate Change. CASES Project, EU DG Research, Brussels.

2. Economic valuation of land use change and biodiversity loss

Luke Brander, Nataliya Nikitina, and Onno Kuik

2.1 Introduction

The monetary valuation of ecosystems and biodiversity is controversial. One can argue that it is either not possible to "reduce" nature to a commodity that can be exchanged for commodities that can be bought and sold in markets, or at least, that it is not morally right to do so. Even if one assumes that people can have well-defined preferences for ecosystem services, at least at the margin (valuing only incremental changes in provision), it can be argued that a purely anthropocentric approach to valuation discriminates against the inalienable rights of future generations or of other, non-human living species. Without any prejudice to these arguments, we assume in this section that 1) people can have well-defined preferences over marginal changes in the provision of ecosystem and biodiversity services, and 2) that the economic value of these services is determined by these preferences only. To be sure, we do *not* assume that these preferences are necessarily derived from selfish or short-sighted motives. People may well have preferences that take account of the interests of other humans, non-human species, and future generations.

The effects of different fuel cycles and energy infrastructures on ecosystems and biodiversity have not yet been sufficiently assessed (Ott, Baur et al. 2006). In the NEEDS project of the Sixth Framework Programme of EU DG Research, an assessment approach has been developed on the basis of the restoration cost method (Ott, Baur et al. 2006). This section discusses the NEEDS approach. After presenting and discussing this approach, the section also presents a practical suggestion to augment this approach in the future with willingness-to-pay estimates based on measured preferences.

2.2 The NEEDS approach

Under the assumption that it is possible and morally justified to assess the economic value of changes in ecosystems and biodiversity, there are at least three major practical problems in the actual assessment. The first concerns the concept and the exact definition of ecosystems and biodiversity itself: what is the object to be valued? The second concerns the physical impact pathway: how do certain pressures (such as acidifying substances) affect the object to be valued. The third problem concerns the elicitation of preferences from the relevant population and, indeed, the definition of the relevant population. We will discuss the NEEDS approach to the valuation of land use changes and biodiversity along the lines of these three questions.

2.2.1 The object to be valued

There is now a formidable literature on the concepts of ecosystems and biodiversity, their functions and economic services, and ways and means to assess and measure these functions and services (e.g., Alcamo 2005; Albers and Ferraro 2006). Ecosystems are dynamic complexes of plants, animals and microorganism communities and the nonliving environment interacting as a functional unit (Alcamo 2005). Biodiversity is the variability among living organisms from all sources, including terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part (Alcamo 2005). Biodiversity can refer to variety in genes, species, and ecosystems; it can refer to various spatial and temporal scales; and it can include several attributes such as compositional, structural, and functional biodiversity. Furthermore, for each level (genes, species, ecosystems) biodiversity can be measured in various ways, for example by counting, by determining relative abundance or the degree of similarity (Albers and Ferraro 2006). There is some controversy among experts whether biodiversity should be considered an environmental service itself or that it contributes to the creation of environmental services (Albers and Ferraro 2006). However, the latter view (biodiversity contributes to environmental services) is gaining more support (Alcamo 2005; Albers and Ferraro 2006), and this is also the view underlying the NEEDS approach to the valuation of ecosystems and biodiversity.

The NEEDS approach to valuing ecosystems is based on work by Koellner (2001) who developed practical characterisation factors for ecosystem quality for use in life cycle assessments. The approach takes a relative measure of species' abundance as its starting point. For practical purposes, species' abundance is measured by the number of vascular plant species per square meter. The actual (average) number of vascular plant species for a specific land use *i* (S_i) is compared to a reference land use (S_{ref}). The reference land use in Koellner's approach is a composite of various land uses that occur in the Swiss lowlands. The number of plant species per square meter in the reference land use (S_{ref}) is 40. For a specific land use (e.g., *i* = broad-leafed forest), the ratio of the actual (average) number of vascular plant species per square meter (S_i =24) and the number of plant species of the reference land use (S_{ref} = 40) gives the relative species' abundance (24/40 = 0.60). The NEEDS approach uses the inverse of the relative species' abundance, called the Potentially Disappeared Fraction (PDF). PDF is defined as:

$$PDF_i = \frac{S_{ref} - S_i}{S_{ref}} = 1 - \frac{S_i}{S_{ref}} \tag{1}$$

Hence, if species' richness increases, PDF decreases and vice versa. For the land use type "broad-leafed forest", the PDF is 1 – 0.60 = 0.40. In this way, any land use can be assigned a specific PDF. NEEDS (and Koellner) use the CORINE land cover classification, to assign PDFs to land use types.

A change in land use, from *i* to *j*, can be converted in a change in PDF (dPDF) by the formula:

$$dPDF_{i \to j} = (b+1) * (PDF_j - PDF_i) \tag{2}$$

where *b* is a species accumulation factor that accounts for species' interaction outside the converted area. If a land use change increases the PDF (dPDF>0, hence, reduces species' richness), PDF outside the converted area may also increase. Based on the literature, *b* is assigned a value of 0.2. For example, a change from "broad-leafed forest" (PDF= 0.40) to “Built up land” with a number of vascular plant species of 1 (hence, PDF = 1 – 1/40 = 0.97) results in dPDF = 1.2 * (0.97 – 0.40) = 0.68.

2.2.2 The physical impact pathway

The above formulae allow us to compute the biodiversity impacts of direct land use change. For damage assessment of energy systems, this allows, for example, for the computation of the (physical) impact on biodiversity of land use change due to the construction of a power plant or a hydroelectric dam. But it does not directly allow for the assessment of ecosystem damage via air pollution. In order to do this, additional steps have to be made.

These steps are made with the use of a damage model (Natuurplanner) that was developed to assess the impacts of airborne emissions of acidifying substances (SO_x, NO_x, NH_3) on natural ecosystems in the Netherlands (Goedkoop and Spriensma 2001).

Table.2.1 Impact of deposition of pollutants on PDF of natural ecosystems

Air pollutant	dPDF * m^2 * year per kg deposition
SO_x	1.73
NO_x	9.52
NH_3	25.94

source: (Goedkoop and Spriensma 2001)

In the NEEDS approach, the Dutch dPDF values are directly transferred to other European countries, although potential transferability problems are acknowledged due to, for example, differences in geography, ecosystem composition, and background deposition.

2.2.3 Preference elicitation

The two previous steps allow us to express in a common metric (dPDF) energy-cycle related direct changes in land use and indirect changes due to the deposition of airborne pollutants. The final question is: how do we value these changes in PDF?

Ideally, and in line with the overall ExternE methodology, economic valuation should be based on the willingness-to-pay (WTP) by consumers and firms to protect or enhance biodiversity, or by their willingness-to-accept (WTA) compensation for its decline. However, because of the lack of reliable information on these WTP/WTA measures for a wide variety of ecosystems across Europe, NEEDS takes the approach of valuing these changes by means of restoration costs: what would it cost to restore a unit of land area with a higher PDF value (lower biodiversity value) to a unit with a lower PDF value (higher biodiversity value)?

Information on restoration costs is taken from a number of German studies into the costs of the restoration of damaged habitats. The consulting firm Bosch & Partner defined a set of measures and assessed standardized costs for the restoration of habitats from various “starting” biotopes to various “target” biotopes. For example, restoration from “built-up land” to “broad-leafed forest” involves planning, deep tilling of the soil, afforestation, and maintenance. Total discounted costs (in prices of 2004) are € 2.89/m^2 (Ott, Baur et al. 2006). This gives an estimate of the present value of the external cost of land use change due to the construction of energy infrastructure, for example a power plant. Annual costs can be computed by taking the annuity of the present value, taking into account the life span of the power plant and the interest rate. At an interest rate of 5 percent and an expected life span of 50 years, the annual external cost is € 0.16/m^2. The costs can also be related to the change in PDF. The change in PDF from “broad-leafed forest” to “built-up land” is 0.67. Thus, present costs are 2.89/0.67 = € 4.31/PDF/m^2 and annual costs are € 0.24/PDF/m^2/year.

The German unit restoration costs are transferred to other European countries (EU27 plus Iceland, Norway, Switzerland and Turkey) using purchasing power standards (PPS). The unit costs can directly be applied to the valuation of the “biodiversity” costs of energy infrastructure construction. For the EU25, based on different starting and target biotopes, the present costs are between € 0.57 and € 7.39/PDF/m^2 and annual costs range between € 0.03 and € 0.40/PDF/m^2/year.

The use of the restoration cost method for the valuation of ecosystem damage due to the deposition of airborne pollutants requires some additional assumptions. Deposition of pollutants cannot be directly linked to specific land use changes. Ott et al. (2006) have therefore computed restoration costs per PDF for all land use changes, and selected the lowest restoration costs per PDF as the “cheapest method to “produce” a certain amount of biodiversity as expressed by a PDF-decrease.” (Ott, Baur et al. 2006: 32). The lowest restoration cost per PDF was € 0.49/PDF/m^2 (from integrated arable to organic arable). This is taken to be the minimal marginal cost of improving biodiversity per PDF and m^2 and is used for the damage assessment due to the deposition of airborne pollutants. For other European countries reduction costs per PDF are adjusted by the purchasing power standards discussed above.

Finally, the background acidification and eutrophication pressure is taken into account. It is assumed that unit damage is low at low pressure levels, because ecosystems have buffering capacity to absorb pollutants. Unit damage increases when certain thresholds (critical loads) are exceeded. Critical loads differ per country, based on differences in soil characteristics.

The computation of the unit value of ecosystem damage (VPDF) due to the deposition of pollutant *i* in country *r* is as follows:

$$VPDF_{i,r} = dPDF_i * RC * PPS_r * SNA_r * PRES_r \qquad (3)$$

$VPDF_{i,r}$ = Unit value of ecosystem damage per kg deposition of pollutant i per m^2 in country r
i = SO_x, NO_x, or NH_3
r = country
dPDFi = Change in PDF due to the deposition of 1 kg pollutant i (see Table.2.1).
RC = Restoration cost in Germany (0.49 €/PDF/m2)
PPSr= Purchasing power standard for country r
SNAr = Share of natural land in total land area of country r.
PRESr = Background acidification and eutrophication pressure index for country r (index NL = 1).

As an example we will calculate VPDF of SO_x deposition for Germany. $dPDF_{SOx}$ = 1.73 (see Table.2.1). Germany has a purchasing power standard of 1 and hence RC * PPS_{GER} = 0.49 * 1 = 0.49 €/PDF/m^2. Its share of natural land is 0.34: SNA_{GER}=0.34. Background acidification and eutrophication pressure is 0.899 ($PRES_{GER}$ = 0.899). Thus VPDF $_{SOx,,GER}$ = 1.73 x 0.49 * 1 * 0.34 * 0.899 = € 0.26/kg. VPDFs for all European countries and for the three pollutants have been calculated and are presented in Table 2.2.

Table 2.2 Unit values of ecosystem damage for European countries

Country	Area (m2)	PPS	SNA	PRES	$VPDF_{SOx}$	$VPDF_{NOx}$	$VPDF_{NH3}$
EU25	4.09E+12	0.92	0.49	0.360	0.14	0.76	2.06
Austria	8.42E+10	0.97	0.63	0.526	0.27	1.50	4.09
Belgium	3.10E+10	0.96	0.22	0.959	0.17	0.94	2.57
Cyprus	9.24 E+9	0.89	*0.44*	*0.008*	0.00	0.01	0.04
Czech Republic	7.94 E+10	0.51	0.36	0.628	0.10	0.54	1.47
Denmark	5.23 E+10	1.28	0.31	0.218	0.07	0.40	1.10
Estonia	4.84 E+10	0.58	0.68	0.270	0.09	0.50	1.35
Finland	3.91 E+11	1.16	0.91	0.277	0.25	1.36	3.72
France	5.61 E+11	0.97	0.36	0.290	0.09	0.47	1.29
Germany	3.66 E+11	1.00	0.34	0.899	0.26	1.43	3.89
Greece	1.32 E+11	0.78	0.61	0.008	0.00	0.02	0.05
Hungary	9.37 E+10	0.54	0.26	0.603	0.07	0.39	1.08
Ireland	7.57 E+10	1.16	0.36	0.074	0.03	0.14	0.39
Italy	3.07 E+11	0.94	0.44	0.273	0.10	0.53	1.44
Latvia	6.39 E+10	0.51	0.55	0.172	0.04	0.23	0.61
Lithuania	6.38 E+10	0.51	0.35	0.253	0.04	0.21	0.57
Luxembourg	2.93 E+09	0.97	0.36	0.959	0.28	1.56	4.26
Malta	1.21 E+09	0.68	*0.80*	*0.273*	0.13	0.69	1.89
Netherlands	4.12 E+10	0.98	0.25	1.000	0.21	1.14	3.11
Poland	3.13 E+11	0.49	0.33	0.706	0.10	0.53	1.45
Portugal	9.08 E+10	0.80	0.50	0.034	0.01	0.06	0.17
Slovak Republic	4.94 E+10	0.46	0.45	0.808	0.14	0.78	2.13
Slovenia	2.00 E+10	0.72	*0.62*	*0.679*	0.26	1.41	3.85
Spain	5.16 E+11	0.80	0.49	0.032	0.01	0.06	0.16
Sweden	4.50 E+11	1.14	0.69	0.299	0.20	1.10	2.99
United Kingdom	2.49 E+11	0.95	0.36	0.307	0.09	0.49	1.33
Bulgaria	1.12 E+11	0.39	0.44	0.080	0.01	0.06	0.17
Croatia	5.70 E+10	0.51	*0.40*	*0.679*	0.12	0.65	1.76
Romania	2.39 E+11	0.38	0.37	0.153	0.02	0.10	0.27
Turkey	7.79 E+11	0.51	*0.40*	*0.044*	0.01	0.04	0.11

NB: numbers in italics have been taken from similar countries

Source: (Ott, Baur et al. 2006)

2.2.4 Discussion

The NEEDS approach to the assessment and valuation of biodiversity or ecosystem damages due to energy-related externalities is based on a number of assumptions:

1. A change in PDF (dPDF) is an acceptable indicator for ecosystem damage
2. The PDF change (dPDF) per kg pollutant used in the approach is 1) a reasonably accurate description of "true" ecosystem damages in the Netherlands; and 2) it is valid for all other European countries.
3. Pollution-induced ecosystem damage only takes place on natural lands (no ecosystem or biodiversity damage on agricultural and urban lands).
4. There is a direct (linear and positive) relationship between background levels of acidification and eutrophication, and marginal unit damage.
5. Restoration cost is a reasonable proxy for willingness-to-pay; and transferring restoration cost to other countries by adjusting it with purchasing power standards is a valid methodology.

With respect to the first two assumptions, it should be mentioned that the current NEEDS method only takes damage of terrestrial ecosystems into account. Damages to aquatic ecosystems have not been taken into account. This report offers an overview of valuation studies of the damage due to acidification and eutrophication to aquatic ecosystems (Section 3). In the mid-1990s, a fairly comprehensive study in Norway on the damage of acidification to fish stocks found a willingness-to-pay to reduce the emissions of SO_x of 4.0 to 7.7 €/kg for sulphur deposition above critical loads. We can compare this to the NEEDS estimate of SO_2 damage to terrestrial ecosystems in Sweden,[1] setting PRES to 1 to reflect that we are only considering deposition above critical loads. When PRES is adjusted in this way, the NEEDS estimate for terrestrial ecosystem damage is € 0.67/kg SO_x. This would suggest that the value of damages to aquatic ecosystems could be substantially higher than damages to terrestrial ecosystems. In our conclusions on aquatic damages we do stress, however, the uncertainties in the valuation studies and the difficulties of transferring country-specific estimates to other European countries.

Another aspect of the first two assumptions, specifically the validity of the relationships between deposition of airborne pollutants and PDF change across Europe, could benefit from additional future research. We have, however, not addressed this relationship in the current study. We have also not addressed the third and fourth assumption, that is, we have not studied biodiversity impacts on agricultural and urban lands and we did not address the relationship between background levels of acidification and eutrophication and marginal damage.

We did, however, address the fifth assumption on the restoration cost as a reasonable proxy for willingness-to-pay, and the assumptions on transfer of these costs across Euro-

[1] NEEDS does not give results for Norway; Sweden is the closest alternative.

pean countries. In fact, we developed and estimated a willingness-to-pay function as an alternative to the restoration cost approach. The willingness-to-pay function can be directly used to estimate values for changes in biodiversity for different parts of Europe and we therefore do not have to make specific transfer assumptions. The derivation of the willingness-to-pay function is described in the next section.

2.3 Meta-analytic value function for land use change and biodiversity loss

This section presents the results of a meta-analysis of economic studies related to the valuation of land use change and biodiversity loss. The purpose of this meta-analysis is to provide a basis for the transfer of existing value estimates for biodiversity loss to the impacts of air pollution on biodiversity.

2.3.1 Economic valuation literature on land use change and biodiversity loss

The economic valuation literature addresses a broad range of changes to land use and biodiversity. We collected over 160 valuation studies that deal in some way with land use change and biodiversity. The literature has been compiled from existing online databases of economic valuation studies (including the Environmental Valuation Reference Inventory – EVRI); economic journal databases; internet searches; and contact with researchers. For the purposes of conducting a statistical meta-analysis, however, we were only able to code and standardise data for 24 of these studies. These studies are listed in Table 2.3 below. In order to be included in the statistical analysis a study needed to provide: a clear description of the land use or biodiversity change being valued, the size of the area under investigation, and a total monetary value for the change. From these 24 studies we were able to code 42 separate value observations. In some cases we are therefore taking more than one value observation from a single study.

Table 2.3 List of studies included in the database

No.	Lead author	Year	Title
1	Adger N.	1994	Towards estimating total economic value of forests in Mexico.
2	Alvarez-Farizo B.	1999	Estimating the benefits of agri-environmental policy: Econometric issues in open-ended contingent valuation studies.
3	Amigues J.- P.	2002	The benefits and costs of riparian analysis habitat preservation: a willingness to accept/willingness to pay contingent valuation approach.
4	Bann C.	1998	An economic analysis of tropical forest land use options, Ratanakiri Province, Cambodia.
5	Banzhaf S.	2004	Valuation of natural resource improvements in the Adirondacks.
6	Berrens R.	1998	Exploring nonmarket values for the social impacts of environmental policy change.
7	Beukering P.	2003	Economic valuation of the Leuser National Park on Sumatra, Indonesia.
8	Bonnieux F.	1997	Valuing the benefits of landscape restoration: A case study of the Cotentin in Lower-Normandy, France.
9	Brouwer R.	1998	Contingent valuation of the public benefits of agricultural wildlife management: The case of Dutch peat meadow land.
10	Bulte E.	2002	Forest conservation in Costa Rica when non-use benefits are uncertain but rising.
11	Drake L.	1999	The Swedish agricultural landscape - economic characteristics, valuations and policy options.
12	Farber S.	1999	Using conjoint analysis to value ecosystem change.
13	Garber-Yonts B.	2004	Public values for biodiversity conservation policies in the Oregon coast range.
14	Georgiou S.	2000	Contingent ranking and valuation of river water quality improvements: Testing for scope, sensitivity, ordering and distance decay effects.
15	Gianni C.	2000	Willingness to pay for landscape preservation: a case study in Mediterranean agriculture.
16	Hanley N.	2006	Estimating the economic value of improvements in river ecology using choice experiments: an application to the water framework directive.
17	Jenkins D.	2002	Valuing high altitude spruce-fir forest improvements: importance of forest condition and recreation activity.
18	Johnston R.	2001	Estimating amenity benefits of coastal farmland.
19	Kniivilä M.	2002	Costs and benefits of forest conservation: regional and local comparisons in Eastern Finland.
20	Lehtonen E.	2003	Non-market benefits of forest conservation in southern Finland.
21	Muriithi S.	2002	Conservation of biodiversity in the Arabuko Sokoke Forest, Kenya.
22	Naidoo. R	2005	Biodiversity and nature-based tourism at forest reserves in Uganda.
23	Ruijgrok E.	2004	Reducing acidification: The benefits of increased nature quality. Investigating the possibilities of the contingent valuation method.
24	Zhongmin Xu	2003	Applying contingent valuation in China to measure the total economic value of restoring ecosystem services in Ejina region.

These 42 value observations are for different locations, ecosystem types, and have been estimated using different valuation methods. Table 2.4 gives an overview of the number of observations in each category.

Table 2.4 Number of value observations by continent, ecosystem type, and valuation method.

Continent	Europe 19	N America 14	S America 2	Asia 5	Africa 2
Ecosystem	Forest 16	River 17	Coastal 2	Other 6	
Valuation method	Contingent valuation 18	Choice experiment 17	Other 7		

2.3.2 Conversion of land use change to Ecosystem Damage Potential (EDP)

The changes in land use described and valued in the economic valuation studies cover a broad spectrum of land use types and degrees of change. Valuation studies have addressed changes from one land use type to another, changes in area of a particular land use type, and changes in ecosystem quality. In order to compare the estimated values of these various changes, we have described each change in a generic metric of land occupation and land use change, namely the Ecosystem Damage Potential (EDP) characterisation developed by Koellner and Scholz (2007; 2008). For all practical purpose, EDP and PDF are identical.

This generic characterisation of land use types is based on an extensive meta-analysis of information about species diversity from 5,581 sample plots in Central Europe. Species diversity characterisation factors for 53 land use types and six intensity classes have been calculated from this data. The land use typology is based on the CORINE Plus classification. The EDP for each land use category is a linear transformation of the relative species numbers in each category.

Using the CORINE Plus land use classification and the descriptions of land use given in each valuation study, we assigned EDP values to the land use types in our valuation database. In other words, an EDP value was assigned to the original land use and the new (post-change) land use for each value observation. The change in EDP for each land use change was then simply calculated by subtracting the original EDP from the new EDP.

2.3.3 Standardisation of value estimates

Value estimates for changes in land use have been reported in the literature in many different metrics, currencies, and for different years. In order to compare these values, we standardised them to Euros per EDP per hectare per year at 2004 prices. Values in other currencies were converted to Euros using purchasing power parity exchange rate factors from the World Bank World Development Indicators (WDI). Values were converted to 2004 prices using GDP deflators, again from the WDI.

2.3.4 Biodiversity value data descriptives

Having described land use changes in terms of EDP to reflect impacts on biodiversity and standardised monetary value estimates to a common metric, we are now able to examine the value of biodiversity and the factors that influence this value. For our data set the average value per EDP per hectare per year is Euro 4,706. The median value is Euro 604, which indicate that the distribution of values is skewed with a long tail of high values.

We might expect that the estimated values will vary with a number of important variables, such as income, population density, the size of the area under consideration, and the magnitude of the change in EDP being considered. Figure 2.1 to Figure 2.4 present scatter plots of these variables against value per EDP.

There appears to be no discernable relationship between GPD per capita and the value of biodiversity (Figure 2.1). Although we might expect that people's willingness to pay to protect biodiversity would increase as they become richer, it appears that the value of biodiversity is constant across income levels.

We do observe a clear positive relationship between population density and biodiversity value (Figure 2.2). As population increases, so does the value of biodiversity. This makes sense in that more people living in the vicinity of an area with high biodiversity means more people that hold values for that biodiversity.

Figure 2.3 shows that there are diminishing returns to scale for areas supporting biodiversity. As the size of area increases the value of biodiversity per hectare declines. In other words, adding an additional hectare to a large ecosystem area is worth less in terms of biodiversity than adding an additional hectare to a small ecosystem area.

We also observe that values per EDP decline as the magnitude of change in EDP increases (Figure 2.4). This suggests that the amount that people are willing to pay per unit of biodiversity diminishes as the change in biodiversity increases. The value of a change in biodiversity is therefore not a simple linear function of change in biodiversity, but depends on the scale of change.

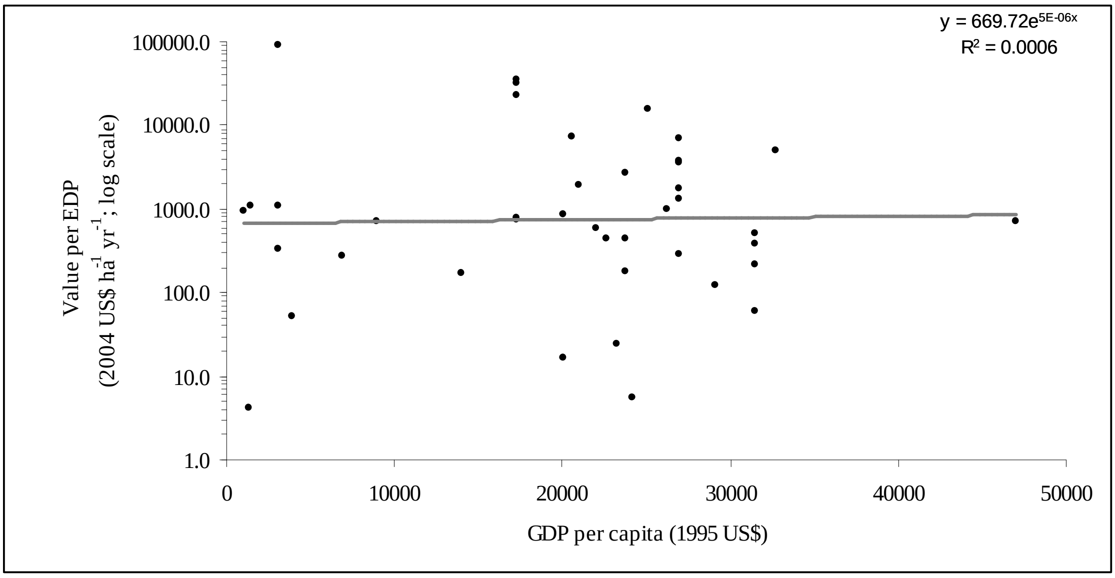

Figure 2.1 Value per EDP plotted against GDP per capita

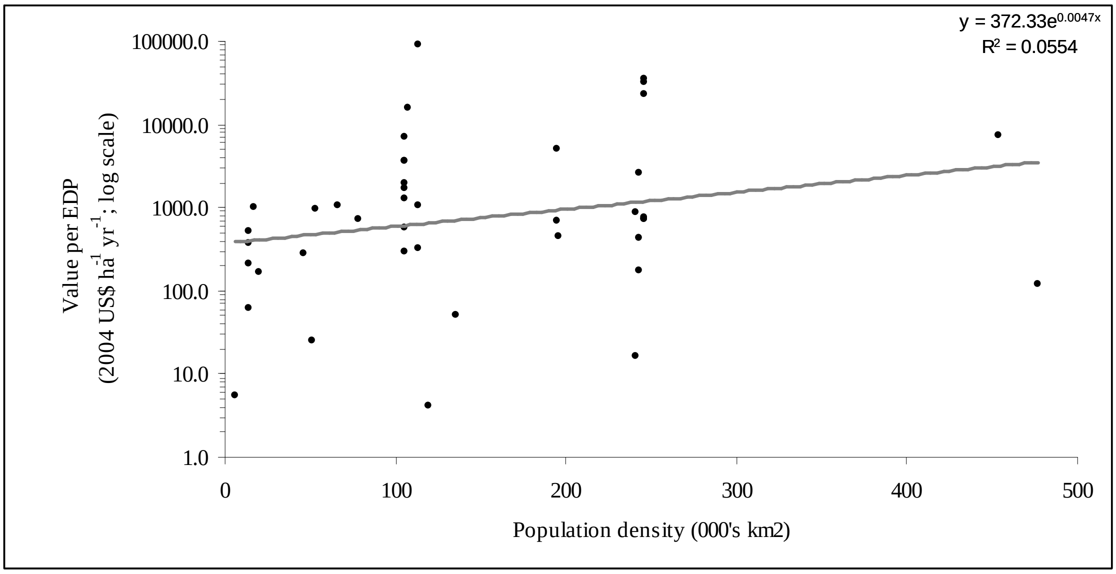

Figure 2.2 Value per EDP plotted against population density

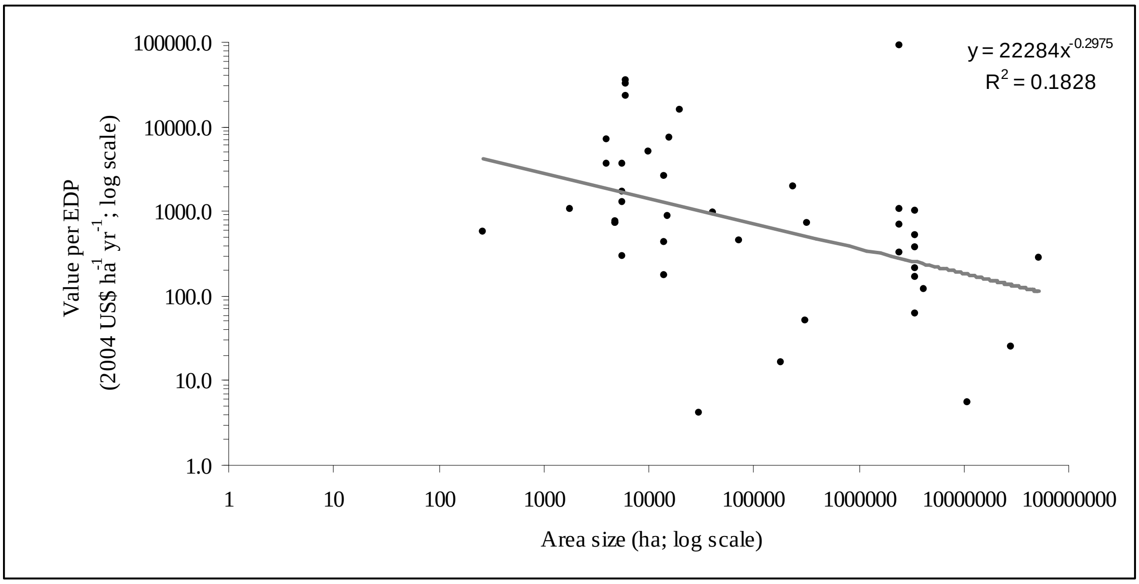

Figure 2.3 Value per EDP plotted against area size

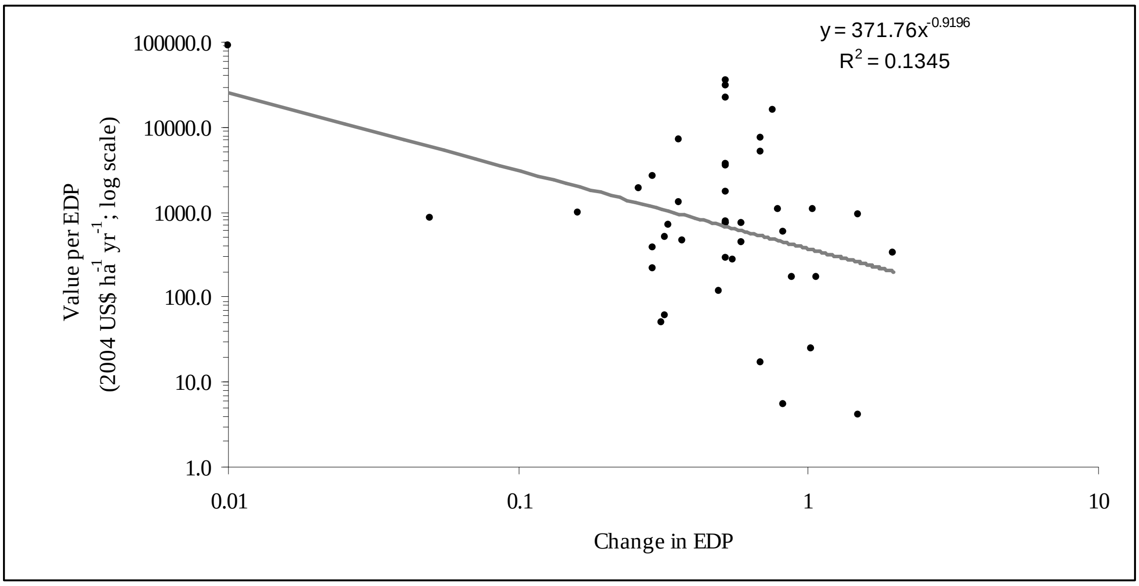

Figure 2.4 Value per EDP plotted against change in EDP

2.3.5 Value function for biodiversity

In order to provide values for changes in biodiversity that can be adjusted to reflect important local characteristics, we estimate a value function. Using a statistical regression analysis we can relate the variation in estimated biodiversity values to a number of key explanatory variables. The natural log of Euros per EDP per hectare per year was used as the dependent variable. The explanatory variables include population density, size of affected area, change in biodiversity, and type of ecosystem. The best-fit model is presented in Table 2.5. The results confirm the sign of the relationships between biodiversity value and population density, size of affected area, and change in biodiversity illustrated in Figures 2.1-2.4. The results also indicate that biodiversity values vary depending on the type of ecosystem, with forests and coastal ecosystems having higher biodiversity values than rivers.

Table 2.5 Estimated biodiversity value transfer function

Variable	**Coefficient**
Constant	8.740
Population density (ln)	.441
Forest (dummy variable)	1.070
River (dummy variable)	-.023
Coast (dummy variable)	.485
Change in EDP	-2.010
Area (ha)	-.312

We propose that this function is used to estimate monetary values for changes in biodiversity (EDP) estimated by the EcoSense model. Location specific biodiversity values can be calculated using data on each explanatory variable in the locality of the biodiversity under consideration. Table 2.6 provides a numerical example of how this function can be used to estimate different biodiversity values for different situations. Due to differences in population density, ecosystem type, extent of change in EDP, and size of area, each site has a different biodiversity value.

Table 2.6 Numerical example of different estimated biodiversity values.

	German forest	Dutch coast	Spanish river
Constant			
Population density	233	393	88
Forest (1=yes; 0 = no)	1	0	0
River (1=yes; 0 = no)	0	0	1
Coast (1=yes; 0 = no)	0	1	0
Change in EDP	0.6	0.4	0.7
Area (ha)	3000	1000	2000
Value per EDP per ha	4963.8	7333.7	1005.3
Total value of change	8,934,790	2,933,465	1,407,396

2.4 A comparison of the meta-analytic value function and restoration costs

Here we will highlight how the function can be used within the current NEEDS approach and how its estimated damage values compare to the damage values from the restoration cost approach.

As described in section 2.3.5, we estimated a willingness-to-pay function for biodiversity change by carrying out a meta-analysis of the international literature on the external costs of land use change. The dependent variable is the natural log of the value of Ecological Damage Potential (EDP) per hectare per year. The explanatory variables include characteristics of the ecosystem: the size of the ecosystem and its type (a distinction is made between forests, rivers and coastal ecosystems); the scale of the ecosystem's change (dEDP); and the size of the population that benefits from the ecosystem's services (expressed as population density). The estimated function is as follows:

$$\ell n(VEDP) = 8.740 + 0.441 \ell n(PD) + 1.070 FOR - 0.023 RIV + 0.485 COA$$
$$- 2.010 dEDP - 0.312 \ell n(AREA) \quad (4)$$

VEDP = Value of Ecological Damage Potential (EDP is basically the same as PDF, but measured per hectare)
PD = Population density ('000 inhabitants/km2)
FOR = Dummy variable for forest ecosystems
RIV = Dummy variable for river ecosystems
COA = Dummy variable for coastal ecosystems
dEDP = Change in EDP
AREA = Size of ecosystem in hectares

$VEDP_r$ is the willingness-to-pay for one unit reduction of EDP per hectare per year for country *r*. In Equation (3), the terms RC * PPS_r can be substituted for $VEDP_r$*10^{-4}, thereby transforming a restoration cost function into a willingness-to-pay function. Equation (3) then becomes:

$$VPDFalt_{i,r} = dPDF_i * VEDP_r * 10^{-4} * SNA_r * PRES_r \quad (5)$$

We will compare the restoration cost approach (Equation (3)) and the willingness-to-pay approach (Equation (5)) in computing the ecosystem damage due to the deposition of 1 kg SO_x in Germany. We already found the damage according to the restoration cost approach: it was € 0.26/kg SO_x. To compute VPDFalt, we first need to compute VEDP for Germany. Assumptions on the explanatory variables are in Table 2.7.

Table 2.7 Explanatory variables in VEDP for Germany

Variable	Dimension	Value	Source/assumption
PD	'000/km2	233	Table 2.6
FOR	share	0.8	Assumption
RIV	share	0.1	Assumption
COA	share	0.1	Assumption
dEDP		1.73*10E-04	$dPDF_{SOx}$*10E-04
AREA	hectare	17472	1)
VEDP	€/ha/year	9000	Equation (4)

1) AREA is the average area of ecosystems in Germany. Calculated as the average size of "Areas of unseparated spaces with roads with little traffic (UZVR)" in Germany. KIS: German Federal Environment Agency.

Filling in VEDP = 9000 in Equation (5) results in a value for VPDFalt of 0.48 €/kg SO_x.

2.5 Conclusion

In summary, the willingness-to-pay approach to valuing ecosystem damage instead of the restoration cost approach increases the unit value of ecosystem damage due to SO_x deposition in Germany from € 0.26/kg to € 0.48/kg. The VPDFalt function can be used for all countries (or even the sub-country level), provided information is available on population density, the shares of different ecosystems, and the average size of ecosystem areas in these countries (or at the sub-country level). Note that a big difference with the NEEDS approach is that country values are not dependent on per capita income, but on ecosystem characteristics and population density.

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3. Estimates of impacts of acidification, eutrophication and visual impacts of wind and hydro

Ståle Navrud and Kristin Magnussen

3.1 Introduction

This chapter provides an overview of recent literature on the external cost estimates of energy-related impacts on land use change, acidification, eutrophication, visual intrusion and climate change across Europe and for selected non-EU countries.

In this section we present an overview of economic studies related to the valuation of

- Monetary estimates of acidification impacts on fresh water fish (chapter 3.2.)
- Monetary estimates of impacts of eutrophication on use values (drinking water, boating, swimming, recreation fishing) and non-use values; (chapter 3.3)
- Monetary estimates of landscape aesthetic impacts of renewable energy- wind (chapter 3.4.)
- Monetary estimates of landscape aesthetic impacts of renewable energy- hydro-power (chapter 3.5.), and
- Benefit transfer methods and tests of validity of benefit transfer (chapter 3.6.)

For each valuation topic above we will provide summaries of recent European studies using different valuation approaches, and discuss the studies and results regarding the availability of estimates, and the potential for benefit transfer.

We have included more studies in the database than the ones described in this section. The database is compiled mainly from the "Environmental Valuation Reference Inventory" (EVRI), and the database includes studies both from Europe and North-America (and one from Asia). In the literature review, however, we focus on recent European studies, because these are the most relevant from a benefit transfer perspective.

We have tried to concentrate on studies which value the goods in question separately, or where it is possible to distinguish the valuation estimates for the relevant topic. This means that for instance studies of WTP for "renewable energy" are not included in chapter 3.4 or 3.5 because they do not separate valuation estimates for aesthetic effects, often not even for wind <u>or</u> hydro, but for a mixture of different "green" energy sources. The same principle is applied for choice of studies for the other topics.

For the relevant topics, we will start with a summary of some of the most relevant studies, then provide an overview of locations valued, benefits included, examine important environmental stressors addressed, and finally an "overall" discussion and conclusion on the availability of studies and the potential for benefit transfer.

3.2 Monetary estimates of acidification impacts on fresh water fish

3.2.1 Summary of European studies

Studies using Contingent Valuation

ECOTEC Research and Consulting (1993): "A Cost Benefit Analysis of Reduced Acid Deposition: UK Natural and Semi-Natural Aquatic Ecosystems: a Contingent Valuation Study of Aquatic Ecosystems" (ECOTEC Research and Consulting, Birmingham, U.K. and the U.K. Department of Environment Food and Rural Affairs, London, U.K.)

This study aims to estimate use and non-use benefits of reducing the acid deposition damage to the aquatic ecosystems of the upland areas of the UK. A total of 2391 respondents (1606 non-users and 587 users) were interviewed between July and September 1993. The non-user group (1,606 respondents) comprised a sample of the general population of the UK, interviewed in their own homes, and provided a basis for estimating the aggregate population WTP. The user group (587 respondents) was interviewed in order to compare use and non-use values. The objective was therefore to interview a specialist sub-group of the population to compare its WTP with estimates of population WTP and to examine the WTP of specific users such as anglers. The survey began by asking a series of questions about attitudes to environmental and other issues. The CV scenario and WTP questions followed. Section C asked respondents some general knowledge questions about environmental issues followed by a series of true/false questions about behaviour and intended behaviour towards the environment. The final section asked a series of questions to establish the respondent's socio-economic characteristics. The payment vehicle was water rates and the WTP was asked for the following 10 years. Each respondent was shown a map and asked to concentrate on the shaded areas, which represented critical loads exceedance maps. The fact that the waters with which the scenario was concerned are concentrated in upland areas was stressed on several occasions during the scenario. The respondent was then asked to concentrate on two pictures, one showing an upland stream with good water quality, demonstrating a variety of animal and plant life, and a second depicting an ecosystem in a stream with poor water quality, with a reduction in the number of species and number of individuals of some species. The changes depicted aimed to demonstrate the type of effects which might result from the exceedance of critical loads for freshwater such as a reduction in Brown Trout numbers, the loss of more sensitivity species but the survival of less acid sensitive species. The interviewer read a description of the changes (observed in many of the streams, rivers, lakes etc) depicted in the two pictures but did not reveal what the cause of the poor water quality was. Refusal rate was 22% for non-users and 26% for users leading to the net respondent numbers mentioned above.

The environmental good valued was aquatic ecosystems (including fish and plant life) that are being negatively affected by acid deposition, more specifically: an improvement in the aquatic ecosystems in upland areas which increases a variety of animal and plant life. The environmental stressor in the case was bio-accumulative substance.

The respondents were shown maps and pictures depicting the effects of acid deposition but were not told that the reason in the decline in some species of fish and other wildlife was acid deposition. The respondents were asked how much more they would be willing to pay as increased water rates for the following 10 years to improve the quality of aquatic environment in upland areas. They were asked the same question three times allowing them to revise their initial bid twice. The average WTP for non-users was £25.33; for non-angler users £33.82 and anglers £38.69 per household per year. The following variables have been found to be significant (at 5% level) in explaining the magnitude of WTP bids: annual gross income, discretionary income status, environmental knowledge; series of dummy variables relating to environmental behaviour and intended environmental behaviour; age; number of visits to upland waters per year; dummy variable to distinguish between non-users and users; and expenditure on environmental protection per year.

The regression analysis for the combined sample is presented in the table below and includes the following variables: (note: the variables with (*) after them are significant at 5% level) annual gross income (*), discretionary income status (*), attitudes to environmental protection measured on scale of 1 to 5; environmental knowledge score out of 5 (*); series of dummy variables relating to environmental behaviour; series of dummy variables relating to environmental behaviour (C3C) and intended environmental behaviour (C3D) (*); age (*); number of visits to upland waters per year (*); whether or not respondent has children under 18; dummy variable to distinguish between non-users and users (*); level of education completed; expenditure on environmental protection per year (*); dummy for whether or not respondent is an angler; gender and whether or not respondent pays the household's water rates. The letter-number combination in brackets refers to question numbers.

Table 1.2.1. presents the mean willingness-to-pay responses of user and non-user populations based on the second revised bid (third bid in total).

Table 1.2.1.: Mean Willingness-to-Pay to Improve Quality of Aquatic Ecosystems in Upland Areas

	Mean Revised Willingness-to-Pay	Standard Deviation	N
Non-Users	25.33	42.92	1519
Users (Non-Anglers)	33.82	33.82	419
Users (Anglers)	38.69	45.07	147

Table 1.2.2: Regression Results for the Combined Sample: Dependent Variable in the Last Willingness-to-Pay Bid

	Coefficient	Standard Error	t-statistics	Significance T
Income	0.000004	0.000001	3.252	0.0012
C3C	0.165128	0.032427	6.060	0.0000
Age Dummy	-0.200899	0.031459	-6.386	0.000
Knowledge	0.029431	0.009663	3.046	0.0024
Visit	0.000658	0.000208	3.169	0.0016
C3D	0.122516	0.032427	3.778	0.0002
Disincom	0.052121	0.016452	3.168	0.0016
EP10.000916	0.000320		2.864	0.0043
User	0.077396	0.030244	2.559	0.0106
Constant	0.938733	0.039489	23.772	0.0000

Notes: Sample size was 1,259 and the F-statistic was 29.89. The adjusted R-square = 0.171

MacMillan, D., N. Hanley and S. Buckland (1996): "A Contingent Valuation Study of Uncertain Environmental Gains"(Scottish Journal of Political Economy, 43, 5, 519-533).

The purpose of the study was to estimate use and non-use values for different levels of acidification impacts to biodiversity in Scottish upland ecosystems. The good valued was biodiversity in Scottish upland ecosystems, the respondents were asked to state their WTP for reductions in losses of/WTP for recovery from damage to, biodiversity in Scottish upland ecosystems from acidification caused by acid rain.

The survey was pre-tested and piloted in a Contingent Valuation (CV) survey of 254 randomly selected Scottish households, that used an open-ended Willingness to Pay question to determine the bid levels used in the main survey. The main survey consisted of a Dichotomous Choice CV postal survey of 2720 households, which resulted in a 67% response rate, or 1820 useable questionnaires. The sample was representative of the Scottish population in terms of age and gender of respondents. Information on income levels was not available.

Each respondent received one of 15 different scenarios and payment amounts. The scenarios varied in terms of the damage and recovery options presented: respondents were told either that action taken to reduce acid rain will result in recovery of the ecosystem to pristine levels; to moderate, fishable levels; or that no recovery would result above current levels. The level of damage was also varied, in that respondent were told that either that it was predicted to be minimal or high, where high was likely to result in species extinction. Respondents were asked whether they were willing to pay the given bid amount and their reasons for paying/not paying. The payment vehicle was higher prices in pollution, generating items such as electricity, cars etc. The survey also collected data on respondent's socio-economic characteristics.

Respondents were asked whether they were willing to pay (yes/no/don't know) a given bid amount for a given level of impact reduction or ecosystem recovery, in terms of levels of biodiversity in upland areas. Fifteen different damage/recovery scenarios and 21 different payment amounts were used across the sample. The questionnaire encouraged respondents to consider available substitute sites before stating their WTP. Mean household Willingness to Pay (WTP) per annum for abatement in acid rain when predicted damage to biodiversity was minimal was found to be £247 and £351 when damage was predicted to be high (1996 British Pounds.) The study found that respondents were willing to pay more to avoid high damage but also preferred the status quo over paying for potential environmental improvements. The authors concluded that in the face of scientific uncertainty over damages/recovery, respondents were risk averse to both environmental gains and losses.

The probability of a Yes response to the bid level was modelled as a function of twenty-nine explanatory variables, two of which were entered first in the regression: the bid level and one of three variables representing the damage/recovery level or time. The remaining variables were selected using a stepwise procedure. A linear model with logit link function (between predictor and explanatory variables) was used. The explanatory variables were not fully listed, but the regression identified eight as significant at the 5% level: income; government (not explained); level of understanding of acid rain; perceived importance of pollution as a social issue; whether the respondent returned their questionnaire quickly or not; and whether they were a member of an environmental organisation.

Mean Willingness to Pay (WTP) amounts per household per annum for recovery to pristine, fishable and no improvement (impacts stay at current levels) were calculated for two different damage scenarios. These are shown in the table below. Mean aggregate Willingness to Pay per year for acid rain abatement when damage to biodiversity was predicted to be minimal was £484m and £688m when predicted to be high (1996 British Pounds.)

Table 1.2.3. Mean (and 90% Confidence Intervals) Willingness to Pay (WTP) amounts for alternative predicted damage and recovery scenarios from acid rain per household per annum (1996 British Pounds**)*

	Pristine	**Fishable**	**No future recovery**	**Mean**
Damage minimal	239 (195-344)	241 (181-592)	272 (201-676)	247 (213-308)
Damage high	299 (237-465)	339 (248-678)	503 (283-2959)	351 (280-500)
Mean	272 (232-345)	298 (230-512)	351 (257-761)	308 (265-365)

Notes:

* Confidence intervals in brackets ** Authors do not state year of values

Ian J. Bateman, Philip Cooper, Stavros Georgiou, Ståle Navrud, Grefory L. Poe, Richard C. Ready, Pere Riera, Mandy Ryan and Christian A. Vossler (2005): Eco-

nomic valuation of policies for managing acidity in remote mountain lakes: Examining validity through scope sensitivity testing". (Aquatic Science 67 (2005) 274-291)

The paper focuses on validity testing of results from Contingent valuation studies. The tests are applied to a large sample study of schemes to alter the acidity levels of remote mountain lakes.

For environmental goods the description of the good and the consequences of any policy designed to affect its provision should have its basis in scientific understanding. Hence, the initial stage of the research involved close cooperation between economists and natural scientists focusing on basic issues of remote acid lakes. Emphasis was also put on developing methods that could effectively convey to the public the current status of water quality in the lakes and potential changes to that status. One important tool used to convey the changes in water quality was the water acidity ladder, which showed the relationship between acidity level and effect on plants and animals in lakes.

Two versions of the survey questionnaire were refined corresponding to the "WTP to avoid further degradation" and "WTP for an improvement" valuation scenarios. In introducing the issue of remote mountain lakes the researchers defined the term and showed the respondent a map of Europe highlighting areas containing remote mountain lakes. Attention was drawn to the Highlands of Scotland as the only such area in the U.K., with respondents being told that there are 400 such lakes there and about 10,000 in Europe as a whole. Respondents were then shown photographs of a single illustrative lake in Scotland taken in summer and winter, followed by photos of various flora and fauna currently found within such lakes. Subsequently, the concept of lake acidification by airborne pollutants were explained via the acidity ladder.

Respondents were told that the costs of the liming programme would be met through a fixed addition to domestic electricity bills. The number of lakes limed in each programme was specified to the respondents. Respondents were then shown a WTP response card which consisted of two "payment cards" next to each other. The lower number of lakes (L) was specified at the top of the left hand columns of WTP-amounts, and the higher number of the lakes (H) specified at the top of the right hand columns. Respondents were asked to work their may down the L column first, a tick being placed next to all those amounts they were definitely prepared to pay for the schemed.

Sampling was undertaken in various locations falling into one of the following categories: On-Site (the area around the Highland lake of Lochnagar, including the nearby Lock Muik visitor Centre), Scotland Off-Site (in the cities of Aberdeen, closer to the highlands and Glasgow) and England (ordered by increasing distance from the highlands: Leeds, Norwich, London and the south coast). A total of 1275 questionnaires with valuation responses were completed. However, this sample size is reduced because of factors such as omitted response to other questions resulting in a final sample of 1096 respondents which was used in the validation modelling exercises.

Mean WTP values are recorded in the table below.

Table 1.2.4. Mean WTP values (£ p.a.) (Numbers in brackets are standard errors)

Site	Question	**WTP for an improvement, £ p.a.**				
		5 lakes	40 lakes	200 lakes	360 lakes	400 lakes
English	Low	11.75 (1.97)	16.39 (1.73)	19.98 (1.96)	21.29 (2.12)	n.a.
Scottish Off	Low	16.13 (1.55)	20.77 (1.27)	24.36 (1.59)	25.67 (1.79)	n.a.
Scottish On	Low	18.30 (2.10)	22.94 (1.72)	26.53 (1.83)	27.84 (1.95)	n.a.
English	High	n.a.	16.54 (2.39)	20.64 (2.04)	22.13 (2.12)	22.13 (2.14)
Scottish Off	High	n.a.	24.31 (1.93)	28.41 (1.49)	29.91 (1.60)	30.18 (1.64)
Scottish On	High	n.a.	23.91 (2.50)	28.01 (2.02)	29.50 (2.04)	29.77 (2.06)
		WTP to avoid further degradation, £ p.a.				
English	Low	16.39 (3.62)	17.59 (3.38)	18.52 (3.55)	18.86 (3.69)	n.a.
Scottish Off	Low	20.17 (2.61)	21.37 (2.20)	22.30 (2.42)	22.64 (2.60)	n.a.
Scottish On	Low	20.29 (2.35)	21.49 (1.76)	22.42 (1.93)	22.76 (2.13)	n.a.
English	High	n.a.	23.40 (4.63)	24.13 (4.22)	24.39 (4.32)	24.44 (4.35)
Scottish Off	High	n.a.	27.35 (3.46)	28.08 (2.75)	28.35 (2.84)	28.40 (2.88)
Scottish On	High	n.a.	24.54 (3.16)	25.27 (2.19)	25.53 (2.24)	25.58 (2.28)

Regression models are reported in the paper.

Individual WTP for the protection of additional lakes seems to decline rapidly as we move beyond a minimum number of lakes, in this case five, protected. Thus, while the average estimated value of £16.39 to £20.29 to protect five lakes suggests a substantive aggregated WTP to protect a portion of the lakes, protection of additional lakes beyond this minimum level is not valued highly by individuals. The maximum values for protecting all 400 lakes ranges from £22.40 to £30.18.

Navrud, S.: "Linking Physical and Economic Indicators of Environmental Damages" (2001a)

(Chapter 6 in C. L. Spash and S. McNally (eds.) 2001: Case Studies in Ecological and Environmental Economics. John Wiley & Sons Ltd.)

This contingent valuation study aims to find the willingness to pay (WTP) amongst Norwegians for an increase in the fish populations of Norwegian lakes. Aquatic life in these lakes has been severely damaged due to acid rain from long transported air pollution from Europe. The scenario used in the study was liming Norwegian lakes so as to raise the amount of fish in, and the environmental good valued was to get back viable fish stocks in the brown trout lakes and the salmon and sea trout rivers in areas where the stocks are currently reduced or extinct.

Most of the Norwegian lakes have had a major loss of aquatic life due to acidification from transboundary air pollution. A nation-wide survey with in-person interviews of 1009 Norwegian households was carried out in April 1996. Each respondent was asked to answer "yes", "no" or "don't know" to only one amount, and the amount varied from

100 to 1,000 Norwegian Kroner (NOK) (1 NOK = 0.11 Euro). This dichotomous choice WTP question was then followed by an open-ended WTP question.

Both open ended bidding and dichotomous choice (DC) was used. In the DC part, the amount varied from 100 to 1000 NOK (1996). The results from the DC part found the WTP to be between 653 to 705 NOK per household per year, whilst the mean WTP from the open ended part was 367 NOK/household/year. Combining the OE and DC results the annual WTP per household seems to be in the range of 367-705 NOK.

Household income and recreational fishing had a significant, positive effect on WTP while a low educational level had a significant, negative effect. The mean WTP calculated from the single and multiple logit models was 705 and 653 NOK/household/year, respectively.

Table 1.2.5.Percentage of respondents who answered yes to the dichotomous choice question, at different values (WTP for increased fish stocks per household per year, NOK, 1996)

	100 NOK	200 NOK	300 NOK	500 NOK	1000 NOK
Percentage of yes answers	70	66	59	51	43

Table 1.2.6. Results from the open ended part (WTP for increased fish stocks per household per year, NOK, 1996)

Mean WTP	367
Median WTP	200
Standard deviation	664
Percentage of zero WTP	31

3.2.2 Discussion of acidification studies

- Overview of locations

The European surveys carried out for the valuation of acidification seems to be concentrated to Norway and Scotland – areas where acidification of lakes have been considered a major environmental problem.

- Benefits included

The surveys on acidification focus on biodiversity – plant and animal life – in lakes, and fish populations used for angling in particular.

- Important environmental stressors

Acid rain is the main environmental stressor addressed in the acidification surveys – mainly through (trans-boundary) long-distance air-borne pollution.

- Availability - methodology – possibilities for benefit transfer

There are a limited number of European valuation studies and estimates for acidification, although there are probably more studies around than cited in this literature review. For instance, there are several studies in Norway regarding WTP for liming of different lakes in Norway. However, these are not included in the EVRI database, and are often written in the native language and not available in English.

The WTP estimates obtained in the cited studies vary between countries and between the "good" specified. Regarding the possibilities for benefit transfer, the results of the Bateman et al. study, are a bit worrying, because WTP for improvements in an "additional" number of lakes, is rather limited, while there is a substantial WTP for the first (five) lakes.

Another methodological challenge as regards benefit transfer (as well as the original valuation studies) is the way to link the actual physical changes in the lake (environmental good) and the valuation, and to convey this good to the respondents. Knowing exactly which "good" or "change in good" which has been valued is vital for good benefit transfer exercises.

One study which took this really seriously and where the economists worked together with natural scientists in order to get the physical links correct, is the Navrud (2001a) study. The Bateman et al. (2005) study builds on this approach as well. Hence, we will report how the linking between the physical and economic indicators of environmental changes was carried out in the Navrud paper. (The text below is a shortened version of the text in the original paper.)

3.2.3 Linking physical and economic indicators in acidification

The damage function approach was used as the framework in the study. The damage function approach applied to the impact of air pollution on ecosystems has four main steps:

i. use air dispersion models to estimate how changes in emissions of a pollutant affect atmospheric concentrations;

ii. calculate how changes in atmospheric concentration will affect deposition and concentration of the pollutant in the recipient (soil, water);

iii. use dose-response functions to calculate the impacts on affected ecosystems (e.g. fish stocks) from changed depositions, and

iv. estimate damages (or benefits) by calculating the economic values of the impact. In the case of reduced emissions damage function approach should rather be termed the benefit function approach.

The concepts of critical levels and critical loads were developed in the framework of the 1979 Convention on Long Range Trans-boundary Air Pollution (LTRAP) of the United Nations Economic Commission for Europe, and widely accepted as a basis for designing control strategies in the Second Sulphur Protocol signed in Oslo in June 1994 (UN-ECE 1996). Critical levels refer to the direct effect of pollutant concentrations while critical loads are derived for pollutant depositions, also taking into account accumulation effects in soil and water. For sulphur and nitrogen acidity the critical load is defined as "the highest deposition of acidifying compounds that will not cause chemical changes leading to harmful effects on ecosystem structure and function" (UN-ECE 1996). Thus, step ii) and iii) in the damage function approach would consist of calculating changes in critical load exceedance and its impact on fish stocks.

3.2.4 Linking changes in emissions and exceedance of critical loads

Critical loads for surface water is calculated from the concept that the annual acid deposition to a water shed should not exceed the amount of alkalinity (i.e. buffer capacity) that is produced annually in the water shed and the lake. The buffering capacity is determined by the geology and soil characteristics of the water shed. The acid deposition should be less than the buffering capacity in order to leave a minimum level of buffer capacity which is necessary to avoid damages to aquatic organisms including fish. Thus, this critical biological value varies with the natural buffer capacity of surface water, measured as acid neutralising capacity.

In Norway this limit has been set at 20 µeq/l (microequivalents per litre) which was found to be a realistic value for aquatic organisms (Lien et. al. 1992). This value is based on the most common freshwater fish species in Norway, brown trout *(Salmo trutta L.)*. In other countries other fish species are more abundant, and a variable limit is necessary to protect most of the aquatic organisms. However, the natural acid neutralising capacity of surface water in areas dominated by granite and gneiss and a thin soil layer can be 20 µeq/l or less, which would produce "negative" critical loads. For these lakes the critical load is set to zero. In areas with little acid rain the probability of fish damages is small even if acid neutralising capacity is close to zero, while in areas with much acid rain fish damage can occur even at this value. Therefore, the minimum buffering capacity was treated as a variable, i.e. as a function of the acid deposition to the lakes, when the Norwegian Institute of Water Research (NIVA) used the Steady State Water Chemistry model to calculate critical loads for surface water in Norway with respect to acidification (Henriksen et. al. 1995a, b, 1999). The model is based on the assumption that sulphate is a completely mobile anion and that the sulpur deposition can be used to indicate the acidifying effect of sulphur. It further assumes that the only acidifying effect of nitrogen deposition is the part that is leached as nitrate in runoff. Most of the surface water area in Southern Norway and parts of Northern Norway have low critical loads; i.e. below 50meq/m^2/year. This means that the annual deposition must be less than 0.80g sulphur/m^2/year to avoid exceeding the critical load. These areas represent areas which are most sensitive to acidic deposition.

NIVA then used the critical load function (Posch et. al. 1997) to calculate exceedance of critical loads for each of the geographical grid cells at the 1990 deposition level (i.e. an average value for the period 1988-1992), and in year 2010 according to the commitments of the Second Sulphur Protocol. Deposition of nitrogen is assumed to be constant from 1990 to 2010. The largest areas with the highest exceedance level can be found in the south and western parts of Southern Norway, but also the eastern part and the west coast of Southern Norway are affected. The critical loads for surface water are also exceeded in the northern tip of Norway, which is caused mainly by sulphur emissions from industrial plants and coal-fired power plants in the neighbouring Kola Peninsula in Russia.

3.2.5 Dose-response functions for exceedance of critical loads and fish damage

A "1000 lake survey" collected data on water chemistry and status of the fish populations from 1005 lakes throughout Norway in 1986. Fish damage was then presented in the same grid system as the critical loads; i.e. a system of grids of 0.5 ° latitude by 1° longitude divided into 16 sub-grids. Additional data for areas poorly covered was collected in 1989 and 1990. (Henriksen et al 1988, 1989, 1999). Water samples were collected from each lake and analysed. Status for all fish species (brown trout, arctic char, perch and pike) was recorded for each lake, based on interviews with representatives of the environmental authorities at the county level. The status of the fish society in each lake was classified using a fish damage index with three categories. Class 1 (unaffected) indicates no reduction in the fish populations, class 2 (reduced) indicates a reduction in the densities of one or more populations, and class 3 (extinct) indicates that all populations are extinct. Naturally thin populations were classified as unaffected. After excluding limed lakes, 697 lakes with acceptable fish information remained

Dose-response functions between exceedance of critical loads and fish status were then estimated for each class. Using a logistic regression model Henriksen et. al. (1995b, 1999) calculated how exceedance of critical loads affects the probability for a fish population to be classified in each of the three damage classes. When the critical load is exceeded the probability of fish damage increases with increasing exceedance.

It is difficult to predict damage class 2, since the curves for damage class 1 and 3 are so close. The reason is that damage class 2 fails to represent a stable state, since the fish population will disappear (i.e, class 3) without a large reduction in deposition. Predicting if a lake has damages to its fish population or not (i.e. damage class 3 or 1) at a given level of critical level exceedance is relativelt easy. These three damage classes formed the basis for the Contingent Valuation survey performed in April 1996. However, in a re-analysis of the data with only two damage classes (damaged and undamaged), Henriksen et al. (1999) were able to identify dose-response functions which could correctly predict 83% and 85% of the undamaged and damaged fish populations respectively.

3.2.6 Economic valuation of fish damages

Since acid rain is one of the largest environmental problems in Norway, the entire Norwegian population was considered to be affected by this change in fish stocks, even though the largest impacts occur in Southern Norway. Thus, a nation-wide survey with in-person interviews of 1,009 Norwegian households was carried out in April 1996.

The stated preference method of contingent valuation was used to estimate the economic value of the increased number of lakes with undamaged fish stocks. The maps in Figure 5, together with the following oral presentation, were used as the scenario description in the CV survey:

> *Map A shows the areas in Norway with fish stocks damaged by acid rain. Acid rain is mainly due to long range transported air pollutants from other European countries. International agreements will reduce emissions of sulphur dioxide and nitrogen oxides, which cause acidification. In spite of reduced emissions the acidification damages to fish stocks have increased over the last decade.*

While we wait for the reductions in emissions to become large enough to reduce the damages, the Norwegian government has started liming lakes and rivers. They are now considering a plan to increase the level of liming, which will reduce acidification damages to fish stock as shown in Map B. We will then get back viable fish stocks in the brown trout lakes and the salmon and sea trout rivers in areas where the stocks are currently reduced or extinct. These areas have changed colour from yellow and red in map A to blue in map B. This new liming program will cost each household in Norway X Kroner annually. The government will issue a new tax earmarked for this liming program.

Is your household willing to pay X Kroner annually to get the increment in areas with viable fish stocks as shown in map B. Remember that this means you have to use less money for other purposes.

This WTP question was part of a larger CV study of respiratory symptoms (from air pollution), acid rain damages to fish stocks and noise from road traffic. Thus, this WTP-question came after WTP-questions about avoiding symptoms like cough, bronchitis, itching eyes, headaches; see Navrud (1997, 2001b). This procedure, should avoid the potential problem of respondents overstating their WTP for a particular environmental problem, when a CV survey focus on only one topic.

Each respondent was asked to answer "yes", "no" or "don't know" to only one amount, and the amount varied from 100 to 1,000 Norwegian Kroner (NOK) (1 NOK = 0.11 Euro). This dichotomous choice WTP question was then followed by an open-ended WTP question:

The liming costs are uncertain. What is the most your household is willing to pay annually to get to the situation shown in map B?

Thus, a national liming program having the same impact as the Second Sulphur Protocol was used instead of ascribing the impacts to the international agreement. Pre-tests of the survey instrument showed that respondents protested against a scenario where they were asked to pay for increments in fish stocks which was the result of reduced sulphur emissions mainly from other European countries (Norway receives about 90% of its sulphur depositions from other countries). Many respondents stated zero WTP because they thought that the countries causing the depositions should pay, rather than because they had no utility from increased fish stocks. Thus, these protest zero-answers will lead to understatement of WTP and aggregated benefits to the Norwegian households of this environmental improvement. This clearly illustrates the problem of valuing transboundary pollution impacts. Even though the national liming program scenario avoided much of this protest behaviour, it is impossible to avoid protesters completely since some people also find it unfair that they should pay for the liming even if their utility from the program is positive. However, this type of protest behaviour is found in most CV studies.

A new tax earmarked for liming was accepted as a fair and realistic payment vehicle both in the pre-test and the final survey. Given the relatively high income tax level in Norway, an increase in general income taxes are known to create protest behaviour in CV surveys. Increased fishing licenses would fail to capture the non-use value, and would also probably understate use value (mainly recreational value of angling). This is because the WTP could be anchored in current costs of fishing licences and express what the respondents thought was a ‘fair’ price of fishing licences rather than their welfare gain from increased fish stocks.

3.2.7 Contingent Valuation of reduced fish damage

In the dichotomous choice WTP question the proportion of yes-answers decreases as the bid amount increase. Thus, 70, 66, 59, 51 and 43 % said "yes" to paying 100, 200, 300, 500 and 1000 NOK respectively. This is as expected, but there is a surprisingly large proportion of yes-answers to the highest amount, i.e. the distribution (survival curve) has a fat right tail. The amounts were selected based on the results from two pre-tests that used open-ended WTP questions. Several studies have found dichotomous choice to give higher mean WTP than open-ended (Boyle et. al. 1996, Brown et. al. 1996 and Ready et. al. 1996). One reason for this discrepancy could be that the respondents use a lower certainty level of paying when answering dichotomous choice compared to open-ended questions. When the certainty level is specified (i.e. asking questions like "Are you 95 % sure you would pay X NOK?") the open ended and dichotomous choice results come together. Thus, we should expect different results from our dichotomous choice and open ended questions where certainty levels are unspecified, which is the usual approach.

Household income and recreational fishing had a significant, positive effect on WTP while a low educational level had a significant, negative effect. The mean WTP calculated from the single and multiple logit models was 705 and 653 NOK/household/year, respectively.

The reported WTP from the OE question ranged from 0 to 10,000 NOK with mean and median values of 367 and 200 NOK/household/year, respectively (Std. = 664). 31 % of the respondents stated zero WTP. Table 3 shows that the two reasons stated most frequently are protest answers. These respondents state zero WTP because they think others should pay or take responsibility for reducing the damages. Thus, as many as 77 % of the zero WTP answers are at least partly protest answers (since respondents are allowed to state more than one reasons for their zero answer). If all these answers are left out, i.e. implicitly assuming that these respondents have a ‘real’ WTP of the average of the other respondents’ mean WTP, increases by 35% to 496 NOK.

Combining the OE and DC results the annual WTP per household seems to be in the range of 367-705 NOK. The annual aggregate benefits for all 1.98 million Norwegian households can then be calculated at 727 - 1,396 million NOK, which is equivalent to 80.0-153.6 million Euros. These benefits can be compared to the saved costs of liming these water courses from fulfilment of the Second Sulphur Protocol of 211 million NOK/year (Henriksen and Hindar 1997; table 4). Thus, the benefit-cost ratio of such a liming project would be 3.45 - 6.62.

3.2.8 The potential for benefit transfers

There are at least two options in selecting a *unit of transfer* of this valuation estimate: i) per tonne of reduced sulphur deposition, measured as critical load exceedance sulphur, and ii) per square kilometre reduced land area of critical load exceedance. Both these options combine the weak and strong sustainability indicators of economic value and critical loads.

Looking at the weight-based unit first, we find that in 1990 145,000 tons of sulphur was deposited in Norway, which is 30,000 tonnes above the critical load. In 2010, with the fulfilment of the Second Sulphur Protocol, the annual deposition in Norway would be reduced to 69,000 tons, of which 10,000 tons exceeds the critical load. Thus, there is a reduction of 20,000 tonnes (66.7 %) in terms of "critical load exceedance" by sulphur. If we assume a linear relationship between the reduced deposition and the economic value, we can calculate the annual economic value per tonne of "critical load exceedance" of sulphur at 4,000-7,680 Euros. The corresponding numbers on a per household basis is 2.0-3.9 mEuros/tonne/year (1 mEuro = 10^{-3} (0.001) Euros). Since the impact caused by 1 tonne of sulphur varies from location to location due to differences in buffering capacity of the soils, and thus the critical loads, the economic value as euro per reduced tonne of critical load exceedance of sulphur is a better measure than reduction in total sulphur deposition for transfer of these estimates to other locations in Europe.

The second area-based unit can be calculated in a similar way. Henriksen and Hindar (1997) found that the area where the critical load for sulphur was exceeded was 80,040 km^2 in 1990, i.e. 25,0 % of the total area. They predict that this area will be 34,550 km^2 (10.8 % of total land area) with the Second Sulphur Protocol in 2010. This is a reduction in area with exceeded critical loads of 45,490 km^2 (56.8 %). Again assuming a linear relationship (i.e. constant economic value per unit of reduced area with exceeded critical loads), we find an economic value of 15,982 - 30,688 Euro/km^2/year. This is equivalent to 8.1 -15.4 mEuro/ km^2/year.

However none of these approaches account for the difference in population density, and thus the number of users and non-users affected by the policy. This will greatly affect the aggregated damages or benefits of a policy.

According to the UN (1998) Europe had 729 million inhabitants in 1997, 374 million lived within the European Union, and 4 million lived in Norway. The land area of Europe is 23 million km^2, out of which the Union covers 3 million km^2, and Norway 0.3 million km^2. The population density of Europe, the Union and Norway is 31.65, 115.38 and 13.5 inhabitants per km^2, respectively. Norway is much more sparsely populated than the average for Europe. Thus, the number of people affected by a change in fish stocks and environmental quality in general, is higher within the Union and Europe as a whole. Therefore, a direct transfer of the Norwegian damage estimate per ton of sulphur would underestimate the damage costs to Europe. Differences in income level, other socio-economic variables, cultural preferences, and institutions in different countries should also be taken into account.

3.2.9 Conclusion

There is large uncertainty inherent in these estimates. The uncertainty stems both from the calculation of impacts (i.e. uncertain dose-response functions) and the economic valuation procedure (i.e. uncertainty of the contingent valuation surveys). To be able to use these estimates to calculate parts of the social benefits of second generation international agreements on emission reductions of SO_2 and NO_X based on critical load, we must assume that the marginal WTP is constant, i.e. independent of the current level of acid deposition, and that Norwegians are representative of all Europeans when it comes to WTP for reducing fish damages. However, this study shows that it is possible to link the critical loads concept with economic valuation. Similar studies in other European countries should be performed to test the validity of these estimates. The possibility of applying this methodological framework to other potential impacts from sulphur and nitrogen depositions, e.g. impacts on forest ecosystems, should be considered.

3.3 Monetary estimates of impacts of eutrophication on use and non-use values

In this section we will review and update monetary estimates of impacts of eutrophication on use values (drinking water, boating, swimming, and recreation fishing) and non-use values.

3.3.1 Summary of European studies

Studies using Contingent Valuation

Atkins, J.P. and D. Burdon (2006): "An Initial Economic Evaluation of Water Quality Improvements in the Randers Fjord, Denmark" (Marine Pollution Bulletin, 53 (1-4): 195-204).

Randers Fjord in Arhus County (Denmark) has experienced increased eutrophication levels as a result of nutrient inputs from sewage wastewater and agricultural runoff. As a consequence there have been toxic algal blooms, a reduction in eelgrass beds and their associated communities, reduced catch rates for recreational fisherman and a reduction in aesthetic values (foaming on shores and turbid water).

The article examines the costs and benefits associated with a reduction in eutrophication of Rangers Fjord. The costs associated with changing agricultural practices and treating urban wastewater (industrial and public service) are reported.

The valuation study was carried out in Denmark. A random sample of 1510 residents of Arhus County (Denmark) was obtained. Postal code population relative to the overall county population weighted the sample. Overall in the sample 63% were male, 68% were between 30 and 60 years old and income ranged from <13, 400 euros/year to > 13, 400 000 euros/year. Respondents participated in a variety of recreational activities. 43% had knowledge of eutrophication while, 36% were aware of or had viewed the effects of eutrophication in Randers Fjord.

Randers Fjord is a 27 km long shallow estuary located in Arthus County on the east coast of Jutland, Denmark. The main industries supported by the Fjord include tourism and recreation. Recreational activities consist primarily of boating, angling, water-sports, camping and bathing. Eutrophication is a problem in the spring and summer as a result of sewage plant wastewater and agricultural runoff.

Current secchi disk depth (measure of water transparency) in Randers Fjord is 1.6 m. This study examines the economic value associated with returning Randers Fjord to pristine condition by implementing an action plan that would reduce agricultural runoff and treat sewage wastewater. In the scenario presented to respondents it is expected that with nutrient reduction programs, water transparency could increase to between 2.5 to 3 m over 10 years. At this depth respondents would be able to see the bottom of the Fjord while participating in recreational activities.

The study focused on the use the contingent valuation (CV) method to estimate public preference for an action plan that would reduce eutrophication in Randers Fjord. The authors employed the open-ended CV method. Respondents were told that in order to implement an action plan that would increase water transparency to between 2 and 3.5 m, an increase in taxes would be required. Respondents were asked to state the maximum amount of tax increase they would pay on a monthly basis for 10 years. Preceding the willingness to pay question respondents were asked about activities related to Randers Fjord and knowledge on eutrophication. Written and pictorial information was provided. The survey concluded with demographic questions. Between October and December 2003, the survey was administered by mail to 1510 respondents for a response rate of 19%. This followed a pre-test involving 66 respondents in September 2003. Data were collected in 2003.

The mean willingness to pay per month over a ten-month period for a program to improve water quality by reducing nutrient input was estimated to 12.02 euros. The range was between 0 and 134 euros with a median of 6.70 euros and a standard deviation of 19.37 euros. Based on Statistics Denmark's estimate of the population of Arhus County (649, 177), an aggregate benefit for improved water quality was estimated at 5.5 million euros per month over 10 years (2003-euros).

Frykblom, P. (1998): "Halved Emissions of Nutrients, What are the Benefits? - A Contingent Valuation Method Survey Applied to Laholm Bay" (Doctorate dissertation, Swedish University of Agricultural Sciences, Uppsala).

The overall aim of this contingent valuation study was to obtain a monetary approximation of the gain in utility as a result of less eutrophication in the Laholm Bay, a coastal area on the West coast of Sweden. Laholm Bay is frequently affected by eutrophication.

A questionnaire was developed and two test surveys were sent out. 90 individuals received a preliminary version of the questionnaire. The response rate of the test surveys was 38 %. After that every fifth individual was surveyed per telephone about his or her opinions regarding the design of the survey. After processing the answers, another revised version was sent to 60 individuals. The response rate this time was 43 percent. The final revised questionnaire was sent to a random sample (from the Swedish census register) of 500 individuals between 18 and 75 years of age living in Båstad, Halmstad and Laholm. The survey was carried out in April and May 1996, following Dillman's (1978) Total Design Method. The net sample consisted of 485 respondents, of whom 327 responded. This gives a response rate of 67 %.

Dichotomous choice was used in the CV study. A regression analysis was conducted where willingness to pay was regressed on the following explanatory variables: sex, age, marital status, children, education, big city, midcity, respondent is a farmer and household income.

The valued good is improved water quality from reduced eutrophication in salt water, 50 percent reduction of the nutrient emissions (nitrogen and phosphorus) in Laholm Bay.

The mean annual individual willingness to pay to reduce nutrient emissions by 50 percent in Laholm Bay is estimated to 747 (1996 Swedish kronor) and the median annual individual willingness to pay is estimated to 244 kronor. Sensitivity tests were done by dropping different categories of rejecters to the willingness to pay question. The results from the sensitivity tests showed that median willingness to pay increased through the entire dropping procedure while the mean both increased and decreased.

Table 1.3.1. Estimation of Total Benefits as Present Values (1996 Swedish kronor)

Mean annual WTP	Total annual WTP (mill SEK)	Median annual WTP	Total annual WTP (mill SEK)
747	90	244	29

Soderqvist, T. and H. Scharin (2000): "The Regional Willingness to Pay for a Reduced Eutrophication in the Stockholm Archipelago" (Discussion paper no. 128, Beijer International Institute of Ecological Economics, The Royal Swedish Academy of Sciences).

This study estimated the willingness to pay to reduce eutrophication in the Stockholm archipelago. The environmental goods and services valued were eutrophication of coastal areas from agricultural runoff; expressed by increase in sight depth of water from 1 metre currently to 2 metres in 10 years.

Data were collected from mail survey of a sample of adult population of Stockholm and Uppsala counties in 1998. The sample consisted of 4,000 individuals; the survey was used to collect data about people's recreational behaviour.

The method used was Contingent valuation with open ended and payment card, respectively. Ordinary least squares procedure was used for model estimation. The willingness to pay was estimated as a function of socio-economic characteristics, recreation behaviour and place of residence.

Characteristics of the respondents and non-respondents were analyzed. Analysis of the non-respondents showed that females were more willing to respond to the survey, place of residence and age statistically differed among respondents and non-respondents. The non-respondents to the questionnaire were assumed to have a zero willingness to pay. The non respondents to the open ended portion of the questionnaire were assumed to have willingness to pay equal to the mean of the respondents. Protest responses were categorized as zero willingness to pay. Ordinary least squares procedure was used for model estimation. The mean monthly willingness to pay ranged from 37 to 60 Swedish Krones, per adult. Multiplying the monthly value by the total study population yielded a value of 506 to 842 million (1998,1999) Swedish Krones annually. An area of further research identified include linking water flow with nutrient loading to arrive at optimum abatement level.

Mean monthly willingness to pay ranged from 37 to 60 (1998,1999) SEK per adult, depending on the scenarios. Multiplying this by the total study population yielded a value of 506 to 842 million (1998,1999) SEK annually.

Soderquist, T. (1996): "Contingent Valuation of a Less Eutrophicated Baltic Sea"

(Beijer Discussion Paper Series No. 88 Beijer International Institute of Ecological Economics, The Royal Academy of Sciences, Stockholm.ISSN 1102-4941).

A contingent valuation-dichotomous choice mail survey of a random sample of 679 Swedes (net sample) was carried out to estimate their Willingness-to-pay (WTP) as an extra environmental tax (for households, firms etc) introduced in all Baltic Sea countries and ear-marked to a large-scale international action plan that would reduce nitrogen and phosphorous load to the Baltic Sea by 50 %, which in 20 years will reduce eutrophication to a level the Baltic Sea can sustain.

The environmental goods and services valued were ecological functions, extractive uses, non extractive uses, and passive uses connected to water quality / eutrophication level of the Baltic Sea. The good valued was improved water quality in the Baltic Sea from a plan to reduce euthropication through a 50 % reduction in nitrogen and phosphorous load accomplished by a cost-effective allocation of reductions among the nine countries around the Baltic Sea (Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Russia and Sweden. In 2o years the plan will decrease the eutrophication to a level that the Baltic Sea sustains (equivalent to the eutrophication level in the 1960s).

A conservative mean WTP estimate (assuming all protesters and non-respondents having zero WTP) is 3,300 SEK/person/year. A similar contingent Valuation survey of a random sample of 600 Polish adults gave a conservative estimate of 10 % of the Swedish WTP.

Both non-parametric (Ayer) and parametric (maximum likelihood estimation (MLE), probit and logit) methods were used to estimate mean and median WTP. The independent variables that had a significant effect on the probability of voting for the eutrophication reduction plan were the size of the bid (eight bids ranging from 0 to 25,000 SEK), annual gross personal income, and whether they considered eutrophication of the Baltic Sea to be a problem. The last two had a positive impact, and the bid (stated cost of the plan) had a negative impact. Other socio-demographic variables did not have significant effects.

Mean WTP/person/year in 20 years is 3,300 - 5,900 SEK. The lowest estimate assumes that protesters and non-respondents have zero WTP. Among those that had a positive or a real zero WTP (excluding protesters) the mean WTP was 7,000 SEK/person/year. Mean WTP from a similar mail questionnaire to a random sample of 600 Polish adults was 300 and 600 SEK/person/year assuming zero WTP of protesters, and zero WTP of protesters and non-respondents,respectively. 1 US$ = 8.80 SEK. Discount rate used in estimations was 7 %.

Adjusting the Swedish and Polish estimates with GDP/capita (at purchasing power parity) a benefit transfer to respectively market and transition economies around the Baltic Sea result in aggregate annual benefits of the action plan of 31,527 million SEK for the nine Baltic Sea countries. This is considered to be a conservative estimate. Present Value of the aggregated national benefits was calculated at 630 billion SEK (20 year time horizon and 7 % discount rate). Annual aggregate costs of the action plan were estimated at 31,070 million SEK.

Magnussen, K. and S. Navrud (1992) "Valuation of Reduced Pollution to the North Sea (Verdsetting av redusert forurensning til Nordsjøen)" (Norwegian Agricultural Economics Research Institute (NILF)).

Due to international agreements, Norway has to reduce the pollution of the watercourses leading to the North Sea. The plan to reduce this pollution is referred to as the NSP plan. The study aims to find the willingness to pay for this plan among the inhabitants of Norway. In order to do this, contingent valuation was used. Two studies are included in the report, one of the total population of Norway, the other for the county of Østfold. The primary study (whole of Norway) aimed at determining the accumulated WTP for the NSP, whilst the other study was of more methodological nature. Dichotomous choice and open ended bidding were used for different sub-samples of the main study.

Approximately 1000 respondents were interviewed person-to-person in their homes.

The NSP Plan meant 50 per cent reduction of nitrogen and phosphorus leaching to the North Sea. In order to link these physical indicators to economic valuation , natural research scientists were involved in “translating” these reductions to effects on water quality, measured as sight deep, oxygen percentage, algae growth etc (dose-response-functions), and then to what these water quality levels imply for users of water for angling, swimming, drinking water (for fresh water bodies) etc. These effects for water quality and usage were illustrated with the aid of maps, cards and pictures.

The mean WTP varied with method and with the first bid. The mean WTP per household per year for the whole country was 2110 NOK (1991) for use and non-use values.

Le Goffe, P. (1995):"The Benefits of Improvements in Coastal Water Quality: A Contingent Approach" (Journal of Environmental Management 45, no. 4, 305-317).

This study estimated willingness to pay (WTP) for improved water quality in Brest Harbor, France using the payment card contingent valuation method.

The sources of water pollution are animal faeces, fertilisers, and domestic and industrial waste. Both extractive uses, non extractive uses, and passive uses were valued in harbour water quality. The baseline level of provision was the current water quality in Brest Harbour. The two alternate levels of provision were improved water quality to the degree that bathing and shellfish consumption would be safe, and prevention of asphyxiation of harbour waters due to eutrophication.

The survey was administered at five sites, four of which were chosen to represent the recreational uses of the Brest Bay: beach, sandbank use, sailing and coastal walking. The fifth site, Brest Botanical Gardens, was chosen to include nonusers in the sample. The questionnaire used in the study was pre-tested. Personal interviews were conducted on-site with 607 individuals between June and September of 1993. The interviews were conducted by twelve people, students in Geography at the University of Brest. They received one day of training. On average, two people were approached to obtain one interview, (50 percent participation). Photographs of water quality conditions were used to help survey participants respond to the contingent valuation questions. Contingent valuation with payment card was used.

Depending on the model, estimates of mean annual household WTP for water quality improvements that ensure safe bathing and shellfish consumption ranged from FF 214 to FF 218 (1993 French francs), or \$37.23 to \$37.93 (1993 U.S. dollars). Estimates of mean annual household WTP for the prevention of eutrophication were between FF 158 and FF 173, or \$27.49 to \$30.10 (1993 U.S. dollars).

Willingness to pay (WTP) for risk-free bathing and shellfish consumption in Brest Harbour was modelled as a function of gender, age, income, education, profession, respondents' priority of Brest Natural Harbour preservation, whether respondent observed coloured water in the harbour, annual number of visits to the harbour, whether respondent sailed in the harbour, whether respondent went diving in the harbour, and availability of seaside substitute sites. WTP to prevent eutrophication was modelled as a function of the same independent variables used in the previous model, and the risk of asphyxiation of harbour waters.

Mean annual household willingness to pay (WTP) for improved water quality in Brest Harbour was calculated for the sample directly from the responses to the contingent valuation questions. Mean annual household WTP was also estimated for the population by evaluating the equations at the sample mean values for the independent variables. Annual household WTP of the population for safe bathing and shellfish consumption and prevention of eutrophication are presented in Table 1.3.2.

*Table 1.3.2. Annual Household Willingness to Pay (WTP) for Improved Water Quality in Brest Harbour (1993 French Francs)**

	WTP- Model 1**	WTP- Model 2***	WTP- Model 3^	Mean WTP^^
Safe Bathing and Shellfish Consumption	214	215	215	218
Prevention of Eutrophication	162	160	158	173

Notes:

*On June 30, 1993, 5.7476 French francs was equivalent to one U.S. dollar (Pacific Exchange Rate Service, http://pacific.commerce.ubc.ca).

**In Model 1 the variables used to explain variation in WTP were gender, age, income, respondents' priority of Brest Harbour preservation, whether respondents observed coloured waters in the harbour, annual number of visits to the harbour, and availability of seaside substitute sites.

***In Model 2 the variables used to explain variation in WTP included all of the variables used in Model 1 except age, plus education.

^In Model 3 the variables used to explain variation in WTP included all of the variables used in Model 1, except age, plus whether respondent sailed at the harbour, and whether respondent went diving in the harbour.

^^Mean value for the sample, excluding protest bids.

3.3.2 European Contingent Valuation and Travel Cost studies

Alice Brunel (1996): Evaluation des bénéfices liés à la réalisation d'une réserve d'eau potable à partir de l'Erdre et évaluation des bénéfices touristiques liés à l'amélioration de la qualité de l'eau de l'Erdre (Université des Sciences Sociales - Toulouse I Mémoire de DEA - Economie de l'environnement, des ressources naturelles, de l'énergie et de l'agriculture. Part I).

The Erdre river, the water quality forbidding any kind of uses, was chosen to illustrate the CV method by determining the benefits that could be obtained from the construction of a drinking water storage destined to the area of Nantes, which implies a significant improvement of the water quality.

The sample was not representative of the population of Nantes due to the fact that only those who wanted to have answered the questionnaire. They are principally women, workers, with a monthly income situated between 800 and 2200 euros. The way they accessed to drinking water was precised and also the amount of their water invoice.

30% of the people interrogated have announced a null willingness to pay. Protest zeros have been distinguished from the real zeros, which at the end represents only 6%. The influence of different criteria on the answer given as a willingness to pay was analysed using the Student t-Test based on the average equality. The correlation coefficient was used to select the most pertinent variables for a modelling of the willingness to pay, for which three ways were compared (linear, log-linear and Tobit). The Tobit model was led apart, the log-linear one seems to under-estimate the WTP, when the linear one predicts it precisely.

The contingent method has given a willingness to pay of 32.7 euros per year and household to retrieve the benefit of the construction of a drinking water storage. This amount corresponds with a 13.4 % augmentation of the water invoice if this one is around 240 euros. Considering the fact that the sample used was not representative of the people concerned, a costs-benefits analysed can not be launched.

Alice Brunel (1996): Evaluation des bénéfices liés à la réalisation d'une réserve d'eau potable à partir de l'Erdre et évaluation des bénéfices touristiques liés à l'amélioration de la qualité de l'eau de l'Erdre (Université des Sciences Sociales - Toulouse I Mémoire de DEA - Economie de l'environnement, des ressources naturelles, de l'énergie et de l'agriculture - Part II).

This study seeks to analyse the empirical valuation techniques that are the contingent valuation method and the travel cost one. The Erdre river, the water quality forbidding any kind of uses, was chosen to illustrate those two models by determining the tourist benefits that could be obtained from a significant improvement of the water quality and most particularly the benefits linked to the swimming and recreational fishing activities.

The two questionnaires used in the study were almost the same, even if the second one was more oriented to fit with the recreational fishing activity. The frequentation characteristics of the Erdre were detailed (number of visits/year (on average: 80), motivation, activities practised, time spent on the spot per visit.)

Considering the bathing activity, 35% of the people interrogated have announced a null willingness to pay. Protest zeros have been distinguished from the real ones, which at the end represents only 13%. In the fishermen case, half of them refused to express their opinion as a protestation against the payment support proposed (as a matter of fact, the augmentation of the fish-card price).

In both cases, the influence of different criteria on the answer given as a willingness to pay was analysed using the Student Test based on the average equality. The correlation coefficient was used to select the most pertinent variables for a modelling of the willingness to pay, for which three ways was compared (linear, log-linear and Tobit). The log-linear model seems to give less probable results, so only those given by the linear and Tobit one were kept to finalise the estimation.

The results were then compared with those obtained using the travel cost method. This one was applied on the both samples gathered (swimmers and fishermen) and aimed to estimate the frequenting demand of the Erdre riversides. The way the people questioned travel and the distance made was mentioned, but the time was not taken into account. The effects of the different variables that may influence the frequency with which people come were analysed before to build a modelling of the frequenting demand.

At the end the variation of well-being induced by an improvement of the water quality was analysed through the variation surplus estimated per individual.

In the first place, the study shows that the potential swimmers are ready to pay between 15 and 21 euros (according to the model used) per person and year to join a club which would deal with the management, the set up, the supervision and the maintenance of green beaches on the riversides.

The slight number of fishermen questioned does not allowed to conclude on their willingness to pay for an improvement of the water quality permitting to fish.

The travel cost approach has allowed the researchers to estimate the actual surplus at 192 euros per person and per year. If the water quality improves significantly, this surplus could rise to 209 euros or 17 euros per visit. The results obtained are almost the same as those with the contingent method and even if the sample was reduced, the two methods appeared to be quite coherent and to lead to an equal evaluation of the well-being variation bounded to the water quality of the Erdre.

3.3.3 European Travel Cost studies

Soutukorva, Å (2005): "The Value of Improved Water Quality - A Random Utility Model of Recreation in the Stockholm Archipelago" (Beijer Discussion Paper Series No. 135, Beijer International Institute of Ecological Economics, the Royal Swedish Academy of Sciences).

The purpose of this travel cost study was to estimate the recreational benefits of an improved water quality in the Stockholm archipelago, Sweden. The archipelago is seriously affected by eutrophication.

The goods valued were recreational benefits from improved water quality in the Stockholm archipelago, 1-metre improvement of mean sight depth.

The data used in this travel cost study was collected by the Beijer Institute of Ecological Economics in 1998 and 1999. 4000 mail questionnaires were sent out to the adult population of Stockholm and Uppsala counties in 1998, with a response rate of 47.2 percent after three reminders. Another 1500 questionnaires were sent out in 1999, with a response rate of 60 percent.

The aggregated consumer surplus for a 1-metre improvement of mean sight depth in the archipelago was estimated to 85-273 million (1998 Swedish Kronor) for the adult population of Stockholm and Uppsala counties. The author concludes that welfare measures of improved water quality depend on what determinants of recreational demand are included in the model and how travel time is treated.

3.3.4 European Benefit Transfer studies

Horton, B., and J. Fisher (2004): "The 4th Periodic Review of the UK Water Industry: A Large-Scale Practical Application of Environmental Cost-Benefit Analysis" (Paper presented at the Applied Environmental Economics Conference, the Royal Society, London, United Kingdom).

A large scale benefits transfer was used to assess the environmental benefits and costs of nearly 500 potential water quality and water resource improvement schemes impacting upon groundwater, river ecosystems, freshwater fisheries, habitats, bathing waters, shell fish waters, low flow alleviation in rivers, and local priority schemes (e.g. eutrophication) in England and Wales. Various types of marginal environmental change were valued.

The purpose of this benefits transfer study was to assess the environmental costs and benefits of nearly 500 environmental water quality and water resource improvement schemes in England and Wales. For the benefits transfer a review was conducted of all available and relevant studies, with the best and most appropriate used to suggest a value for each environmental attribute that is applicable to different contexts, and to derive average estimates of the number of people who would gain from a particular type of benefit in a particular type of location. Water companies provided estimates of the financial and environmental costs of the schemes. The results were allocated between six different categories depending on their Benefit-Cost ratios and local priority. 62% (272) of the schemes were recommended for implementation, representing 38% of the total costs (645 million 2003 British Pounds) and 80% of the total benefits (1,154 million 2003 British Pounds). Non-use benefits dominated the calculated benefits for water quality and water resources schemes.

The evaluated schemes were assigned to one of six categories based on their benefit cost (BC) ratio and factors of particular importance to local stakeholders: 1) Schemes of high regional priority but which had a BC ratio of less than 1.2; 2) Monetary benefits at least twice as great as costs; 3) Monetary benefits less than double but greater than 1.2 times costs; 4) Monetary benefits less than 1.2 times but greater than 0.8 times costs; 5) Monetary benefits less than 0.8 times costs; 6) Monetary benefits greater than 1.2 times costs but which the Environment Agency recommended should be deferred. Non-use benefits dominated the calculated benefits.

Turner, R.K., S. Georgiou, I-M. Gren, F. Wulff, S. Barrett, T. Soderqvist, I.J. Bateman, C. Folke, S. Langaas, T. Zylicz, K-G. Maler, and A. Markowska: "Managing Nutrient Fluxes and Pollution in the Baltic: An Interdisciplinary Simulation Study" 1999. (Ecological Economics 30: 333-352)

Eutrophication is now pronounced in the Gulf of Finland, Gulf of Riga and in limited coastal areas in the eastern, southern and south-western Baltic Sea area. There have been many toxic algae outbreaks. The purpose of the study was to report the results of a study into the costs and benefits of eutrophication reduction in the Baltic Sea.

A total of 14 empirical valuation studies in three countries - Poland, Sweden, and Lithuania - were carried out to look at benefit estimation issues. One of the studies was a contingent valuation study focusing on Baltic Sea use and non-use values in Sweden. A mail questionnaire was sent out to 600 randomly selected adult Swedes. The response rate turned out to be about 60%. A similar study was carried out in Poland and the response rate was about 50%.

Table 1.3.3 shows estimates of aggregate benefits for the total economic value of a Baltic Sea nutrient reduction strategy. Data from the Polish and Swedish mail surveys were used since they were both concerned with total economic value, and they contain the same valuation scenario.

Table 1.3.3 Basin-wide Benefit Estimates for the Total Economic Value of a Baltic Sea Nutrient Reduction Strategy in SEK

(a)Annual Willingness to Pay per Person, (b)National Willingness to Pay year one (MSEK), (c)National Willingness to Pay, Present Value (MSEK); d) National Willingness to Pay, Present Value per Year (MSEK)

	(a)	(b)	(c)	(d)
Transition Economies:	700 (355)	790 (401)	8,369 (4,248)	418 (212)
Estonia	569 (284)	1,100 (549)	11,653 (5,816)	583 (291)
Latvia	665(337)	1,743 (883)	18,465 (9,355)	923 (468)
Lithuania	840 (426)	21,958 (11,136)	232,623 (117,974)	11,631 (5,899)
Poland	909 (461)	6,585 (3,340)	69,761 (35,384)	3,488 (1,769)
Market Economies:				
Denmark	6,770 (3,790)	23,365 (13,080)	247,529 (138,570)	12,376 (6,929)
Finland	5,430 (3,040)	20,387 (11,414)	215,980 (120,920)	10,799 (6,046)
Germany	6,500 (3,640)	15,800 (8,848)	167,385 (93,736)	8,369 (4,687)
Sweden	5,900 (3,300)	39,122 (21,882)	414,458 (231,818)	20,723 (11,591)

Notes:

Figures in brackets are for benefit figures which assume zero willingness to pay of non-respondents.

Countries whose national jurisdiction lies in the southern basins are revealed as the biggest net economic gainers from the abatement strategy. Net benefits of reducing the nutrient load to the Baltic Sea by 505 ranges from 15,423 SEK/year for Sweden to minus1,523 SEK/year for Lithuania. This calls for a differentiated approach with abatement measures being concentrated on nutrient loads entering the Baltic Proper from surrounding southern sub-drainage basins. Although there are a range of feasible individual Nutrients reduction (N-reduction) and Pollution reduction (P-reduction) measures available, results indicate that the simultaneous reduction of both N and P loadings into the Baltic is more environmentally effective as well as cost effective. The results also indicate that the greatest environmental and economic net benefits are to be gained by an abatement policy that is targeted on areas which lack treatment works of an acceptable standard, rather than on making further improvements to treatment facilities that already provide a relatively high standard of effluent treatment.

3.3.5 European Meta-analysis

Hokby, S. and T. Soderqvist (2003): "Elasticities of Demand and Willingness to Pay for Environmental Services in Sweden"(Environmental and Resource Economics 26, 361-383).

The purpose of this study was to determine the price and income elasticities for the demand for eutrophication reduction. The authors used willingness to pay data from five studies which undertook contingent valuation surveys to determine the willingness to pay for eutrophication reduction in the Baltic Sea. The authors merged the data from the five studies into one data set, and modelled willingness to pay as a function of income, percent reduction in Nitrogen, whether the respondent answered a dichotomous choice or an open ended willingness to pay question, and the suggested payment (price). The authors used the coefficients on the explanatory variables to estimate elasticities. They found that 1% increase in income would result in a 0.6% to 1.3% increase in demand for eutrophication reduction, and a 1% increase in the price of eutrophication reduction would result in a 1.6% to 2.1% decrease in demand for nitrogen reduction.

Markowska, A. and T. Zylicx (1999): "Costing an International Public Good: The Case of the Baltic Sea” (Ecological Economics 30, 301-316).

The purpose of this study was to allocate abatement tasks among Baltic Sea countries in order to achieve a 50% reduction in nutrient input to the Sea. The authors used contingent valuation survey results for Poland, Lithuania, and Sweden which elicited the willingness to pay among residents of these countries for eutrophication abatement to extrapolate to all Baltic Sea countries.

The results from these studies were extrapolated to the rest of the Baltic Sea countries, and an optimization model was used to allocate an appropriate abatement scheme among the countries. They do this using two assumptions: that the three countries can represent different groups of Baltic Sea countries; and that mean willingness to pay is proportional to GDP in the countries. The authors used the extrapolated willingness to pay values in an optimization model to estimate the appropriate sharing of abatement costs among the countries.

The authors estimated that Finland, Sweden, Denmark, and Russia were net payers, while Germany, Lithuania, Latvia, Poland, and Estonia would receive compensation.

Mean willingness to pay for nutrient abatement among the Baltic countries was a function of the results of contingent valuation surveys performed in three of the ten countries (Poland, Sweden, and Lithuania), and the GDP of each country. Table 1.3.4 shows the mean per capita willingness to pay for nutrient abatement in each of the Baltic Countries.

Table 1.3.4 Mean Per-Capita Willingness to Pay for Nutrient Abatement in Each Baltic Country

Mean Per-Capita Willingness to Pay	
Finland	232
Sweden	252
Denmark	289
Germany	278
Poland	56
Lithuania	28
Latvia	24
Estonia	29
Russia	38
Drainage Basin	110

Notes:Willingness to pay estimates are reported in U.S. dollars.

3.3.6 Discussion of eutrophication studies

- Overview of locations

Many of the European surveys on eutrophification seem to be concentrated to the Baltic Sea, Sweden and Norway. There are also studies from other European countries. Eutrophication of fresh and salt water is a big problem in most European countries. Mainly coastal (salt water) locations are valued in the studies surveyed here. But there are also valuation of eutrophication of fresh water, for instance in Norway where water quality was valued in two lakes according to the scheme in Magnussen and Navrud (1992) cited above, and then used for benefit transfer, recorded in Bergland et al. cited later in this paper.

- Benefits included

The surveys on eutrophification focus on use and non-use values, often water quality improvement is valued as a bundle of goods, however, a few studies focus on drinking water, recreation use etc. Of course, the travel cost studies give use-values of the users only. It is difficult from the studies in the literature overview to disaggregate the total economic value into use and non-use values respectively.

- Important environmental stressors

The environmental stressors causing the environmental problems are mainly a mixture of man-made pollutions from sewage, agriculture and industry.

- Availability - methodology – possibilities for benefit transfer

There are several studies available, especially from northern part of Europe on this topic. There are also several examples of benefit transfer studies and meta-analysis. Linking of physical indicators and economic valuation is a major challenge for benefit transfer for this topic as well as the others discussed.

It should be noted that most of the eutrophication studies were carried out in the early 1990s, and the methodology and values reflect this fact. With the EU “Water Framework Directive”, the interest for water quality, eutrophication and valuation has risen again lately and new valuation studies in several European countries will be carried out under EC´s 6th Framework Program for Research in the project “Aquamoney”; see http://www.aquamoney.ecologic-events.de/

3.4 Monetary estimates of landscape aesthetic impacts of renewable energy - Wind

3.4.1 Summary of European studies

Studies using Contingent Valuation

Scherrer S. (2003): "Les dommages visuels et sonores causés par les éoliennes : une évaluation par le consentement à payer des ménages dans le cas des éoliennes de Sigean"(Working paper, Direction of economic studies and environmental valuation, French Ministry of Environment).

The purpose of this study was to evaluate the costs of visual effects and noise from wind turbines in Sigean (Languedoc-Roussillon region). Using contingent valuation, the study assessed the willingness to pay for avoiding visual and noise pollutions caused by the wind turbines and willingness to accept for being compensated: both the acceptance rate for paying or being compensated and the amount of money (open-ended method) that would be paid or received, are estimated.

The contingent valuation survey was conducted in the autumn 2001 (between October 27th and November the third) on people older than 18 living in Languedoc-Roussillon region. It was a phone survey, realised on a representative sample of 2000 persons, living in towns close to the wind turbines (15-20 kilometres around), selected on the quotas method (sex, age, and profession). Only people who declared (during the phone survey) to have seen the wind turbines (close or far from them) were questioned about the inconvenience they could feel because of the presence of wind turbines and were then presented contingent valuation scenarios.

The questionnaire elicited information about the feeling about the environmental damages caused by the presence of wind turbines (visual damages, noise, and other pollutions), the determinants of this feeling (distance from the wind turbines,...), willingness to pay (in terms of donations to a fund) for avoiding visual and noise impacts due to wind turbines (by dismantling wind turbines or locating them off-shore), and willingness to accept for being compensated (by an annual decrease of local taxes) for visual and noise impacts.

When excluding the protest zeros, the mean willingness to pay to avoid noise and visual effects amounted to 78 French francs (2001) when the scenario proposed is to dismantle the existing wind turbines against 369 French francs (2001) when the scenario proposed is to locate them off-shore.

The acceptance rate for paying in order to avoid visual and noise impacts caused by wind turbines or the acceptance rate for being compensated for these impacts, and the level of the willingness to pay or to accept were modelled as a function of the age, sex, profession, family income, number of children, environmental sensitivity, general housing specifications (collective/particular, distance from the wind turbines, view on wind turbines), opinions of people about the wind turbines and their pollutions. According to the scenario (financial compensation for visual and noise pollutions, financial contribution for dismantling wind turbines, financial contribution for locating them off-shore), not all explanatory variables were significant. The coefficients were estimated in a Tobit model, using the Heckman method, integrating or not the protest zeros, according to the versions of the model.

Table 1.4.1 shows the mean Willingness to Accept for being compensated for noise and visual effects of the wind turbines. It is higher when including only people who accepted the financial compensation. Table 1.4.2 shows the mean Willingness to Pay to avoid noise and visual effects.

Table 1.4.1 Mean Willingness to Accept for being compensated for noise and visual effects

	French Francs (2001)
Strictly positive amounts for people accepting financial compensation by a decrease of local taxes (1):	3186
All persons being questioned :	463

Notes:

(1) People who accepted financial compensation by a decrease of local taxes correspond to 11.1 % of people being questioned.

Table 1.4.2 Mean Willingness to Pay for avoiding noise and visual effects caused by the wind turbines (French Francs, 2001)

	Dismantling wind turbines (1)	Off-shore implantation (2)
Strictly positive amounts	1730	1175
Including protest zeros	78	369
Excluding protest zeros	94.3	429
Protest zeros replaced by mean WTP strictly positive	378	533

Notes:

(1)The acceptance rate for paying to dismantle wind turbines is 3.3%. (2) The acceptance rate for paying for the off-shore implantation is 24.5%.

Axelsen, L.K. (2003): "Environmental Accounting for Wind Energy - Miljøregnskap for vindkraft" (Norwegian University of Life Sciences (UMB)).

The study is a contingent valuation study of the willingness to pay for having wind power as the only electricity source. This is compared to import of coal and nuclear power, as well as more development of hydroelectricity. The respondents were presented to 4 scenarios: 1) WTP for electricity supply from wind power only, this means that 500 windmills with a total production of 3 TWh would be built (this is but a fraction of Norway's total production), 2) WTP for having all these windmills erected offshore, 3) WTP for having a local wind power production, providing the local energy demand, 18 windmills would have to be built, 4) Local scenario with the electricity cables laid underground. The WTP question was asked in an open ended way, and the payment vehicle used was increased electricity bill.

The sample, of a total of 126 persons, was taken from the following areas: Haramsøy, Flemsøy, Brattvåg, Hildre and Skjelten. The population of the area is positive towards wind power, and there is a wind farm on a neighbouring island. Half of the respondents are likely to see the windmills from their homes, if there is a development of a wind farm on the islands of Haramsøy and Flemsøy.

Table 1.4.3 Mean household WTP for having wind power as only electricity source (NOK, 2002)

	Mean WTP
1) National scenario	1425
2) National scenario, offshore	1444
3) Local scenario	1263
4) Local scenario, underground cables	1397

Regression analysis was performed, and the following correlations found: the only variable that was significant was whether the respondent would see the built windmills from his/her home. If this was the case, it would lower the WTP.

Gjøsund, H. C. S. (2003): "Environmental Effects of Windmills in Sandøy, a Contingent Valuation Study - Miljøkonsekvenser av vindkraft i Sandøy kommune, en verdsettingsstudie" (Norwegian University of Life Sciences (UMB)).

The study object is the small island of Harøy, in north-western Norway, on which 5 wind turbines are already operating. The island might have 12 more turbines built in the future, and the study explores the sentiments of the inhabitants of the island in relation to the possible enlargement of the park.

The study utilises the contingent valuation method for this purpose, and five scenarios were constructed:

1) WTP to ensure that all electricity used by the household comes from wind power only

2) WTP to avoid enlargement of wind farm from 5 to 17 turbines (and to import electricity from mainland)

3) WTP for island to get self sufficient on electricity (and thus enlarge the wind farm)

4) WTP to build the turbines offshore instead of on the island

5) WTA for enlargement of wind park (from 5 to 17 turbines)

The interviews were conducted person to person, and the CV was open ended. The payment vehicle was an increase in the electricity tax.

The whole municipality, of which Harøy is part, has a population of 1305 people. The sample size was 69 households. The study showed that the island's inhabitants were largely positive to wind power on the island, as the WTP in several of the scenarios where the island gets more turbines is positive.

The most attractive scenario was that of #3, in which the wind park is enlarged, and the island gets self sufficient in terms of electricity production. The mean household WTP for this scenario was 859 NOK (2003). The number of zero bidders ranged from 42 (62%) to 54 (78%) in the different scenarios.

Table 1.4.4 Mean household WTP/WTA for different scenarios (NOK, 2003)

1) WTP for electricity only from wind power	837
2) WTP for avoidance of enlargement of wind park	236
3) WTP for making island self sufficient on energy supply	859
4) WTP for sea placement of the new windmills	630
5) WTA enlargement of wind park from 5 to 17 turbines	1907

An extensive regression analysis was performed, with multiple variables. The WTP to avoid an extension of the wind park from 5 to 17 windmills increased significantly with the variable "aggrieved", and this variable again correlated to the degree of exposure to the windmills, in terms of noise and visibility.

Nordahl, E. (2000): "A Contingent Valuation Study of Environmental Impacts of Windmill Development of Smøla - Miljøkostnader av vindkraftutbygging på Smøla"(Norwegian University of Life Sciences (UMB)).

The study explores the economic values of a proposed wind park project on the island of Smøla, in western Norway. The study is a contingent valuation (CV) study, and both willingness to pay (WTP) and willingness to accept (WTA) methods were used. The study used in-person interviews with representatives for 100 households on the island. The survey was conducted with payment cards, as well as cards depicting a visualisation of windmills in the scenery. The payment vehicle used was increase in tax on electricity.

100 households were included in the sample, out of a total population on the island of 993. The sample was representative of the population of the island, in demographical facts.

Three scenarios were explored: 1) no wind park, 2) wind park with electrical cables above the ground, 3) wind park with electrical cables in the ground.

At the time of the study, the proposal for the wind park was ranging between 20 and 75 windmills.

Table 1.4.5 WTP/WTA for different scenarios per household per year (NOK, 1999).

WTP to avoid cables above ground	689
WTP to avoid wind park (interval)	271-742
WTA wind park	887

"Real" zero bids were 31.

The following variables were used in the regression analysis: 1)whether or not the respondent has seen windmills in landscape before, 2)knowledge of the plans for the island, 3)sentiment in relation to wind mills (positive or negative), 4)sentiment to electrical cables above ground, 5)whether or not the respondent was present at the information meeting on the wind park plans, 6) whether or not the respondent is an active user of the area (recreation), 7)age, 8) whether or not a member of the households has ownership rights in the area, 9)income, 10)gender, 11)sentiment to tax.

Studies using CV and the Hedonic Price (HP) method:

Jordal-Jørgensen, J. (1995): "Samfundsmæssig værdi af vindkraft. Delrapport: Visuelle effekter og støj fra vindmøller - kvantificering og værdisætning." (Social Costs of Wind Power: Partial Report of Visual Impacts and Noise from Windmills) (Institute for Local Government Studies (AKF), Copenhagen, Denmark).

The purpose of this study was to evaluate the costs of visual effects and noise from windmills in Denmark.

Three studies were carried out: i) a contingent valuation study; ii) a hedonic pricing study and iii) a study of the number of households affected by windmill-installations.

The contingent valuation study used three scenarios. Scenario 1 involved one more windmill (35 metres * 35 metres) close to the house. Scenario 2 replaced the existing windmills in the windfarm close to the house with larger mills (35metres * 35 metres). Scenario 3 moved the windmills away from the house to uninhabited areas. The hedonic price study estimated the effects on property prices of having windmills close to the house, as compared to not having windmills.

A random sample of 102 of 1,931 windmill-plants was drawn from Denmark's Windmill Association statistics. The contingent valuation study was based on personal interviews.

The sample size was 395 (of 850 addresses), of which 58 were not available. Of the 281 who participated, 44 experienced a nuisance. Eighteen of these reported a willingness to pay.

In the hedonic price study, property prices of 74 properties situated close to windmills were studied. Sixteen of these had windmills close to the house at the moment of sale. The windmills were grouped into single mills (1-2), clusters (3-5) and wind farms (>5). The windmills were situated approximately 475 meters from the houses.

For the contingent valuation study, the willingness to pay (WTP) for moving the windmills was 1236 Danish Kroner (1995) per household per annum among those 18 reporting a WTP. For the other 26 reporting nuisance, and thus having motives of positive WTP, but not reporting WTP, it was assumed their WTP was between 684 and 885 Danish Kroner. This leads to a WTP of 982 Danish Kroner per household per annum for the 13 percent of the interviewees experiencing nuisance. This corresponds to 0.0004 Danish Kroner/Kilowatt Hour. The hedonic study showed that property values are 94.147 Danish Kroner lower close to a wind farm than other houses, when other characteristics of the house are similar. This corresponds to 0.0098 Danish Kroner/Kilowatt Hour. However, these estimates are not statistically significant, because of the small data set.

Table 1.4.6 reports the willingness-to-pay per year for windmill nuisance grouped by respondent type such as voting to move or not to move, and whether they want to pay more and are reporting an amount. Table 1.4.7 presents the effect of windmills on property values grouped by type of mill installation (single, cluster or wind farm) and effect, (per installation, per windmill and per windmill per year). In the contingent valuation study, the costs of the nuisance by the windmills were estimated at 0.0004 Danish Kroner/Kilowatt Hour. For single mills, the costs were estimated at 0.0011 Danish Kroner/Kilowatt Hour, and for clusters the costs were estimated at 0.0009 Danish Kroner/Kilowatt Hour and for wind parks the costs were 0.0002 Danish Kroner/Kilowatt Hour. For the Hedonic price study, the costs of the nuisance by the windmills were estimated at 0.0098 Danish Kroner/Kilowatt Hour.

Table 1.4.6 Willingness-To-Pay (WTP) of Respondents Experiencing a Nuisance from Windmills: Contingent Valuation Study, (1995 Danish Kroner)

	Number of Respondents	WTP Per Year
Respondents Reporting Nuisance, But Not Voting For Moving	14	885
Respondents Voting For Moving, But Will Not Pay More For Electricity	6	684
Respondents Wanting to Pay More, But Not Reporting Amount	6	741
Respondents Reporting Amounts	18	1236
Total/Mean	44	982

Notes:

In June, 1995, the exchange rate was 5.5 Danish Kroner for 1 U.S. Dollar.

Table 1.4.7 The Effect of Windmills on Property Values: Hedonic Price Study (1995 Danish Kroners)

	Total Effect per Installation	Effect per Windmill	Effect Per Year Per Windmill
Single Mills	15891	15891	453
Clusters	122755	34385	980
Wind farms	94147	7654	218
Total	68196	16046	457

Notes:

In June, 1995, the exchange rate was 5.5 Danish Kroner for 1 U.S. Dollar.

Studies using Conjoint analysis

Alvarez-Farizo, B. and N. Hanley (2002): "Using Conjoint Analysis to Quantify Public Preferences over the Environmental Impacts of Wind Farms. An Example From Spain" (Energy Policy 30, 107-116).

Wind, as a source of energy is considered ecologically sound. However, infrastructure required for wind energy production is unsightly, destroys landscapes, and interrupts migratory bird flight paths. This study incorporated both contingent ranking (CR) and choice experiment (CE) methods in order to estimate respondent preference for wind farms on the ecologically unique La Plana landscape in northern Spain. This landscape is considered ecologically unique especially in regard to the variety of birds of prey (eagles, goshawks, owls, and sparrowhawks) that nest in the gypsum cliffs and old pine trees.

The CR and CE each consist of 4 attributes: impacts on cliffs, impacts on fauna and flora, impacts on landscape and cost. The environmental impact attributes have two levels: protected and lost. The cost attribute which is defined as a tax has 3 levels: 500, 1000 and 1500 pesetas (PTA). The authors took the resulting 24 possible combinations and selected the minimum efficient set based on Addleman's (1962) method. The choice sets are based on main effects fractional factorial design.

The survey utilized in this study consisted of three parts: attitudinal questions regarding the environment, CR/CE questions and demographic questions. Respondents were given information on electricity production from renewable resources as well as information on the potential effects of wind farms. They are also shown pictures of the current landscape and manipulated photos of the future landscape should there be the development of wind farms. Prior to administration, the survey was tested with focus groups and pilot surveys. The survey was administered using a personal interview format. 488 usable surveys were obtained.

The Contingent Ranking (CR) data are estimated using a double censored Tobit model while the Choice Experiment (CE) data are estimated using a Conditional Logit model. In both the CR and CE coefficients for the environmental impact variables (cliffs, fauna and flora, landscape) are positive and significant indicating that respondents prefer preservation opposed to loss. The cost variable in both models is negative and significant. However, the cost variable is more significant in the CR model.

Marginal willingness to pay for the three environmental attributes (impact on the cliffs, flora and fauna and landscape) is calculated for both the CR and CE questions. Using both methods the authors find that marginal willingness to pay to avoid an increased level of impact on flora and fauna is the greatest (CE: 6290 pesetas (PTA) CR: 3978 PTA). This is followed by landscape (CE: 6161 PTA, CR: 3378) and lastly cliffs (CE: 3580 and CR: 3062).

Table 1.4.8 provides a comparison of the marginal willingness to pay values calculated for the Contingent Ranking and Choice Experiment methods. These values are a weighted average across all of the groups in the sample.

Table 1.4.8 Marginal Willingness to Pay Values for the Contingent Ranking and Choice Experiment Techniques (pesetas)

	Choice Experiment	Contingent Ranking
Cliffs	3580	3062
Fauna and Flora	6290	3978
Landscape	6161	3378

Ek, K. (2002): "Valuing the Environmental Impacts of Wind Power: A Choice Experiment Approach" (Thesis, Lulea University of Technology, Sweden).

This study examined the preferences over the different attributes of wind power in Sweden using a choice experiment approach.

A mail survey of 1000 randomly selected Swedish residential homeowners was conducted beginning in March 2002. Two follow up reminders were mailed subsequent to the initial mailing. A total of 547 completely or partially usable surveys were returned. Adjusting for unknown addresses and unable to answer, the overall response rate was 56%. Data analysis was conducted from surveys that had complete answers (488 surveys which yielded 2,928 observations).

The first part of the survey solicited respondents' attitudes towards the environment, towards electricity production in general, and towards wind power generation in particular. In the second part respondents were asked to state their choices in six different choice sets followed by questions on why these choices were made. The last part of the survey collected socio-economic information.

Respondents were asked to value change in current monthly electricity prices from 50 - 60 Swedish kroner to decrease (increase) in price by 10, 5 (5, 10, 15) Swedish kroner.

The mean implicit price estimate ranged from -2.18 to 3.47 (2002) 10^{-2} Swedish kroner, depending on the attribute, per kilowatt hour of electricity. The study provided recommendations on how best to expand wind power in Sweden while minimizing the environmental external costs associated with wind power development.

Random effects binary probit technique was used for model estimation. The willingness to pay was modelled as a function of the characteristics of wind power, its location, price of wind power, membership in environmental organization, environmental attitude, and age.

Table 1.4.9. shows that the mean (95% confidence interval) implicit price estimate range from -2.18 (-3.05 to -1.32) to 3.47 (2.55 to 4.30) *10^{-2} Swedish Kroner* per kilowatt hour of electricity. Values listed in (2002) Swedish kroner.

Table 1.4.9 Implicit Price Estimates of Various Attributes Associated with Wind Power in Sweden (2002 10^{-2} Swedish Kronor (SEK)/kWh).

	Mean	Min	Max
Noise	0.67	-0.01	1.32
Mountain	-2.18	-3.05	-1.32
Offshore	3.47	2.55	4.30
Height	0.26	-0.46	0.98
Small	1.55	0.49	2.60
Large	-1.63	-2.54	-0.70

Notes:

Min, max values are based on 95% confidence interval of mean values; Noise - generated by the turbine; Mountain - if located in mountainous area; Offshore - if located offshore; Height - if higher than 50 metres; Small, large - grouping of wind turbines.

Ladenburg, J., A. Dubgaard, L. Martinsen and J. Tranberg (2005): "Economic Valuation of the Visual Externalities of Off-shore Wind Farms"(Food and Resource Economic Institute).

In order to value the benefit of having wind mills erected further off shore in new wind farms, this study utilises the choice experiment method to find the willingness to pay (WTP) for Danish citizens for increasing the distance from shore to wind farm.

Three different sub samples were asked their preferences for different sets of wind park layouts. The differences in the sets were number of windmills in the parks, as well as distance from shore. As the distance increases, less windmills will be seen from the shore, and the study aims to find the willingness to pay for having the windmills erected further off shore.

The three samples analysed were from 1) the Danish general population, 2) local population close to Horns rev, and 3) local population in Nysted. The two latter local populations have wind farms close by. The questionnaires were sent postal. The sample sizes were 700 for the national and 350 each for the local samples. The survey used four alternative wind farm distances from the coast, namely: 8, 12, 18 and 50 km. The payment vehicle was additional costs of electricity per household per year. The questionnaires were tested with focus group and pre-tests. In the national sample, 375 out of 700 mailed questionnaires were returned. In the Nysted sample, 170 out of 350 were returned, and in the Horns rev sample, the numbers were 140 out of 350.

From these samples, three sub samples were used; constructed using the full sample (B-model), a sample containing respondents who were certain in their choice (C-model) and finally a sample containing respondents, who according to a defined set of questions were considered consistent and rational in their choice (R-model).

The results showed that WTP increases with distance in all samples, except for the Horns rev sample, in which semi-long distance from the shore is valued most highly.

A logit model is used to estimate the effects of seven independent variables; six wind farm attributes plus one interaction variable on the choice of the respondents.

Table 1.4.10 Willingness to pay for having future off-shore wind farms located at the specified distances from the shore - relative to an 8 km baseline (DKK per household per year 2004)

	12 km from shore	18 km from shore	50 km from shore
National	332	707	904
Horns rev	261	643	591
Nysted	666	743	1223

3.4.2 Discussion of aesthetic effects of wind studies

- Overview of locations

The European studies that value aesthetic effects of wind parks are from the Nordic countries (Norway, Denmark and Sweden) and from southerns Europe (France and Spain).

- Benefits included

The surveys on aesthetic effects of wind focus either on visual effects only, or visual effects and noise and environmental effects of wind mills in general.

- Important environmental stressors

The environmental stressor in this case is development of wind parks.

- Availability - methodology – possibilities for benefit transfer

One should note that there are more studies carried out for both hydropower and wind, but where the aesthetic effects are not distinguishable. These have not been included. Also, there are studies of valuation of “green electricity” where one can not distinguish hydropower, power from wind parks etc, which have not been included either.

One special feature of the valuation studies on visual effects of wind mills, is that the alternative (or the reference scenario) seems to be of some or even great importance, at least the Norwegian studies point in that direction. That may imply that not only visual effects have been included in the WTP, but also tells that this may be difficult to keep apart for respondents. If the alternative is “worse” that is involving polluting substances etc, wind mills seem to be preferred. However, if the electricity alternatively can come from other environmental friendly options, the choice is more uncertain. These same problems do not occur when wind mills are moved "further away" ashore – where the difference is only that (hardly) nobody can see them – and the electricity production is the same.

3.5 Monetary estimates of landscape aesthetic effect of hydropower

3.5.1 Summary of studies

Studies using Contingent valuation

Bergland, O. (1998): "Valuing Aesthetical Values of Weirs in Watercourses with Hydroelectric Plants - Verdsetjing av estetiske verdiar i tilknytning til tersklar i regulerte vassdrag" (Norwegian Water Resources and Energy Directorate (NVE)).

The study is a contingent valuation study, to find the WTP for people in the Meråker/Stjørdal municipalities, for increasing the aesthetical value of watercourses by building weirs. Answers from 120 interviews in Meråker and Stjørdal municipalities have been analysed and the response to the costing part of the study was excellent.

The payment vehicle used was increase in electricity bill. Two valuation techniques have been used. There were 5 protest zero bids, and 21 "real" zero bids. Using a double yes/no question, the WTP was estimated to be NOK 300 per year per household. Those who used the watercourse for recreating had a higher WTP, NOK 390. Willingness to pay increased with income.

Using the question of maximum willingness to pay, the estimated payment fell to NOK 175. In this case, there was no distinction between users and non-users of the watercourse. The true value for willingness to pay probably lies somewhere between these values. About 2/3 of those interviewed reported that they considered the biological consequences in addition to the impact on the landscape.

Table 1.5.1. Mean WTP for aesthetic improvement of the Stjørdal watercourse by building weirs, per household per year (NOK, 1998)

	Mean WTP	Median WTP
Single dichotomous choice	507	333
Double dichotomous choice	299	209
Open ended bid	173	133

Regression analysis was performed, and out of the explanatory variables the following had a significant effect on the WTP: whether or not the respondent was an active recreational user, and income. Location (municipality) did not have a significant correlation to WTP.

Hansesveen, H. and G. Helgås (1997): "Environmental Costs of Hydropower Development - Estimering av miljøkostnader ved en vannkraftutbygging i Øvre Otta" (Master Thesis Norwegian University of Life Sciences (UMB)).

The purpose of this paper was to estimate the environmental cost connected to hydropower development in Øvre Otta. The method chosen is contingent valuation.

The analysis is based on about 350 personal interviews among Norwegian tourists and the inhabitants of the municipalities of Skjåk, Lom and Vågå. The respondents were first given information about the project and then asked if they were willing to pay a certain amount to avoid the negative effects. A number of different sub samples were asked about their willingness to pay for preventing the hydroelectric power plant to be built. The sub samples were: 1) cabin owners and Norwegian tourists, 2) Foreign tourists, 3) inhabitants of Skjåk, Lom and Vågå. All respondents were interviewed person to person, and all were asked their willingness to pay for preservation of the Øvre Otta watercourse. Of a total of 352 respondents interviewed, 3 interviews were incomplete, and 6 interviews were deemed unreliable, so the effective sample was 343.

The questionnaire went through pre-testing before the survey was conducted. The method employed was contingent valuation, and the bidding was dichotomous choice and open ended. In the dichotomous choice part, the bid in question was varying between a list of bids (high and low). Payment vehicle was an increase in the electricity bill.

The total WTP was estimated to be 39.3 million NOK/year. The environmental cost was estimated to be 0.04 NOK/kWh.

Table 1.5.2 Mean WTP per household per year to preserve the Øvre Otta watercourse (NOK, 1997)

	Mean WTP
Norwegian tourists and cabin owners	516
Skjåk	971
Lom and Vågå	1399

Logistic regression analysis was performed. The sub sample of inhabitants of Skjåk, Lom and Vågå, age was negatively correlated to the respondent agreeing to pay the sum in question for preventing the hydroelectric power plant from being built. Whether or not the respondent is environmentally aware is significant for whether or not he/she is willing to pay to preserve the watercourse (degree of awareness positively correlated to WTP).

Navrud S. (2004): "Environmental Costs of Hydropower, Second Stage - Miljøkostnadsprosjektet Trinn 2"(EBL report 181- 2004).

The survey was carried out to estimate willingness to pay to avoid development for 4 alternative hydropower plant developments in Hordaland, Western Norway.

The Contingent Valuation survey was used on a random sample of 360 households in Western Norway. More specifically, the households were located in Hordaland. The interview subjects were interviewed mainly face to face, whilst those subjects who own vacation cabins (and who live in different parts of the country) were interviewed over telephone (CATI; Computer Assisted Telephone Interview).

Respondent were shown maps and illustration of the alternative development. The description of effects on conservation of nature, natural heritage and outdoor recreation, hunting and fishing was approved in advance by the interested parties; developer, communities, property owners and an organisation of protester to the development. Respondents were asked to state their willingness to pay to avoid the development after being given the description of effects. The payment vehicle was an additional charge to the electricity bill.

Regression analysis was applied, and the following factors identified as significantly contributing to a higher WTP for conservation of the river: high environmental awareness, higher education, higher income, and that the subject was an active user of the natural environment (hunting, fishing, etc). Factors contributing to a lower WTP were: high age, willingness to develop Geitåni and Rasdalselva.

Table 1.5.3 Annual environmental costs (2002-NOK) of developing four rivers for hydropower in Voss and Vaksdal, Norway. Based on stated willingness-to-pay (WTP) to avoid the development by the affected population; i.e. the population in the communities Voss and Vaksdal and owners of cabins in the affected area.

Location (river)	Annual environmental cost (NOK)	Mean annual electricity production (GWh)	Environmental cost per kWh
Rasdalselva	1,623,442	20.0	0.08
Geitåni	1,054,222	21.0	0.05
Skårdalselva	519,523	1.8	0.29
Fossegjelet	491,647	0.7	0.70

Lienhoop, N. and D. MacMillan (2007): "Valuing Wilderness in Iceland: Estimation of WTA and WTP Using the Market Stall Approach to Contingent Valuation" (Land Use Policy, 24 (1): 289-295).

Iceland's remaining wilderness area contains unique flora, fauna and landscape features (deserts and rich ecosystems). In an effort to diversify the economy of Iceland, there has been a proposal to harness three rivers in a hydro-scheme within the wilderness area of Eastern Iceland. The hydro-scheme would include reservoirs, dams (up to 190 m), road networks, diversion canals and tunnels, powerhouses and power-lines. Potential recreation benefits associated with these changes include scenic jeep driving and walking.

The potential development of a hydro-scheme in one of Europe's few remaining wilderness areas demands that a full account of benefits and costs are considered. In this article the non-market values, associated with a project in the Eastern Icelandic wilderness, were explored using the market stall method (MS) of contingent valuation (CV).

The market stall (MS) approach to contingent valuation (CV) involves 2 contacts with participants to obtain their willingness to accept (WTA) and willingness to pay (WTP) values. Thus, 6 groups were recruited in the summer of 2002 using e-mail, telephone directory and word of mouth in Reykjavik, Iceland. 65 of a possible 82 respondents were recruited. 53 respondents attended the 1st meetings for a response rate of 65%. The 1st involved a moderated information session and question and answer period followed by the initial elicitation of willingness to pay (WTP) and willingness to accept (WTA) using the payment card and open-ended CV methods. The WTA question asked respondents to consider what they would accept as compensation for loosing the wilderness area while the WTP questions asked respondents to consider what they would pay for a hydro-scheme. Both were considered in terms of household expenses. Between the 1st and 2nd meetings respondents were asked to consider the issue, review information and talk with others. At the 2nd meeting respondents were again asked to consider their WTP and WTA values.

The 1st meeting consisted of a moderated information session and question and answer period. Using both the payment card (-14, 000 Krona (Kr) to 13, 500 Kr) and the open-ended method, respondents were asked to provide WTA/WTP values. The WTA value was framed as compensation for the loss of wilderness and the WTP value was framed in terms of gaining the benefits of the hydro-scheme. The payment vehicle was presented as household expenses (electricity bills, VAT and prices of goods). Between the 1st and 2nd meeting respondents were encouraged to review information, consider the issue and talk with family and friends. At the 2nd meeting respondents again stated WTP/WTA values. Attitudinal and demographic information was collected.

Results indicated that respondents were WTA a mean value of 780, 107 Kr at the 1st elicitation followed by 863, 929 Kr at the 2nd elicitation while respondents were WTP a mean of 17, 925 Kr at the 1st elicitation and 19, 498 Kr at the 2nd elicitation.

Analysis on the willingness to accept (WTA) data was completed using linear regression. Explanatory variables included strength of respondent preference, income, environmental organization membership, and outdoor interests. Strength of preference had a significant negative effect on WTA while, income and environmental organization membership had a significant positive effect on WTA.

Table 1.5.4 gives the mean and median open-ended willingness to pay and willingness to accept values for the first and second meetings.

Table 1.5.4 Mean and Median Willingness to Pay and Willingness to Accept Values from the First and Second Meetings Based on the Open Endend Contingent Valuation Method (2002, Icelandic Kronas)

	Willingness to Accept		Willingness to Pay	
Number of Respondents	First Meet	Second Meet	First Meet	Second Meeting
Number of Respondents	23	21	15	14
Mean	780, 107	863, 929	17, 925	19, 498
Median	50, 000	50, 000	12, 500	10, 000
Standard Deviation	2, 267, 717	2, 361, 612	26, 139	27, 003
Standard Error	472, 852	515, 346	6816	7217
Minimum Bid	465	500	0	0
Maximum Bid	10, 000, 000	10, 000, 000	100, 000	100, 000

Kosz, M. (1996): "Valuing Riverside Wetlands: The Case of the 'Donau-Auen' National Park" (Ecological Economics 16, pp. 109-127).

The study examines the costs and benefits of different proposed projects of the Donau-Auen wetlands east of Vienna. These include the establishment of an internationally recognised national park and the construction of hydroelectric power stations. A contingent valuation study was undertaken to estimate the willingness-to-pay for wetlands in an unchanged natural state.

A sample of 962 Austrians were chosen by a random-quota procedure and interviewed in-person.The sample of respondents corresponded to a representative sample concerning the socio-economic characteristics of the Austrian population 14 years of age and older.

Respondents are presented with alternative development projects for the wetlands area. These were 1) the establishment of a national park in all available areas (11,500 ha) including private property, with measures taken to avoid further river bed erosion; 2) construction of a hydroelectric power station which would leave 9,700 ha of remaining upstream areas; and 3) the construction of a power station that would leave only 2,700 ha of wetlands undisturbed.

Information was obtained on preferences of respondents for: the national park with and without the use of energy potential e.g. hydroelectric power stations; willingness-to-pay for entrance fees, and their motives for payment. Five-hundred and seventy-two individuals indicated they would prefer the national park without the use of energy potential. Of these, 50.2 % were willing to pay a contribution to assure the establishment of the park, whereas 49.8% refused payment. The contingent valuation question was formulated as an open-ended question where the payment vehicle was an earmarked tax.

The willingness-to-pay for the 3 variants of the national park was obtained. Using the conservative estimate of average annual WTP per Austrian (over the age of 14) of 329.25 Austrian Schillings, multiplying by the Austrian population (over the age of 14), and discounting at a rate of 2% over an infinite time horizon yields an estimated present value of non-use benefits of 109.5bn Austrian Schillings. Respondents were also asked to divide their WTP into three categories: existence values, bequest values and option values, of which the respective percentages were: 50.84%, 37.24%, and 11.29% respectively.

Studies using Conjoint Analysis

Carlsen, A.J., J. Strand, and F. Wenstop (1993): "Implicit Environmental Costs in Hydroelectric Development: An Analysis of the Norwegian Master Plan for Water Resources" (Journal of Environmental Economics and Management 25, no. 3, 201-211).

This study examined the decision-making process behind the Norwegian Master Plan for Water Resources and derived values for environmental attributes impacted by hydroelectric development. The attributes considered included: nature conservation, outdoor recreation, wildlife, fish stocks, water supply, preservation of cultural monuments, agriculture/forestry, and reindeer herding.

Data for this study came from the Norwegian Master Plan for Water Resources, which considered 542 hydroelectric projects for development. User-interest scores were assigned to the attributes of each project by the Ministry of Environment. These scores were based on the expected impact on environmental attributes, total hydroelectric capacity, and the regional economy given the development of any of the proposed hydroelectric projects. The user-interest scores were assigned to eight project attributes, including: nature conservation, outdoor recreation, wildlife, fish, water supply, preservation of cultural monuments, agriculture/forestry, and reindeer herding. The impact evaluations were based on information from experts, planners, power companies and interest groups. Projects were then ranked in order of priority with lower-ranked projects being considered more appropriate for development. Construction and operating costs of the projects were obtained from the Ministry of Environment.

Project priority ranking was modelled as a function of project size, construction and operating costs per unit of expected output, user-interest scores, and the regional economic index. Two model specifications were tested: i) ordinal logistic and ii) ordinary least squares.

Attribute ratings made by the Ministry of Environment were used in conjunction with construction and operating costs to derive willingness to pay (WTP). WTP to avoid the impacts of hydroelectric development on attributes ranged from NOK 0.03 to 0.26 ($0.004 to $0.04) depending on the estimation method used and the attribute valued (1983 Norwegian kroner/U.S. dollars). The highest values were water supply and agriculture, rather than traditional environmental attributes such as wildlife, fish, and recreation.

Willingness to pay (WTP) to avoid environmental damage was specified as the ratio of the estimated coefficients of the user interest scores to the estimated coefficients of the construction and operating costs per unit of expected output. WTP is presented graphically in the article. Tables 1.5.5. and 1.5.6. report the WTP values for 8 public goods expressed in ore per kilowatt hour (ore/kWh), Norwegian kroner per kilowatt hour (NOK/kWh), and in United States dollars per kilowatt hour ($US/kWh). Table 1.5.5 presents results from an ordinal logistic regression and table 1.5.6 presents results from an ordinary least squares regression. The article also calculated expected total implicit cost per kilowatt-hour for each project by adding the construction and operating costs to the estimated values.

*Table 1.5.5 Willingness to Pay Values for Eight Environmental Attributes Impacted by Hydroelectric Development, Ordinal Logistic Results (1982 Norwegian Kroner and U.S. Dollars)**

	Ore per Kilowatt Hour**	Norwegian Kroner per Kilowatt Hour	U.S. Dollars per Kilowatt Hour***
Nature	13	0.13	$0.01 to $0.02
Recreation	10	0.10	$0.01
Wildlife	4	0.04	$0.006
Fish Stocks	11	0.11	$0.01 to $0.02
Water Supply	20	0.20	$0.02 to $0.03
Cultural Sites	13	0.13	$0.01 to $0.02
Agriculture	19	0.19	$0.02 to $0.03
Reindeer Herding	10	0.10	$0.01

Notes:

*Values are derived from ordinal logistic estimation. All values presented here are approximations and detailed calculations can be obtained from the authors. User interest scores for each of the public goods ranged from -4 being a very serious negative impact to +4 being an equivalent positive impact. Values given for projects with user interest scores of -4 are reported here. Values for scores of -1 to 4 are reported in the article.

**One hundred ore are equal to one Norwegian kroner .

***One U.S. dollar is approximately equal to 7 Norwegian kroner.

Table 1.5.6 Willingness to Pay Values for Eight Environmental Attributes Impacted by Hydroelectric Development, Ordinary Least Squares Results (1982 Norwegian Kroner and U.S. Dollars) per Kilowatt Hour*

	Norwegian ore per kilowatt hour	Norwegian Kroner per Kilowatt Hour**	U.S. Dollars per Kilowatt Hour***
Nature	14	0.14	$0.02
Recreation	11	0.11	$0.01 to $0.20
Wildlife	3	0.03	$0.004
Fish Stocks	13	0.13	$0.01 to $0.20
Water Supplies	26	0.26	$0.03 to $0.04
Cultural Sites	12	0.12	$0.01 to $0.20
Agriculture	20	0.20	$0.02 to $0.03
Reindeer Herding	12	0.12	$0.01 to $0.20

Notes:

*Values are derived from ordinary least squares estimation. All values presented here are approximations and detailed calculations can be obtained from the authors. User interest scores for each of the public goods ranged from -4 being a very serious negative impact to +4 being an equivalent positive impact. Values given for projects with user interest scores of 4 are reported here. Values for scores of -1 to 4 are reported in the article.

**One hundred ore are equal to one Norwegian kroner .

***One U.S. dollar is approximately equal to 7 Norwegian kroner.

Sundqvist, T. (2002): "Quantifying Household Preferences over the Environmental Impacts of Hydropower in Sweden: A Choice Experiment Approach" (Dissertation, 2002:26, Luleå University of Technology, Sweden).

The purpose of this choice experiment study was to analyze Swedish households' attitudes towards green electricity from hydropower. Specifically is investigated the households' willingness to pay for different environmental attributes and, thus, indirectly mitigation measures to limit the environmental impacts from hydroelectric production.

The environmental good valued was environmental impacts arising from hydroelectric production: downstream water level, erosion and vegetation, fish. The payment vehicle was increase in electricity price per kWh.

The extent of the environmental change was described as follows:

Downstream water level (status quo, +25%, +50%), erosion and vegetation (status quo, -25%, -50%), fish life (status quo, mitigating measures adapted to migratory species or all species) and price change per kWh (6 levels ranging from 0 öre to 25 öre, where one öre is equal to one Swedish krone divided by 100).

The study was conducted in the spring of 2002 by distributing a mail-out survey to 1000 randomly chosen Swedish house-owning households in the National Register. Respondents to the final send-out were not pre-recruited nor given reply-incentive. Two complete send-outs were made with three weeks in between them. Another three weeks later a reminder was mailed out. The final sample consisted of 397 individuals, which implies a response rate of 40 percent.

The questionnaire was developed by using input from meetings with hydropower industry representatives, colleagues and graduate students. A draft questionnaire was tested in a focus group and a limited pre-test of the survey was carried out among house owners to test the design of the questionnaire. The pre-test resulted in minor changes. The final questionnaire included two parts: 1. questions about the respondents' attitudes towards the environment and electricity in general, and "green" electricity in particular, and 2. the choice experiment including descriptions of the different environmental attributes as well as cost information to the respondents.

The estimated individual marginal willingness to pay for the attributes ranged between -0.57 (2002 Swedish öre) per kWh for water level (+50%) and 1.67 öre per kWh for preservation of all fish species.

The model is specified as probit where the choice of alternative (improvement or status quo) is a function of the following explanatory variables: whether the respondent regularly buys green products, regularly fish for recreation, prefer government provision of green electricity, chose alternative which gave the most value for the money, could not afford to pay more for green electricity, would rather spend money to make other power sources more environmentally benign, weighted all attributes against each other, gender, income, children, education and electricity heated house.

Implicit prices are estimated and interpreted as marginal willingness to pay, measured as öre (one Swedish krone divided by 100 or approximately 0.16 US cents 2002) per kWh. For the attributes in the choice experiments the estimated marginal willingness to pay is: water level (+50%) -0.57, water level (+25%) 0.55, erosion and vegetation (-50%) 1.48, erosion and vegetation (-25%) -0.26, fish (all) 1.67 and fish (migratory) 0.60. These mean values reflect the individual willingness to pay for changes in the different attributes.

Table 1.5.7. Implicit Price Estimates, 95% Confidence Intervals (2002 Swedish öre per kWh)

	Mean	Min	Max
Water level (+50%)	-0.57	-1.39	0.25
Water level (+25%)	0.55	-0.38	1.47
Erosion & vegetation (-50%)	1.48**	0.58	2.39
Erosion & vegetation (-25%)	-0.26	-1.31	0.78
Fish (all)	1.67**	0.54	2.81
Fish (migratory)	0.60	-0.32	1.49

Notes:

** Statistically significant at the 5-percent level.

3.5.2 Discussion of aesthetic effects of hydro power studies

- Overview of locations

The number of European surveys valuing aesthetic effects of hydro power is limited. The largest number of surveys is from Norway, but there are also valuation studies from Sweden, Iceland and Austria. It is not surprising that a large number of studies are from Norway, because this country has a much higher percentage of hydropower (99 %) in their electricity production than any other country. Further, large proportion of the country's rivers and water courses are already used for power production, implying that the interest in the remaining (not-developed) water courses increase. However, also Iceland and Sweden do have proportions of their electricity production stemming from hydro power.

- Benefits included

The surveys on aesthetic effects of hydro power, often include other factors than aesthetic effects as well, because development of hydro power implies visual effects as well as changes in biodiversity etc, and the aesthetic effects may be difficult to distinguish. As an example, the Bergland study aiming at valuing aesthetic effects only, reports that a proportion of the reported WTP is for biodiversity. Other studies, like the Austrian, Carlsen et al. from Norway and the study from Iceland, do not even aim at distinguishing visual effects, but value a spectre of environmental effects from hydro power development. For some of the studies, it is difficult to see whether aesthetic effects are actually included (as part of other effects) at all.

- Important environmental stressors

The environmental stressor in this case is always the hydro power development.

- Availability - methodology – possibilities for benefit transfer

The relatively few studies available, and the “bundle” of environmental goods valued (purposely or not) in the existing studies, makes it difficult to foresee good benefit transfer of these values for aesthetic effects of hydro power. More original studies of good quality, and with clear understanding of which (bundle of) environmental effects valued, are needed first.

However, for economic estimates of environmental effects of hydro power development as such, there are studies available, and benefit transfer has been carried out in Norway. We will report from this study in order to see how this could be done, and the possible errors and problems involved.

3.5.3 Example of benefit transfer of estimates of hydro power environmental effects

The description below draws heavily on Navrud (2001c): Environmental costs of hydro compared with other energy options (Hydropower and Dams, Issue 2, 2001).

In the ExterneE project a CV study, based on an extensive EIA of the project plan was used to estimate the environmental costs of the Sauda hydro development project in south-western Norway. The Sauda project is an upgrading and extension of an existing hydro project. Such schemes are likely to be the dominating strategy for future hydroelectric development in Norway and many European countries, because the lack of new sites available for development. Sauda consists of further development of a previously developed area (known as the Basis project) and six diversion projects in bordering undeveloped areas. The affected population was assumed to be the local community of Sauda with about 2300 households, but also inhabitants in other parts of Rogaland and Hordaland counties (about 316 000 households), since the project will affect some of the few remaining undeveloped rivers in south-western Norway. Therefore a CV study of 300 households in Sauda and 300 households in Rogaland and Hordaland counties was conducted by way of personal interviews. Photos and a video showing the projects and simulated views with the project, a map showing the locations of the different types of impacts and a table of impacts summarising the EIA, were used to describe the environmental impacts to the respondents in the CV study.

The damage costs to recreation, cultural heritage objects and ecosystems per kWh vary between the seven project areas. The results show how impacts are site- and project specific. Environmental costs range from less than 0.1 eurocents per kWh in the project area being upgraded, to about 0.6 eurocents per kWh for reduced water flow in an important recreational area and 1.7 eurocents per kWh for aesthetic impacts from reduced water flow in the Sagfossen waterfall close to the local community at Sauda.

The low damage cost of the upgrading project, which contributes 51.5 per cent of the overall electricity production of Sauda, explains the low damage cost of about 0.2 eurocents per kWh for the overall Sauda project.

A similar CV survey of a more controversial Norwegian hydro power project, the River Øvre Otta found annual environmental damage costs of this overall project in the order 0.5-0.6 eurocents per kWh (Hansesveen and Helgås 1997, study cited above).

The results from these two CV studies have been used to estimate the damage costs of a portfolio of projects assumed to be representative of Norwegian hydropower projects (Vågnes et al. 2001). This benefit transfer exercise was based on a unit value (WTP/kWh) transfer, both with and without adjustments based on expert assessment according to a list of criteria, including characteristics of the rivers/lakes, the development plan and its potential impacts, as stated in EIAs of the project.

Average annual environmental costs of Norwegian hydro power projects were found to be 0.28-0.30 eurocents per kWh, using unit value transfer with and without adjustments, respectively. Two hydro power EIA experts compared the seven projects within the Sauda and Øvre Otta study with the projects in the national portfolio of hydropower projects, to determine which projects were similar enough to justify unit value transfer. Run-of-river projects in the national portfolio could not be valued, since there are no original valuation studies of such projects in Norway.

3.5.4 Aesthetic effects of overhead power transmission lines

Power transmission lines are not related to wind and hydro only. However, some recent studies have been carried out for transmission lines for hydro and wind power. Therefore a study that sum up the surveys for overhead power transmission lines and report from a new survey will be discussed here.

Navrud, S., R. Ready, K. Magnussen and O. Bergland (2008): Valuing the social benefits of avoiding landscape destruction from overhead power transmission lines- Do cables pass the benefit-cost test? (Landscape Research 33 (3): 1-16).

Increased demand for electricity and increased trade in electricity lead to an increased need to upgrade existing power transmission lines and building new ones. Overhead transmission lines create negative landscape aesthetic impacts. Cables (underground or in the sea) will avoid these impacts, but they come at a much higher cost. In Norway the construction costs for cables are 2.5 times higher for 22 kV lines, and 4, 6 and 8-10 times higher for 66 kV lines, 132 kV and 420 kV lines (which is the central grid), respectively.

There is little empirical evidence on the economic value of negative impacts on landscape from overhead transmission lines. Contingent Valuation (CV), Choice Experiments (CE), and Hedonic Pricing (HP) can be used to value the external costs of transmission lines. For Hedonic Price to work the transmission lines need to pass through or close to residential areas (or areas with summer houses/cabins). Sims and Dent (2005) report Hedonic Price (HP) analyses and expert assessments by real estate assessors and agents of the impacts of high voltage overhead transmission lines in the UK. They found depreciation in house prices in the range of 6 to 38 % from the HP analysis, and 5-10 % from the expert assessments. Note, however, that these estimates include *all* external effects of overhead transmission lines, and the aesthetic impacts on the landscape can not be singled out.

However, CV and CE can be used value aesthetical impacts only, and these Stated Preference (SP)) methods can be used in both pristine natural areas and urban areas. To our knowledge there are only four Contingent Valuation (CV) studies on the visual intrusion from overhead transmission lines[2] before the one reported in the Navrud et al. (2008) paper; one study in each of the countries France, Canada, UK and Norway[3].

Luc Michaud (1995) used CV in both France and Canada to elicit people's willingness-to-pay (WTP), in terms of a tax increase for three years to pay for the costs, to use cables (underground) instead of overhead transmission lines and avoid both the visual intrusion and potential health risks from EMFs. WTP results are also linked to a visual index, based on the percentage of the view from the main living room window covered by the overhead transmission lines.

Atkinson et al (2004) used Contingent Valuation to estimate both positive and negative willingness-to-pay (WTP) to replace existing transmission towers with four types of new design. Positive WTP was elicited in terms of increased electricity bills to get one of the new designs, and negative WTP was elicited in terms of intended actions with regards to signing a petition and making contributions to a group working to preserve the existing towers. (The latter payment vehicle has serious biases since payment is not coercive and the link between payment and the provision of the good is uncertain). This CV study estimates only the (net) effect of visual intrusion, but only the marginal impact of replacing the towers, and not the overall impacts on landscape aesthetics of the transmission line. 800 respondents living from 500 m to 5 km from overhead transmission lines in England and Wales were interviewed in-person.

In her M.Sc. thesis Gurholt (1998) reports a CV survey of households in two residential areas close to a planned new overhead transmission line outside the city of Kristiansand in Southern Norway. This M.Sc. thesis was also a pilot test of the CV questionnaire used in the study reported here.

Navrud et al (2008) focus on a CV study in the Oslo area in Norway. The CV scenario was constructed to value visual intrusion only (and eliminating impacts from EMFs by explicitly stating that there would be the same health risk from EMFs whether the lines were overhead or as cables underground). 608 households in the eastern part of the municipality of Oslo, Norway were interviewed. A main factor determining the *total* aesthetic costs of transmission lines is the number of households affected; i.e. the number of households that potentially could experience a welfare loss from visual intrusion. It is often difficult to define this "extent of the market" to sample from in CV studies. If the market is defined too narrowly, and only the households thought to be most affected are defined as the sample population, we run the risk of underestimating the aggregate WTP. However, surveying people from a larger geographical area could lead to people stating their WTP for a more comprehensive and/or general good than the one described in the CV scenario, and hence overestimating the aggregate WTP. Therefore, we test how WTP varies with the distance from the transmission lines.

The case of upgrading existing above-ground transmission lines in the Abildsø area in Oslo versus putting the transmission lines underground was chosen for several reasons. The area had existing towers and overhead transmission lines, and there was an on-going

[2] In some cases potential health risks of electromagnetic fields (EMFs) and other impacts are also included

upgrading process where the existing lines would be replaced by larger ones. During this process the alternative of putting the lines underground as cables was considered by the local energy company. Thus, the affected households considered this as a realistic option.

Figure 1.5.1 shows that the existing transmission line runs through three main areas, named Link A, B and C. Link A runs from the Abildso area and is mainly a residential area, and so is link B. However, part of the area near to Link B is a small nature conservancy area with Lake Østensjo as an important area for bird life, and open green space and recreational area for the local residents. Finally, the area close to the Skullerud area (Link C) is mainly an industrial/ communications and services areas.

Link A Link B Link C

<--------------o------------------------o---------------------o---------------o-------------------->

Lake Østensjo

Lambertseter Abildso Abildso school Bakkehavn farm Skullerud Hanaveien

Figure 1.5.1. The existing transmission line was divided into three parts; Link A, B and C. Total length of Links A, B and C is about 5 km, out of which A+B is 3.2 km.

Two focus groups and a pilot study of 50 respondents were conducted before the final version of the CV questionnaire was constructed and used in the main survey.

After introductory questions, and information about power transmission lines overhead and cables, the respondents were introduced to alternatives where shorter or longer parts of the total stretch to be up-graded, was converted to an underground cable. Following the introduction to the alternatives, the payment mechanism was introduced.

The WTP questions followed then for the different alternatives in each version of the questionnaire. Before stating their WTP respondents were reminded about their budget constraints; i.e. by telling them that if they used money to have underground cables they would have less money to spend on other things.

Of 601 respondents, 65, or 10.8%, were classified as protest zero responses. A total of 76.8% out of 538 non-protest respondents stated that they would pay something to have some or all of the power line buried as a cable. This percentage varied somewhat with the households´ distance from the transmission line. To assess other factors that influence the probability to state positive WTP, a probit regression was estimated. Two different types of regressions were run, a short regression for WTP calculation, and a long regression to assess validity. Five different distributions were modelled, the Weibull, normal, log-normal, logistic, and log-logistic. In every regression, the Weibull distribution provided the best fit to the data, as measured by the log-likelihood statistic.

Combining the proportion of positive WTP with the estimated magnitudes of WTP, we get the following unconditional mean WTP for each distance zone, see table 1.5.8.

Table 1.5.8 Mean and median WTP/household/year for all respondents with positive WTP and "true" zero WTP (i.e. excluding protest zero answers) for the three distance zones, where distance zone 1 is closest to the transmission lines and distance zone 3 furthest away.

Links	Distance Zone 1 (0- 200 m)	Distance Zone 2 (200 m – 1 km)	Distance Zone 3 (> 1 km)
	Mean WTP	Mean WTP	Mean WTP
A	679.6	222.3	408.5
B	560.6	237.7	230.4
C	155.9	332.1	358.6
A+B	1975.8	277.4	442.4
A+B+C	1597.7	483.8	745.8
B+C	453.4	414.5	388.4

3.5.5 Possibilities for benefit transfer of estimates for aesthetic effects of power transmission lines

The very few existing valuation studies, and varying natural conditions and individual preferences between sites make benefit transfer difficult and the transfer errors potentially big. Thus, more valuation studies are needed in order to improve transfer of landscape aesthetic values for policy use in cost-benefit analysis and externality costing.

Future policy use of results from valuation studies like the ones reported here critically depends on the validity and reliability of transfer of these estimates in both space and time. In particular one should note that some of the exercises were carried out in densely populated urban areas. This means that lots of people are affected. One may also expect that per household WTP may be different in areas full of aesthetic "disturbing" infrastructure from per household WTP in a pristine nature area.

Benefit transfer from the study site in urban Oslo to pristine fjord landscape would probably not work very well, while one would expect the results to be more accurate for "similar" urban areas. Thus, the experiences from these studies should be used to conduct new CV and CE surveys in other urban areas and in natural areas in Norway and Europe, in order to calculate the transfer errors involved in national and international benefit transfer of landscape aesthetic effects. The experience from tests of the validity and reliability of international benefit transfer tests of health impacts (Ready and Navrud 2006), environmental goods (Navrud and Ready 2007), cultural heritage (Tuan and Navrud 2007) and meta analyses of agricultural landscape (Santos 2001, 2007) should be utilised in the quest for improved assessment and economic valuation of landscape aesthetic impacts.

3.6 Benefit transfer methods and validity tests of benefit transfer

3.6.1 Potential for benefit transfer

In Navrud (2001a) the potential for benefit transfers is discussed.

There are three main approaches to benefit transfer:

i) Unit Value Transfer

a) Simple unit value transfer

b) Unit value transfer with adjustment for income differences

ii) Function Transfer

a) Benefit function transfer

b) Meta Analysis

In approach i) the unit value at the study site is assumed to be representative for the policy site; either without a) or with b) adjustment for differences in income levels between the two sites (by using GDP per capita or purchase power parity indices). In approach ii) a benefit function is estimated at the early study site and transferred to the policy site a), or a benefit function is estimated from several study sites using meta-analysis b) then values for the independent variables at the policy site are used in the function to calculate WTP at the policy site. A benefit function from a CV survey would be WTP as a function of site and good characteristics and characteristics of the respondent. Meta analysis would also include characteristics of the different studies as a variable, since estimated values could be affected by even small methodological differences.

Ready et al (1999) was the first study to test the reliability of benefit transfer across several countries. In five European countries, the Netherlands, Norway, Portugal, Spain and the UK, respondents in a CV study were asked their WTP to avoid six specific episodes of ill-health (which correspond to endpoints in exposure – response functions between air and water pollution and ill-health). We found that if the goal was to predict average WTP across a population, approaches i)a), i)b) and ii)a) performed equally well. The transfer error was 37-39%, which should be assessed relative to a random sampling error within each country of 16%. Thus, if the goal is to predict average WTP of the entire population of the target country, the simple unit value transfer method does this cheaply and with relatively low error. The question is then whether the error is acceptably low for policy used like benefit-cost analyses. To answer this question, the policy makers could compare the costs of doing a new study with the expected costs of making the wrong decision when using the benefit transfer estimates in a cost-benefit analysis. Statistical decision theory and Bayesian analysis would be ideal instruments for this purpose.

Whether the result from Ready et al (1999) also applies to environmental goods like freshwater fish stocks, we do not know. However, it shows that benefit transfer across countries in Europe is possible, and could produce estimates reliable enough for cost-benefit analysis at considerable cost- and time-savings compared to conducting new valuation studies.

3.6.2 Practical Benefit Transfer Guidelines for Cost-Benefit Analysis

There are few detailed guidelines on value transfer. In the US there exist guides that cover the key aspects of conducting a value transfer, notably Desvouges et al (1998) aimed at transfer for valuing environmental and health impacts of air pollution from electricity production. Adapted to the economic valuation of environmental goods in general we would propose the following eight steps guidelines (Navrud 2007):

1) Identify the change in the environmental good to be valued at policy site
2) Identify the affected population at the policy site
3) Conduct a literature review to identify relevant primary studies (based on a database)
4) Assessing the relevance and quality of study site values for transfer
5) Select and summarize the data available from the study site(s)
6) Transfer value estimate from study site(s) to policy site
7) Calculating total benefits or costs
8) Assessment of uncertainty and acceptable transfer errors

STEP 1 - Identify the change in the environmental good to be valued at policy site

(i) *Type of environmental good*

The Total Economic Value (TEV) of environmental goods can be broadly classified in three groups: (i) Direct Use Value (i.e. recreational activities like swimming, walking, fishing, hunting) (ii) Indirect Use Values (i.e. eco-system services like biological diversity, climate regulation and carbon sequestration of forests, watershed services (water quality and quantity), soil stabilisation and erosion control, aesthetic value of landscape) and Non-use Values (Existence and preservation/bequest values including historic/cultural heritage values and endangered species habitat).

(ii) *Describe (expected) change in environmental quality*

a) baseline level,

b) magnitude and direction of change

(gain vs. loss; and prevention[4] *vs. restoration)*

STEP 2 – Identify the affected population at the policy site

Desvousges et al. (1998) use this as the last step in their Value transfer guide. However, it is important to identify the size of the affected population at the policy site before we review the valuation literature and evaluate the relevance of selected studies. The trans-

[4] A distinction should be made between prevention (which preserves the original/undisturbed environmental good) and restoration. People have been found to put a higher value on keeping the original (i.e. prevention) than restoration.

ferred value should come from the same type of affected individuals in terms of spatial scale.

If we just want to establish the use value of some activity, the relevant, the affected population is the recreationists. If we would like to estimate both use and non-use values, and the policy site is only of local importance (e.g. a small lake with many substitutes regionally), we should use only the population of the municipality. If there are few substitutes for the sites at the regional level, the population in several communities, or even the county population, should be used. If the good is of national importance, e.g. a national park, or the single site of an endangered species in the country, the national population should be used.

For use values, the number of individual recreationists should be estimated (before and after the change), while for non-use values (or use and non-use values combined) the number of households should be the unit of aggregation at the relevant geographical scale (community, regional/county or national level).

STEP 3 - Conduct a literature review to identify relevant primary studies

The next step is to search the EVRI database www.evri.ca to identify similar studies from the same country or other Nordic countries (and other closely located countries like the UK and Germany). This recommendation is based on value transfer validity tests showing that studies closer spatially tend to have lower transfer errors. Studies closest in time should be selected for the same reason. However, one should note that this evidence is not conclusive. If there are no or only very few primary studies of the environmental good in question, or the valued change in the quality of the environmental good is outside the range considered at the policy site, the same databases and other bibliographies (e.g. the UK valuation studies list) should be searched for relevant studies. Meta-analyses (including also North American studies) could also be consulted, bearing in mind the limitations for value transfer of meta analyses with a broad scope (i.e. too large variation in definition of the environmental good). Some meta analyses can be found in EVRI www.evri.ca.. There are few meta-analyses of Nordic studies only. Lindhjem (2007) have constructed a spreadsheet database of all non-timber benefits valuation studies in Norway, Sweden and Finland, and used this to perform a meta-analysis. Two important conclusions emerged from this study: (1) WTP is found to be insensitive to the size of the forest, casting doubt on the use of simplified WTP/area measures for complex environmental goods; and (2) WTP tends to be higher if people are asked as individuals rather than on behalf of their household. North American meta-analyses could also be consulted (since most valuation studies and meta analyses have been conducted there); e.g. Rosenberger & Loomis (2000) and Shrestha & Loomis (2001) for recreational activities.

Databases of valuation studies do not always have all the data needed for the relevance of the study site to be evaluated, and the full study report should be collected.

STEP 4 – Assessing the relevance and quality of study site values for transfer

Here, the quality of the relevant valuation studies is assessed in terms of scientific soundness and richness of information. Desvousges et al. (1998) identify the following criteria for assessing the quality and relevance of candidate studies for transfer:

i) Scientific soundness *- The transfer estimates are only as good as the methodology and assumptions employed in the original studies*

- Sound data collection procedures (for Stated Preference surveys this means either personal interviews, or mail/internet surveys with high response rate (>50 %), and questionnaires based on results from focus groups and pre-tests to test wording and scenarios
- Sound empirical methodology (i.e. large sample size; adhere to "best practice"-guidelines guidelines for (see e.g. Bateman et al 2002 for a manual in Stated Preference studies, and Söderquist and Soutokorva 2006 for a guideline in assessing the quality of both revealed and stated preference primary valuation studies)
- Consistency with scientific or economic theory (e.g. links exists between endpoints of dose-response functions and the unit used for valuation, statistical techniques employed should be sound; and CV, CR, CE, HP and TC functions should include variables predicted from economic theory to influence valuation)

ii) ***Relevance*** *- the original studies should be similar and applicable to the "new" context*

- Magnitude of change in environmental quality should be similar
- Baseline level of environmental quality should be similar
- Affected eco-system services and environmental goods should be similar
- The affected sites should be similar when relevant (e.g. when assessing recreational values)
- Duration and timing of the impact should be similar
- Socio-economic characteristics of the affected population should be similar
- Property rights, culture, institutional setting should be similar

iii) ***Richness in detail*** *– the original studies should provide a detailed dataset and accompanying information*

- Identify full specification of the original valuation equations, including precise definitions and units of measurements of all variables, as well as their mean values

- Explanation of how substitutes (and complementary) goods were treated
- Data on participation rates and extent of aggregation employed
- Provision of standard errors and other statistical measures of dispersion

All three criteria and their components are equally important for assessing the relevance and quality of the study.

STEP 5 – Select and summarize the data available from the study site(s)

Several parallel approaches should be applied, and the results from these should be used to present a range of values:

Search the studies to provide low and high estimates, which can define a lower and upper bound for the transferred estimate, respectively. Collect data on the mean estimate and standard error, and specific spatial transfer errors if available (if not use the general transfer errors of + 25-40 % based on a review of studies testing the validity of benefit transfer). Consult relevant meta-analyses to see if the scope of these is narrow enough to provide relevant information about the estimate to be transferred. The scope could be too wide to produce reliable estimates if the meta-analysis consists of studies that vary a lot in terms of methodology, and the environmental good considered.

Compare the magnitude of the value from the meta-analyses, when methodological parameters in the meta-function is set according to the best practice guidelines and a context corresponding to the policy site. Methodological variables in meta-analyses (of CV studies) that reflect best practice guidelines include survey mode (preferable in-person interviews or mail surveys with high response rates), studies should be conducted after the NOAA Panel guidelines to CV (Arrow et al. 1993) (year of study often used as a proxy variable for quality in some meta-analyses), similar as possible in magnitude and direction of change, substitutes, characteristics of the population; and a realistic and fair payment vehicle (not voluntary contribution without a provision point mechanism, and not payment vehicles that create a large degree of protest behaviour).

STEP 6 – Transfer value estimate from study site(s) to policy site

a) Determine the transfer unit

The recommended units of transfer for use and non-use values are:

i) use value:

For recreation: Consumer surplus per activity day[5]

For ecosystem services: WTP/household/year

[5] An activity day is defined as one individual performing recreation for a shorter or longer period during one day.

For recreation, consumer surplus per year (or per visit) per visitor could also be used, but then the average number of activity days (or visits) per year should be the same at the study and policy sites.

For some ecosystem services, alternative estimates could be used, i.e. a unit cost per ton of carbon (with a sensitivity analysis) for carbon sequestration if this cost is based on abatement costs (in terms of market price of tradable permits for CO2) of modelled damages e.g. Richard Tol´s FUND –model).

ii) non-use value: WTP/household/year[6]

The use of total WTP per ha ecosystem or landscape type assumes both the same size of the affected population and that the value pr. ha is constant. However, empirical evidence shows that WTP does not increase proportionally with the number of ha.of ecosystems or landscape types (for non-timber benefits of forests; see Lindhjem 2006). Since SP surveys clearly show that WTP per unit of area varies widely, I should caution against converting households´ stated mean WTP for a discrete change in environmental quality to marginal values like WTP per km or ha per household. *However, this unit is "better" than total WTP per km or ha, because in the latter case one also has to assume similar population density at the policy and study sites.*

b) Determine the transfer method for spatial transfer

If the policy site is considered to be very close to the study sites in all respects, *unit value transfer* can be used. If we have got several equally suitable study sites to transfer from, they should all be evaluated and the transferred values calculated to from a value range.

For unit transfers between countries, differences in currency, income and cost of living between countries can be corrected for by using Purchase Power Parity (PPP) corrected exchange rates; see e.g. http://www.oecd.org/dataoecd/61/56/1876133.xls. Within a country we could also use unit value transfer with an adjustment for differences in income level, and an income elasticity of WTP lower than 1.

Function transfer can be used if value functions have sufficient explanatory power[7] and contain variables for which data is readily available at the policy site. Most often the "best" model is based on variables where new surveys have to be conducted at the policy site to collect data. Then one could just as well perform a full-blown primary valuation study. If models are constructed based on variables for which there exist data at the study site, they very often have low explanatory power. In general, WTP functions based on Stated Preference surveys (especially Contingent Valuation) have much lower explanatory power than functions based on Travel Cost (TC) and Hedonic Pricce (HP) studies.

[6] Some studies of use and non-use values have asked for individual WTP. However, we view the household as the smallest "economic" unit for none-use values of environmental goods in forests. Multiplying individual WTP with the mean number of adults per household would tend to overestimate household WTP. Therefore, we have conservatively assumed that the reported individual WTP is equivalent to household WTP.

[7] Roughly said to be having a higher adjusted R^2 than 0.5, i.e. explaining more than 50 % of the variation in value

Thus, it could be more relevant to use function transfer transferring estimates from these Revealed Preference methods.[8]

If relevant *meta-analyses* are identified (see previous step), estimates from these could also be used in a comparison of several transfer methods. Sensitivity analysis could be performed to see how much the transferred value estimate could vary. The constructed upper and lower values should be used to bound the transferred estimate.

To conclude, **unit value transfer** is recommended as the simplest and most transparent way of transfer both within and between countries. This transfer method has in general also been found to be just as reliable as the more complex procedures of value function transfers and meta-analysis. This is mainly due to the low explanatory power of willingness-to-pay (WTP) functions of Stated Preference studies, and the fact that methodological choice, rather than the characteristics of the site and affected populations, has a large explanatory power in meta-analyses. Generally speaking, error bounds of $\pm$ 25-40 % should be used if the study and policy sites are very similar (which we should strive for); see Navrud (2004). If there is less similarity between study and policy sites, error bounds of $\pm$ 100 % should be used.

c) Determine the transfer method for temporal transfer

The value estimate should be adjusted from the time of data collection to current currency using the Consumer Price Index (CPI) for the policy site country. If we transfer values from a study site outside the policy site country, we first convert to local currency in the year of data-collection, using PPP (Purchase Power Parity) corrected exchange rates in the year of data collection, and then use the local CPI to update to current-currency values.

However, environmental goods could also increase more or less in value than the goods the CPI is based on. However, there is no general rule for adjustments of preferences for environmental goods over time.

STEP 7 - Calculating total benefits or costs

For **non-use values**, mean WTP/household/year is multiplied by the total number of affected households to derive the annual benefit or cost. If WTP at the study site is stated as annual WTP for e.g. 5 or 10 years, the total benefits or costs should be calculated as the Present Value (PV) over that same period. On the other hand, if WTP is stated as one-time amounts the amounts must be viewed as a present value (of all benefits from the environmental good in question).

The general equation for calculation the present value of the benefits PV (B) is:

T

[8] This does, however, not mean that we should concentrate on RP studies when we perform new primary studies, as only SP methods are capable of valuing non-use values and future changes in environmental quality.

$$PV(B) = \sum_{t=0} B_t / (1 + r)^t$$

where B_t is the total benefits in year t, *T* is the time horizon (for the stated WTP amounts) and *r* is the social discount rate (e.g. r = 0.03 (3 % p.a.) is the social discount rate currently used by the European Commission). Benefits and the discount rate are stated in real terms, i.e. 2007-kr and the discount rate is a real rate of return (i.e. corrected for inflation, and not a nominal rate).

If the time horizon is not stated in the WTP question in SP surveys, we must assume that this is an annual payment over an infinite time horizon, i.e. $t \rightarrow \infty$. In this case, and if the annual benefits B_t are the same each year, the equation above can be simplified to:

$$PV(B) = B_t / r$$

Annual benefits B_t are equal to aggregated WTP over the affected population (WTP_{tot}), which can be calculated as:

$$WTP_{tot} = n \times WTP_i$$

Where n = number of affected households, and WTP_i = mean Willingness–To–Pay for household *i*. Since WTP per household varies between different parts of the affected population (e.g. with distance from the site, whether users and/or non-users are considered etc.), the estimates from the study site(s) should be based on the same type of affected population as at the policy site. If this is not possible, distance decay in WTP (e.g. percentage reduction in WTP pr km increased distance from the environmental good) could be assumed, based on empirical evidence from relevant study sites (if such evidence does exist and suggests this).

If we calculate use values, we just substitute households with individual recreationists in the equation above and use estimates for consumer surplus per activity day times the increase or decrease in number of activity days to calculate total use value of the project. For uses other than recreation, values are often elicited on a household basis, and the same procedure as for non-use values can be employed.

When aggregating damages and costs of environmental goods, we also need to consider whether these goods are independent (meaning we can just add them up), or if they are substitutes or complementarities. In the first case we would overestimate aggregated damage or benefits, while in the latter case we would underestimate.

STEP 8 Assessment of uncertainty and acceptable transfer errors

Validity tests of benefit transfer (Navrud 2004) indicate that the transferred economic estimates should be presented with error bounds of $\pm$ 40 %. However, if the sites are very similar, or the primary study was designed with transfer to sites similar to the policy site in mind, an error bound of ± 25 % could be used. If the study and policy sites are not quite close, unit transfer could still be used, but arguments for over- and underestimation in the transfer should be listed and the unit value should be presented with error bounds of ± 100 % (based on the observed large variation in individual estimates observed in validity tests (Navrud 2004).

When performing a Cost-benefit analysis of a new project or policy, the estimated PV of benefits (costs) should be compared with the corresponding PV of costs (benefits). The effect on total annual benefits (costs) due to an expected general transfer error of 25-40 % should be calculated in order to see if this reduces the PV of benefits (increases the costs) to a critical level, i.e. the PV of net benefits becomes negative (from positive). If this is so, the transfer errors are large enough to change the outcome of our CBA, and we should try to increase the accuracy of the transferred estimate (either by conducting a full primary study or calibrating the transferred value by conducting a small scale primary study).

When there is a need for estimates of environmental goods for policy purposes, a CBA of conducting a new environmental valuation study should be performed in order to determine whether the costs of a new primary study is worth the benefits in terms of lower probability of making the wrong decision. These decision rules could be used as a rough test of whether value transfer has acceptable transfer errors. For further reading in benefit transfer /value transfer; see Navrud and Ready (2007).

3.6.3 Validity of benefit transfer and acceptable transfer errors

Transfer errors arise when estimates from study sites are adapted to policy sites. These errors are inversely related to the degree of correspondence between the study site and the policy site. Assume there is an underlying meta-valuation function that links the values of a resource (such as a lake) or an activity (such as swimming or recreational fishing) with characteristics of the markets and sites, across space and over time. Further, hypothesize that a primary research project samples from this meta-function. The meta-valuation function may be constructed as an envelope of a set of study site functions that relates site values to characteristics or attributes associated with each site, including market characteristics, physical site characteristics, spatial characteristics, and time (Rosenberger and Phipps 2002). The degree that any of these sets of factors affects value transfer accuracy is an empirical question. However; the greater the correspondence (or similarity) of the policy site with the study site the smaller the expected error (Boyle and Bergstrom 1992; Desvousges et al. 1992).

In the value transfer validity tests, two or more parallel valuation studies are conducted at different sites. Then an imaginary transfer is conducted from a study site (or a pooled data set from several study sites) to a policy site, where we have also performed an original study. The transferred value, WTP_T, is then compared to the value estimated in the original valuation study at the policy site, WTP_P. The *transfer error* (TE) is calculated as the percent difference between the transferred estimate and the policy site estimate

$$TE = \frac{|WTP_T - WTP_P|}{WTP_P}$$

Ready et al (2004) show in their transfer tests of CV estimates of respiratory illnesses in five European countries that even if the distribution of WTP had been the same in all countries, they would have measured an average transfer error of 16%. Thus, they point out the average transfer error of 38 % they did find between should be assessed relative to this background level of random sampling error.

Much academic work has taken place in the past 10 years, testing the validity in of alternative value transfer methods for different environmental goods. However, even fairly small transfer errors can be rejected using the classical statistical tests (usually t-tests with null hypotheses of transferred value being equal to the original value). Bergland et al (1995, 2004) rejected value transfers statistically in cases of average transfer errors of less than 20 % in two CV studies performed simultaneously of similar water quality improvements in two closely located and similar lakes. However, the standards of accuracy required in academic work may exceed those viewed as tolerable by policy-makers, especially in cost-benefit analyses like those likely to be performed at the national level e.g. in relation to the EU Water Framework Directive to prioritise alternative investments in water quality and to show disproportionate costs. Kristofersson and Navrud (2005) suggest the use of equivalence testing as more appropriate and a clear compliment to the shortcomings of the classical tests Equivalence tests test the null hypothesis of *difference* between the original and transferred value estimates (which is in most cases what we would expect rather than similar values). Equivalence tests also combine the concepts of statistical significance and policy significance into one test, by defining an acceptable transfer error prior to the validity test. For examples of applications of these tests, see Kristofersson and Navrud (2003) and Muthke and Holm-mueller (2004).

Table 1.6.1 shows that errors in individual transfers vary a lot, both within and between different validity tests and for all transfer methods. Since some of the transfer validity tests are performed under ideal conditions (i.e. same SP survey instrument used on a similar good in a nearby location at the same point in time; e.g. Bergland et al 1995) they might underestimate transfer errors in practical transfer exercises. However, surprisingly many of these validity tests are performed under less than ideal conditions, and probably reflect quite well the transfer errors in practical value transfers. Brouwer (2000) surveyed seven of these value transfer studies and found that the average transfer error is around 20-40% for unit value transfers, and as high as 225% for benefit function transfers. Ready et al (2004), however, found an average transfer error of 38% in a multi-country transfer tests both for unit and function transfer. Shrestha and Loomis (2001) found an average transfer error of 28% in a meta-analysis model of 131 US recreation studies. Santos (1999) found that a international meta-analysis of CV estimates of landscape features could obtain less than 30% transfer error in 26% of cases; and less than 50% transfer error in 44% of cases.

Several of the studies listed in Table 1.6.1 support the hypothesis that the greater the correspondence, or similarity, between the study site and the policy site, the smaller the expected error in benefit transfers. Lower transfer errors resulted from in-state transfers than from across-state transfers (Loomis 1992; VandenBerg et al. 2001). This is potentially due to lower socioeconomic, socio-political, and socio-cultural differences for transfers within states, or political regions, than across states. In the Loomis et al. (1995)

study, their Arkansas and Tennessee multi-site lake recreation models performed better in benefit transfers between the two regions (percent errors ranging from 1% to 25% with a nonlinear least squares models and 5% to 74% with the Heckman models) than either one when transferred to California (percent errors ranged from 106% to 475% for the nonlinear least squares models and from 1% to 113% for the Heckman models). This suggests that the similarity between the eastern models implicitly accounted for site characteristic effects.

Van den Berg et al. (2001) show accuracy gains when they transfer values and functions within communities that have shared experiences of groundwater contamination than transferring across states, within states, or to previously unaffected communities.

Brouwer (2000) suggests that if non-use values are motivated by overall commitment to environmental causes, they may tend to be relatively constant across populations and contexts. In a contingent valuation survey of the national populations in all Nordic countries Kristofersson and Navrud (2005) found that transfer errors are consistently smaller for the non-use value of a preservation plan for Nordic freshwater fish stocks. The results for a non-use value scenario by non-anglers in Norway and Sweden produced average transfer errors below 20 %. Use values for anglers showed higher transfer errors. It may be that non-use value in these two countries is motivated by similar factors and is relatively context independent, while use value is more context-specific. Clearly, this could be different for other environmental goods, particularly if the good has higher cultural significance in one country (or part of a country).

Table 1.6.1. Summary of benefit transfer validity tests for environmental goods.

Reference		**Resource/Activity**	**Unit value Transfer Percent Error**[9]	**Function Transfer Error**
Loomis (1992)		Recreation	4 – 39	1 – 18
Parson and Kealy (1994)		Water / Recreation	4 – 34	1 – 75
Loomis et al. (1995)	Nonlinear Least Squares Model	Recreation	---	1 – 475
	Heckman model		---	1 – 113
Bergland et al. (1995)		Water quality	25 – 45	18 – 41
Downing and Ozuna (1996)		Fishing	0 – 577	---
Kirchhoff et al. (1997)		Whitewater Rafting	36 – 56	87 – 210
		Birdwatching	35 – 69	2 – 35
Kirchhoff (1998)	Benefit Function Transfer	Recreation/Habitat	---	2 – 475
	Meta-analysis Transfer		---	3 – 7028
Brouwer and Spaninks (1999)		Biodiversity	27 – 36	22 – 40
Morrison and Bennett (2000)		Wetlands	4 – 191	---
Rosenberger and Loomis (2000a)		Recreation	---	0 – 319
VandenBerg et al. (2001)	Individual Sites	Water quality	1 – 239	0 – 298
	Pooled Data		0 – 105	1 – 56
Shrestha and Loomis (2001)		International Recreation	---	1 – 81

<u>Source:</u> Moodified after Brouwer (2000) and Rosenberger (2005)

[9] All percent errors are reported as absolute values

To summarize, the transfer validity studies conducted to date show that the average transfer error for spatial value transfers both within and across countries tends to be in the range of 25% - 40%. However, individual transfers could have errors as high as 100-200%. Function transfer does not seem to perform better than unit value transfer. Meta analyses could also produce high transfer errors, and only those with a limited scope in terms of similar type environmental goods and similar type, state-of-the-art methodology, should be used. The validity tests also supports the hypothesis that it is preferable to find a study site located close to the policy site of interest. The closer the study site is to the policy site, the more likely that both the good being valued and the user population affected will be similar, and therefore the transfer errors would be lower. Transfer validity tests also suggest that transfer errors are smaller if people have had experience with the environmental good in question, but the transfer errors do not seem to be lower for use than for non-use values.

Even if we cannot determine general levels of acceptable transfer errors for different policy use, some general decision rules for how to determine the acceptable transfer errors in cost-benefit analyses (CBA) can be outlined.

There are two main sources of error in estimated values from value transfer:

i) errors associated with estimation of the unit value/value function at the study site

ii) errors from transferring the study site value(s) to the policy site

By using "best practise"-guidelines for original valuation studies we can minimize the first type of errors. The second type of errors arises because we usually would need to transfer estimates both in space and time. Results from validity tests of different value transfer procedures for different type environmental goods have shown that individual transfer errors in spatial value transfer vary from a few present to several hundred percent (see figure 1 below). However, *average* transfer errors, both for national and international value transfer seems to be about ± 25-40 %. In many cases this would be an acceptable transfer error in CBA. Sensitivity analysis should be performed to see if this interval for the estimated values would influence the outcome of the CBA. The size of the critical transfer error, i.e. when Net Preset Value (NPV) of the project is zero, should also be calculated, especially in cases where we suspect the transfer errors could be larger. These cases include international value transfers of complex environmental goods from study sites that are quite different from the policy site in terms of magnitude and direction of change, initial level of environmental quality, availability of substitutes (scarcity), different size of affected areas, different type of population (locally most affected population versus the national population) etc.

In order to reduce the uncertainty and calibrate the transferred value estimate, there is the option of conducting a study at the policy site in terms of a valuation workshop, focus group or a valuation study of a small sample. However, the costs of providing this additional information should be compared to the benefits in terms of reduced uncertainty (and the reduced risk of the CBA showing the wrong outcome). This could be done by adopting a *Bayesian perspective* to inform the decision on whether to conduct primary research at the policy site, and if so how much. Here, value estimates or functions from existing studies can be used to form a prior distribution on the value of the good at the policy site. Valuation research conducted at the policy site provides new information on the value of the good. An updated distribution of the value of the good at the policy site contains information from both previous studies conducted at other sites, and from the new research conducted at the policy site. Thus, the decision should be made based on a comparison of the expected value of the information to be gained and the cost of conducting new research. This Baysesian approach could also be adopted where information on the value of similar goods is available, but there is concern that the value at the policy site may be unique (see Atkinson et al 1992 for a theoretical discussion, and Leon et al 2002 for an application to a national park). However, the approach currently seems not to be sufficiently developed and simplified to be applied on a routine basis in practical value transfer exercises.

One should be even more careful in using value transfer for policy uses where the demand for accuracy is high; especially environmental costing exercises aimed at determining the level of environmental charges and NRDAs aimed at calculating compensation payments to be paid by those that were responsible for the emissions causing the damage.

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4. Marginal damage cost of greenhouse gas emissions

Onno Kuik and Luke Brander

4.1 Introduction

The impact pathway of the emissions of greenhouse gases is extensive as compared to the impact pathways of conventional air pollutants, both in time and in space. Greenhouse gases are stock pollutants that through their build-up in the atmosphere cause an increase in temperatures and changes in related climate variables at a global scale and over a long time period. The time period over which the impacts of current emissions occur extends the lifetime of the gases in the atmosphere because of extensive time lags in the climate system. Apart from the geographical and temporal extend of the impacts, the impacts are also manifold as climate change can potentially affect many sectors of society, including health and safety, economic production and consumption, recreation, and environmental and natural assets. Because of the unprecedented rate of warming and climate change that is predicted by scientific assessments, we are unsure or even in deep ignorance about the extent and the probabilities of some of the more extreme impacts.

4.2 Marginal Damage Costs

4.2.1 Integrated Assessment Models

Because of the extensive and complex impact pathways of greenhouses gas emissions, their damage is commonly assessed with the help of more or less complex Integrated Assessment Models (IAM). An IAM is essentially a computer model of economic growth with a controllable externality of endogenous greenhouse warming.[10] It combines [some] dynamics of global economic growth, with [reduced-form] dynamics of geophysical climate dynamics. Worldwide, there are only a handful of IAMs that have passed some test of professional quality and that have been consequently updated with the most recent scientific information on climate dynamics. These IAMs include DICE (and various variants thereof), FUND and PAGE. For a description of the PAGE model, see Hope (2006*)*. FUND has been used extensively in recent EU projects such as GreenSense, NEEDS and Methodex. We will briefly describe its structure.

The FUND Model

FUND is a computer model that contains a set of exogenous scenarios and endogenous perturbations, specified for sixteen major world-regions. The current model version (FUND 2.9) runs from 1950 to 2300, in time steps of a year. The prime reason for extending the simulation period into the past is the necessity to initialise the climate change impact mod-

10 Weitzman, 2007a.

ule. The period 1990-2100 is based on the FUND scenario, which lies somewhere in between the IS92a and IS92f scenarios. Note that the original IPCC scenarios had to be adjusted to fit FUND's sixteen regions and yearly time-step. The period 2100-2300 is based on extrapolation of the population, economic and technological trends in 2050-2100, that is, a gradual shift to a steady state of population, economy and technology. The model and scenarios for the period 2100-2300 are not to be relied upon. This period is only used to provide the forward-looking agents in *FUND* with a proper perspective.

The exogenous scenarios concern economic growth, population growth, urban population, autonomous energy efficiency improvements, decarbonization of the energy use, nitrous oxide emissions, and methane emissions.

Incomes and population are perturbed by the impact of climate change. Population falls with climate change deaths, resulting from changes in heat stress, cold stress, malaria, and tropical cyclones. Heat and cold stress are assumed to affect only the elderly, non-reproductive population; heat stress only affects urban population. Population also changes with climate-induced migration between the regions. Economic impacts of climate change are modelled as deadweight losses to disposable income. Scenarios are only slightly perturbed by climate change impacts, however, so that income and population are largely exogenous.

The endogenous parts of FUND consist of carbon dioxide emissions, the atmospheric concentrations of carbon dioxide, methane and nitrous oxide, the global mean temperature, and the impact of climate change on coastal zones, agriculture and forestry, energy consumption, water resources, natural ecosystems and human health. FUND uses simple models for representing all these components; each simple model is calibrated to either more complex models or to data.

Marginal costs of carbon dioxide are estimated as follows. First, a base run is made with the model. Second, a perturbed run is made in which one million metric tonnes of carbon are added to the atmosphere for the period 2000-2009. In both runs, relative impacts, GDP and population are saved. Marginal costs are estimated using:

$$\frac{\sum_{r=1}^{16}\sum_{t=0}^{250}\left(\frac{D_{r,t}^{P}}{Y_{r,t}^{P}}-\frac{D_{r,t}^{B}}{Y_{r,t}^{B}}\right)\frac{Y_{r,t}^{B}}{\left(1+\rho+g_{r,t}^{B}\right)^{t}}\ (\$)}{10000000\ \ (tC)}$$

where D is monetised damage; Y is GDP, g is the growth rate of per capita income; ρ is the pure rate of time preference; the subscript r is region; the subscript t is time; and the superscript denotes base (B) or perturbed (P) run. That is, the change in *relative* impacts is evaluated against the baseline economic growth – this is to avoid the complications of differential effects on the economic growth path. Impacts are discounted using the standard neo-classical discount rate, viz., the sum of the pure rate of time preference and the growth rate of per capita consumption.

4.2.2 Discounting and 'deep' uncertainty

The recent publication of the Stern Review (Stern, 2007) on the economics of climate change has attracted a lot of attention, not only in the popular press, but also in academic journals. The Stern Review assessed the economics of moving to a low carbon economy, focusing on a medium to long term, plus the potential of different approaches to adaptation and lessons for the UK, in the context of climate change goals. Using the results from an integrated assessment model (the PAGE model), the review estimated that the total damage costs of climate change could be at least 5% of global GDP. If a wider range of risks and impacts is taken into account, the estimates of damage could rise to 20% of GDP or more. The review suggested a marginal cost of emissions of € 85 per ton of CO_2. In contrast to these high costs of inaction, the costs of action – reducing greenhouse gas emissions to avoid the worst impacts of climate change – can, according to Stern, be limited to around 1% of global GDP each year.

Discussions on the Stern Review in the academic literature note that Stern takes an extreme position by applying a pure rate of time preference of only 0.1 percent in combination with a relatively flat marginal utility function and an infinite time horizon (Nordhaus 2007; Yohe, 2006; Tol and Yohe, 2006; Weitzman 2007). Nordhaus (2007) in particular argues that these particularities of the Stern review drive its results and could lead to bizarre policy recommendations.[11] Weitzman (2007) agrees on the discounting issue, but delves deeper into the 'deep' uncertainties of climate change that to a large extent motivated the use of the 'near-zero' social discount rate by Stern. His analysis reaches the somewhat 'unnerving' conclusion that because of 'deep' or 'structural' uncertainty about the scale and probabilities of rare catastrophes that could be associated with 'extreme' manifestations of climate change, expected utility theory (on which cost-benefit analysis is based) may not really give the right answers.[12] Weitzman (2008) therefore calls upon a radically different refocus of IAM's on extremes rather than on, what he calls, "middle-of-the-distribution" kinds of assessments.

4.2.3 Equity weighting

The impacts of climate change may affect people in very different economic circumstances. The aggregation of monetised impacts of climate change over people with (very) different income levels is problematic. The prime reason for this is that the correct metric of social cost-benefit analysis is utility (or its money-equivalent) and not money itself. A

[11] Nordhaus (2007: 696) gives the following example: "Suppose that scientists discover a wrinkle in the climate system that will cause damages equal to 0.1 percent of net consumption starting in 2200 and continuing at that rate forever after. How large a one-time investment would be justified *today* to remove the wrinkle that starts only after *two centuries*? [By the *Review*'s methodology] the answer is that we should pay 56 percent of one year's world consumption today to remove the wrinkle."

[12] Tol (2003) had already put forward similar arguments.

standard assumption in economics is that utility is a declining function of income, that is, an equal amount of money income to a poor man is worth more (in terms of increments in his utility) than to a rich man (in terms of increments in the utility of the rich man). It is common practice in climate change damage cost assessments to use some form of equity-weighting to reflect these differences in marginal utilities of income between the rich and the poor. Through equity weighting more weight is given to monetised damages that accrue to poor people, or –in practice– to people that live in poor regions, than to rich people. Equity weighting matters in a quantitative sense: equity-weighted estimates of the marginal damage costs of CO2 are substantially higher than estimated without equity-weights (Anthoff, Hepburn et al. 2006).

Equity weighting introduces an additional source of variance in global damage estimates. Anthoff et al. (2006) show that, apart from the parameter value of the elasticity of utility, equity weights are sensitive to the regional aggregation of an IAM, the baseline economic scenario against which the climate change damages are evaluated, and also to the income level that is used to normalise the equity weights. Anthoff et al. (2006) find that estimates of marginal damage costs can differ by two orders of magnitude depending on the region of normalisation.

Why is this important? In most equity-weighted estimates of marginal damage costs, the reference level of income that serves to normalise regional equity weights is “average world income”. In aggregating regional damages into global damages, damages in richer regions receive lower weights and damages in poorer regions receive higher weights. While this would make sense for a “global planner”, Anthoff and Tol (2007) argue that this procedure would make no sense for an assessment by a regional planner, such as, for example, the European Commission. The reason for this is that by using average world income in normalising equity weights, domestic (EU) impacts are not valued at domestic (EU) values (because these EU values receive a lower weight in the aggregation). This can easily lead to inconsistencies in policy assessments. Take for example a cost-benefit assessment of measures relating to externalities form energy generation. Physical health impacts from conventional air pollutants (valued at EU ‘utility-prices’) would be valued higher than equal health impacts from climate change (valued at equity-weighted ‘utility-prices’). Anthoff and Tol (2007) calculate the consequences of some policy-consistent variants of equity-weighting (where EU impacts are valued at EU ‘utility-prices’; the variants ranging from not caring for what happens outside the EU to valuing all foreign damages at EU ‘utility-prices’) and conclude that marginal damage cost estimates can differ by two orders of magnitude depending on the variant chosen.

Concluding then, equity weighting is appropriate in the aggregation of climate change damages over regions (people) with disparately different income levels, not only for ethical reasons but also for strictly economic reasons. However, the standard way of equity-weighting that was used in IAMs to estimate marginal damage cost from a global perspective may be less appropriate for use in regional cost-benefit frameworks as in our current project. Apart from this ‘normalisation’ issue, the impact of equity weighting on marginal damage costs is also very sensitive to the elasticity of utility with respect to income, regional aggregation in IAMs, and the baseline economic scenario.

4.2.4 Illustrative marginal damage values for the CASES project

The science of the integrated assessment of climate change damages is steadily progressing. As is common is fast-developing areas of research, cutting-edge theoretical and conceptual contributions, such as those of Weitzman and Anthoff & Tol, have not yet made it into standard assessments. A radical refocus of research strategy along Weitzman's recommendations has not yet occurred, although stochastic elements are increasingly being built into IAM's. For the CASES project, we propose, for the time being, to rely on conventional cost-benefit analysis, keeping in mind the potential caveats.

Choosing between the various estimates of marginal damage costs that have been produced by IAM's in recent years is not easy because of a number of reasons:

1. The estimates of the IAM's are based on specific sets of parameter values of, for example, the social rate of discount and the elasticity of utility. Much (if not most) of the IAM output has been produced to test the sensitivity of these estimates to parameter assumptions. Before using such estimates some reflection on the choice of parameter values is necessary.

2. There are a number of IAM's with their own specific strengths, weaknesses, and (possibly) subjective biases. Rather than relying on one specific IAM we would prefer to take as much IAM information as possible into account by as many different researchers as possible.

3. The quality of IAM's and their underlying data and assumptions are continuously improving. On the one hand, we would therefore prefer the more recent assessments. On the other hand, the most recent assessments have not yet had the opportunity to be reviewed and judged by a larger community. They are still very fresh and (possibly) immature.

The Social Cost of Carbon project by DEFRA (2005) shares a number of the qualities we seek. It is reflexive, it is very much aware of the policy context in which the values are used, and it combines the results of a number of IAM's in a transparent manner. The drawback is that it is two years old, therefore missing the most recent developments. We are of the opinion, however, that the most recent developments have primarily been at the conceptual and theoretical level, viz. the work of Weizmann and Stern, and less at a more precise assessment of the social cost of carbon per se. Therefore, we think that the social cost of carbon, as reported in the DEFRA study of 2005, could very well be used in the CASES project.

The DEFRA study distinguishes between a central guidance value and upper and lower estimates.[13] The values are based on full Monte Carlo runs of the FUND and PAGE models, in which all parameters varied to reflect the uncertainty surrounding the central parameter values in both models. The lower and upper bounds are the 5% and 95%

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[13] In fact the study reports upper and lower central estimates as well as upper and lower bounds. To simplify, we just report the lower bound and upper central estimates that are based on the 5% and 95% probability values of the PAGE model.

probability values of the PAGE model, while the central guidance value is based on the average of the mean values of the FUND and PAGE models.[14]

Common assumptions that were applied in both models were a declining discount rate according to the UK Government 'Green Book'[15] and equity weighting of damages in different regions.[16] The underlying emissions scenarios are the IPCC SRES A2 scenario for PAGE and an emissions scenario for FUND that is in-between the IPCC IS92a and IS92f scenarios. Both models predict rising marginal damages of emissions over time, that is, the present value (discounted back to the year of emission) of the damage of one tonne of CO_2 emitted in 2020 is larger than the present value (also discounted back to the year of emission) of one tonne emitted in 2005. This is a common finding of all IAM's and basically reflects that underlying assumption that the world economy is continuously growing over the models' time horizons. Therefore, even if physical damage due to one tonne of CO_2 would be similar across time (for example causing a similar rise in sea level), the *economic value* of damage would increase (because the stock of coastal assets would have increased in value).

The time profiles of the lower, upper, and central marginal damage costs of CO_2 emissions are shown in Table 4.1 and depicted in Figure 4.1. The lower estimates of marginal damage costs (PAGE5%) evolve from € 4/tCO_2 in 2000 to € 8/tCO_2 in 2030. The upper estimates (PAGE95%) evolve from € 53/tCO_2 in 2000 to € 110/tCO_2 in 2030. The central estimate evolves from € 23/tCO_2 in 2000 to € 41/tCO_2 in 2030. The central estimate is the average of the means of FUND and PAGE.

14 Because of the fact that FUND's mean estimates are particularly sensitive to outliers, the mean in based on the central 99% of FUND's probability distribution (in short: "trimmed mean 1%"). FUND's estimates in $/tC in 1995 prices were converted to £/tC in 2000 prices using the UK Retail Price Index over the period 1995-2000 (22.5%) and the PPP exchange rate between dollar and pound of 2000 ($1.42 = £1).

15 Discount rate of 3.5% until 2020, declining to 3.25% in 2020-2040, 3% in 2040-2080, and 2.5% thereafter.

16 Elasticity of utility with respect to consumption is one. Normalisation is based on "world average income", see discussion in Section 4.2.3.

Table 4.1 Marginal damage costs of CO2 emissions (€2000/tCO2) – by year of emission.

	Low	Central	Upper
2000	4	23	53
2010	5	28	65
2020	6	33	88
2030	8	41	110

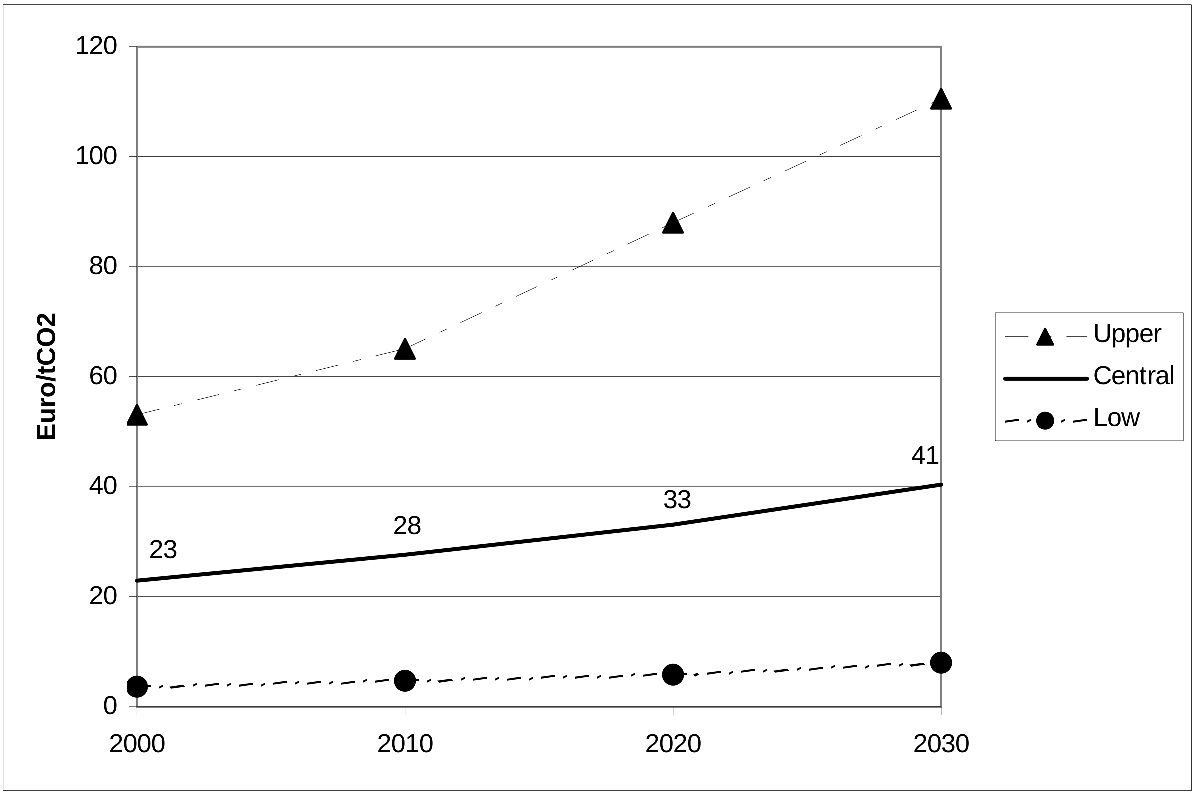

Figure 4.1 Marginal damage costs of CO_2 emissions (€/tCO_2) – Year of emission, prices of 2000 (based on DEFRA, 2005).

4.2.5 Marginal damage costs of other greenhouse gases

The current ExternE methodology uses 1996 IPCC global warming potentials (GWP) to determine the external costs of non-CO_2 greenhouse gases relative to those of CO_2. There are two problems: 1) GWP's have been updated by IPCC; 2) the use of GWP's in economic damage assessment has been criticized.

Table 4.2 shows the GWPs as used by ExternE (1996 IPCC GWP) and the updated values (2001 IPCC GWP) from IPCC's Third Assessment Report (2001). The updated values are slightly higher for methane, and slightly lower for nitrous oxide. Updating the ExternE values of non-CO_2 greenhouse gases with the more recent information on GWP's would be easy.

Table 4.2 Global Warming Potentials (GWP's) of three major greenhouse gases, and global warming damage ratios as computed by the FUND model

Gas	1996 IPCC GWP	2001 IPCC GWP	FUND*
Carbon Dioxide	1	1	1
Methane	21	23	34
Nitrous Oxide	310	296	1413

* FUND, computed for 2025 (average 1% trimmed, 1% pure rate of time preference, no equity weighting, discounted to year of emission), source: (Anthoff 2007) and own computations.

There is, however, a more fundamental problem. The fundamental problem is that GWP's are not very appropriate measures for the ratios of *shadow values* among different gases, nor for the ratios of *marginal damage costs* among gases (Manne and Richels 2001; Fuglestvedt, Berntsen et al. 2003; Tol 2006). Table 4.2 presents the ratio of marginal damage costs among three greenhouse gases as computed by the FUND model. It suggests (much) higher relative damage costs for methane and especially nitrous oxide than indicated by application of the GWP's.

The Fourth Assessment Report of IPCC (Fisher, Nakicenovic et al. 2007) acknowledges this problem in the context to the relative contributions of greenhouse gases to a non gas-specific stabilization target (e.g., in terms of radiative forcing). For example, because of the relatively short lifetime of methane in the atmosphere, its shadow value of mitigation in terms of its contribution to a long-term stabilization target is very low in the short term and becomes relatively high at the end of the planning period. Intertemporal optimization models that calculate a least-cost multigas emissions trajectory therefore do not substantially reduce methane emissions until the end of the planning period (Fisher, Nakicenovic et al. 2007). However, if one considers the costs of climate damage, present methane emissions may have a relatively large effect on near-time global warming (and the rate of change of global warming) and its present value marginal damage costs may therefore be high (cf. Table 4.2). Relative impact weights of greenhouse gases may therefore depend on the type of climate policy target that is selected (long-term stabilization target *versus* short-term targets (e.g., rate of temperature change), the level of the target, and the proximity to the target. IPCC concludes that despite continuing scientific and economic debate on the use of GWP's, no alternative measure has attained comparable status to date (Fisher, Nakicenovic et al. 2007).

4.3 Marginal Abatement Costs

Traditionally the policy debate on climate change has focused on the costs of emissions reductions, i.e. the mitigation of greenhouse gas emissions. Such mitigation costs, or abatement costs, have been used in recent ExternE work (and in the updated ExternE 2005 methodology) as a proxy for environmental cost (externality) analysis. The current recommended value is €19 per tonne of CO_2. This value is based on a marginal abatement cost (MAC) for Europe for emissions reductions required by the Kyoto Protocol for the period 2008-2012. Using agreed IPCC global warming potentials (GWP), this is equivalent to € 399 per tonne of methane (CH4) and € 5890 per tonne of nitrous oxide (N2O)[17].

However, there are many problems with these mitigation costs. The most important of these is that they are not social costs, which are the standard metric required in the ExternE impact pathway approach. There are also other reasons why their use is not ideal - the use of these values can lead to circular reasoning in policy cost-benefit analysis (when comparing 'externalities' against the costs of policies).

An additional problem is that the current estimate only applies to the short term, i.e., until 2012. The current ExternE recommendations do not include estimates of MACs for the period after 2012.

This Section will investigate an approach to the assessment of abatement costs that differs from the current approach in two ways:

1. It focuses on the medium to long term (up to 2050) instead of the short term (2012), and it explicitly describes MAC profiles over time;

2. It focuses on "global" MACs that are consistent with long-run global stabilization targets, in terms of concentrations or radiative forcing potentials, instead of regional (e.g. EU) emissions reduction targets.

The reason for the focus on the medium to long term is obvious when we are considering energy transitions up to 2030 as in the CASES project. The reason for the second point – the focus on "global" MACs consistent with global stabilization targets –needs more explanation. MACs for specific sources (installations, industries, countries) depend in general on the size of the required emission reduction and the flexibility by which the reductions can be spread over different sources, gases, and time ("where", "how", and "when" flexibility). The current ExternE value of € 19 per tonne CO_2 is an example of a MAC estimate that is specific to the targets and rules of the Kyoto Protocol. A change in the rules of the Kyoto Protocol (such as for example with respect to emissions trading) could have major impacts on MACs, without leading to changes in total emissions or concentrations. Moreover, the use of policy-specific MACs can lead to circular reasoning in cost-benefit analysis, as argued above.

In an "optimal" or "intertemporally-efficient" climate policy (see, e.g., (Kolstad and Toman 2001)), emissions in each period would be reduced such that marginal abatement costs and marginal damage costs would be equal in *each* period for *all* sources and would change over time taking into account the dynamics of the economic system *and* the climate system. If marginal damage costs are unknown or surrounded by too much uncertainty, the next-best policy would be a "cost-effective" policy given some (politi-

[17] This uses the 1996 values, which were agreed internationally as the values that Parties are required to use for reporting GHG emissions to the FCCC and the Kyoto Protocol, although they were updated in 2001.

cally) predefined long-term stabilization target. The policy problem can then be mathematically formulated as an optimal control problem, by which the shadow prices of emissions (the control variable) can be solved for some concentration or radiative forcing constraint (the state variable) (see, e.g., (Aaheim, Fuglestvedt et al. 2006)).

In recent years, many research teams have developed computer-based economic models that have computed marginal abatement costs (MAC) of greenhouse gas emissions that are consistent with long-term climate policy targets. These targets are usually expressed in terms of the stabilization at a certain level of concentration of CO_2 or greenhouse gases in the atmosphere or stabilization at a certain level of radiative forcing or global mean temperature. It is possible to interpret these MAC as carbon permit prices in an idealized global emissions trading system that allows the participants maximum “where”, and “when” flexibility, and in some models also “what” flexibility. This means that MAC are equalised across all sources (“where” flexibility), MAC change over time according to some intertemporal optimization rule (“when” flexibility), and that in some models MAC of abating different greenhouse gases are equalised, taking into account their relative warming potentials and different lifetimes (“what” flexibility).

We collected information from 26 different models that were presented in three so-called modelling fora in 2006. A modelling forum is a meeting or a series of meetings of modelling groups that address a common research question, and that use a commonly agreed set of assumptions and a common reporting format. One of the oldest of such fora is the Energy Modeling Forum (EMF) that was established at Stanford University in 1976 to provide a structured forum for discussing important energy and environmental issues. For this study we used the results of the models that participated in EMF-21 that specifically addressed “what” flexibility (trade-offs between different greenhouse gases). We also used results of the models that participated in the Innovation Modeling Comparison Project (IMCP) that specifically addressed the potential impact of induced technical change on long-term abatement and abatement costs and the U.S. Climate Change Science Program (USCCSP) that addressed all these issues.

These different models produce varying estimates of MAC. The analysis presented in this report examines the sensitivity of MAC estimates to the specifications and assumptions underlying these models. By conducting a meta-analysis of model results we aim to identify consensus in the outcomes and the methodological characteristics that drive differences in results. In addition to providing a statistical synthesis of model outcomes, the meta-regression function can also be used to predict MAC given specific values for explanatory variables included in the regression.

The meta-analysis in this report uses more up-to-date model results than previous research (Repetto and Austin, 1997; Barker et al., 2002; Fisher and Morgenstern, 2005). This analysis uses the same model results as Barker et al. (2006), but many more in addition.

4.3.1 Stabilization targets

The ultimate objective of the United Nations Framework Convention on Climate Change (UNFCCC) is the "*stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. Such a level should be achieved within a time-frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure that food production is not threatened and to enable economic development to proceed in a sustainable manner*" (UNFCCC, art. 2). There is no consensus yet on the level at which GHG concentrations would need to be stabilised in order to prevent such dangerous anthropogenic interference, although the European Council and Parliament agreed on the objective to limit average global temperature increase to a maximum of 2°C compared to pre-industrial levels (Asselt and Biermann 2007). The different studies that we analyse in this report have examined different stabilization targets, both in terms of metrics and levels. To be able to compare study results we need to standardise the various stabilisation targets to a common metric. The most commonly used metrics are radiative forcing ($W.m^{-2}$), concentrations of greenhouse gases in the atmosphere expressed in CO_2 equivalents (ppmv CO_2-eq), the concentration of the greenhouse gas CO_2 (ppmv CO_2), and global mean temperature (ºC). Fisher, Nakicenovic et al. (2007) classified stabilization targets into six different categories (I…VI), and showed how the concordance between the targets in alternative metrics (see Table 4.3).

Table 4.3 Concordance between stabilization targets in alternative metrics

	Additional radiative forcing	CO_2 concentration	CO_2-eq concentration	Temperature*
Category	W.m^{-2}	ppmv	ppmv	ºC
I	2.5 – 3.0	350 – 400	445 – 490	2.1 (1.4–3.1)
II	3.0 – 3.5	400 – 440	490 – 535	
III	3.5 – 4.0	440 – 485	535 – 590	2.9 (1.9–4.4)
IV	4.0 – 5.0	485 – 570	590 – 710	3.6 (2.4–5.5)
V	5.0 – 6.0	570 – 660	710 – 855	4.3 (2.8–6.4)
VI	6.0 – 7.5	660 – 790	855 – 1130	5.5 (3.7–8.3)

* Temperature is based on Table TS 5 of the Technical Summary of WG1. The exact correspondence between concentration and temperature is: 450 ppmv CO2-eq = 2.1ºC; 550 = 2.9; 650 = 3.6; 750 = 4.3; and 1000 = 5.5.

Source: based on Fisher, Nakicenovic et al. (2007) and other parts of IPCC 4AR

4.3.2 Research approach: meta-analysis

Meta-analysis is a statistical technique to combine the results of several studies that address a set of related research hypotheses. Meta-analysis extends beyond a standard literature review by analysing and synthesising the results of multiple studies in a statistical manner. This study is the second meta-analysis of MAC estimates, the first being Fischer and Morgenstern (2005).

In this report, meta-analysis is used to examine whether modelled estimates of MAC are dependent upon some key modelling assumptions and structural characteristics of the models. To test such dependencies, a meta-regression model is constructed in which the dependent variable (MAC) is a linear function of a set of i explanatory variables (EVi) and a random error (ε):

$$MAC = \sum_i \beta_i EV_i + \varepsilon \qquad 1)$$

When the function is estimated, the estimate b of the coefficient β shows if and how the explanatory variable affects the dependent (MAC) variable. We are particularly interested in the significance (does the variable have a significant effect on MAC?) and sign (if the effect is significant, what is the direction of the effect: will an increase in the explanatory variable increase or reduce the MAC estimate?).

4.3.3 Description of the database

The 26 models in our database provided "observations" of MAC for different points in time. We collected 62 observations of MAC for the years 2025 and 2050. We normalized these observations that are expressed in different dimensions and currencies into 2005 Euros per tonne of CO_2 (€2005/tCO_2). For normalization, we used consumer price indices (CPI) from the OECD to convert all prices to a common year (2005), market exchange rates from OECD to convert all currencies to a common currency (Euro, €), and molecular weights to convert all physical dimensions to one common physical dimension (CO_2).

Differences between MAC estimates from different studies can be ascribed to differences in the stabilisation targets, assumptions on exogenous developments, and differences in model specification and parameterization. We selected a number of explanatory variables to include in the meta-regression model on the basis of general discussions on MAC in the literature, e.g. IPCC (Fisher, Nakicenovic et al. 2007), and on the basis of an earlier meta-analysis of models that were used to assess the compliance costs of the Kyoto Protocol (Fischer and Morgenstern 2005). The explanatory variables include stabilization target, emissions baseline, various model and policy assumptions, and also the particular forum in which the study was developed. Information on these variables was not available for all MAC estimates. From the 62 observations in our database, 47 (49) observations provided sufficient information to include in the meta-analyses for 2025 (2050). In describing the data below, we therefore make a distinction between the full and restricted data.

The mean MAC value across all 62 observations is € 23.8 per tonne of CO_2 in 2025 and € 63.0 in 2050 (Table 4.4). The median MAC are lower: € 16.2 in 2025 and € 34.0 in 2050. Table 4.4 shows that the spread of MAC across observations is quite large: for 2025 the minimum and maximum estimates are € 0.0 and € 199.9 and for 2050 the spread is € 1.4 to € 449.3 per tonne of CO_2. Table 4.4 also presents the descriptive statistics for the restricted database. The differences between the full database and the restricted database for 2025 are minor. The differences for 2050 are larger, to a large extent because of the exclusion of one study that reported a very high MAC of € 449/t in 2050. The study was excluded from the restricted database because of incomplete information regarding its baseline emissions.

Table 4.4 Summary statistics of MAC of 26 models (€^{2005}/tCO_2)

	2025		2050	
	Full database	Restricted database	Full database	Restricted database
Mean	23.8	23.8	63.0	55.8
Median	16.2	16.2	34.6	32.2
Maximum	119.9	119.9	449.3	209.4
Minimum	0.0	0.4	1.4	1.4
St.dev.	26.7	27.9	72.5	52.9
N	62	47	62	49

The large differences between mean and median MAC values suggest that the distribution of MAC values in our databases is skewed to the right, perhaps with a "thick" right tail with high values. This is indeed the case, as is shown for the restricted data of the year 2050 in the upper panel of Figure 4.2. Because this skewedness may lead to estimation problems, we have taken natural logs of the MAC values and used ln(MAC) as the dependent variable. The lower panel of Figure 4.2 below shows that the distribution of ln(MAC) tends more towards the normal distribution.

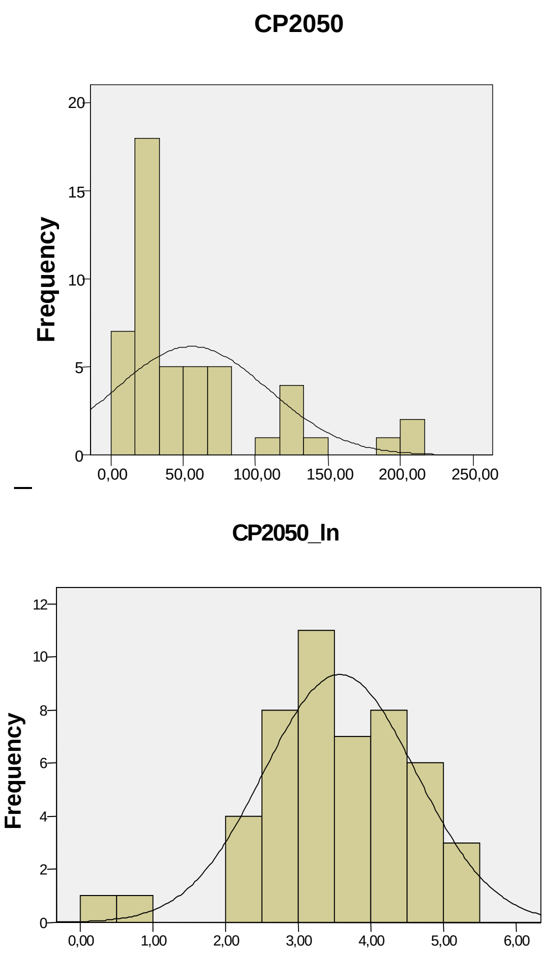

Figure 4.2 Frequency distribution of MAC (upper panel) and ln(MAC) (lower panel)

The following description of data will focus on the restricted database that will be used for the actual meta-analysis.

Most of the studies we collected use stabilization targets in categories III and IV (approx. 3.5 to 4.7 W.m^{-2}). These stabilization targets imply a peak of emissions between 2010 and 2030, and between 2020 and 2060, respectively. The change in global emissions in 2050 relative to emissions in 2000 range between – 30 to + 5 percent for category III targets, and + 10 to + 60 percent for category IV targets (Fisher, Nakicenovic et al. 2007). Results of higher stabilization targets (categories V and VI) have been reported, but as Fisher et al (2007) have commented, they are not very ambitious and even overlap with low to medium baseline scenarios – that is, these targets may be reached without explicit climate policy. We have not included these studies in our analysis. Recent scientific evidence suggests that very low stabilization targets may be needed to avoid irreversible and catastrophic damages, and the European Commission recently confirmed its commitment to a long-term stabilization target of 450 ppmv CO_2-eq (= category I stabilization target) (EC 2007). The modelling fora did not, however, include studies that computed marginal CO_2 abatement costs for stabilization targets below 3.5 $W.m^{-2}$ (categories I and II). In order to increase the range of the stabilization targets included in the data, one study that estimated MAC in accordance with a stabilization target of 350 ppmv CO_2 (450 ppmv CO_2-eq) was added to the database (Vuuren et al., 2006).

We converted all stabilization targets to ppmv CO_2 concentration measures. The variable TARGET ranges between 350 and 550 ppmv CO_2. The average stabilization target across all observations in the restricted database is 506 ppmv CO_2.

The studies use different assumptions on economic growth, industry structure and technological developments, resulting in widely differing baseline emissions paths over time. For example, in our restricted database, the increase in baseline CO_2 emissions from energy and cement over the period 2000-2100 ranges between 8 and 380 percent. Fisher et al. (2007) show that the range of baseline emissions for CO_2 and other greenhouse gases in the EMF-21 studies is comparable to the full range of the IPCC SRES scenarios. They report that the median increase in baseline CO_2 emissions in 133 post-SRES studies is 240 percent. The average increase in baseline emissions over the period 2000-2100 across all observations in our database is 174 percent and the median is 179 percent. The baseline in conjunction with the stabilization target determines the emissions reduction effort and thus, we conjecture, the MAC.

Recent studies have emphasized the cost savings potential of a multigas policy towards the stabilization of greenhouse gas concentrations and radiative forcing. The EMF-21 forum was for example specifically organised to assess this potential. On the basis of the EMF-21 studies, it can be concluded that a multigas policy ("what" flexibility) can potentially reduce marginal abatement costs substantially in comparison to a "CO_2 only" policy. Weyant et al. (2006) report that the EMF-21 studies find on average that MAC of a multigas policy in 2025 are 48 percent lower than a CO_2 only policy for the same long-term stabilization target. The reduction in MAC ranges from 15 to over 70 percent in individual models. In our database, we constructed the dummy variable MULTIGAS, which takes a value of 1 if the study examined a multigas policy, and 0 otherwise. Of the 49 observations in our database, 22 are for multigas.

Another "hot" issue in climate economics research is the impact of "induced" technical change on abatement costs. The IMCP forum specifically addressed this issue. The central idea of induced technical change is that the direction and magnitude of technical change in abatement technologies is dependent upon the overall greenhouse gas reduction policy and the subsequent carbon price. Hence, dynamic economic models should not take technical progress over time as given, but should explicitly model the interactions between policy and technical change. The models in the IMPC forum generally found that induced technical change would lower MAC in comparison to a calculation without this feature.[18] An interesting result of induced technical change is that it can cre-

[18] Note that Smulders and de Nooij (2003) argue that this result is likely in partial models of the knowledge market as used in the IMCP, but unlikely in a complete model of innovation and diffusion.

ate "path dependency" in the sense that the transformation to a carbon-free energy system can become irreversible if the carbon-free technologies become the least-cost option because of (induced) technical progress. If this occurs, the carbon price can begin to decline. Some studies project such a turning point towards the end of this century. The dummy variable ITC has a value of 1 if the model included a specification of induced technical change, and 0 otherwise. 17 observations are for ITC.

In a meta-analysis of economic models that examined the economic consequences of the Kyoto Protocol, Fisher and Morgenstern (2005) found evidence that a model's level of aggregation of regions and sectors had an impact on its estimate of MAC. Statistical significance was found for the number of regions and the number of energy sources in a model. For both variables the relationship with MAC was positive. The authors suggested that greater disaggregation might result in a more realistic representation of rigidities in, for example, international energy markets. We constructed the variables REGIONS and ENERGIES, where REGIONS indicates the number of regions in a model, and ENERGIES the number of primary energy sources. In our database REGIONS varies between 1 and 77; ENERGIES varies between 1 and 9.

From the 1970s there has been a fierce debate on the relative advantages and disadvantages of so-called "top-down" and "bottom-up" approaches in modelling energy-economy interactions. Traditionally, bottom-up models are rich in technical detail, but poor in modelling micro-economic behaviour and macro-economic feedbacks, while the opposite is true for traditional top-down models. The "top-down/bottom-up" controversies have naturally propagated into the area of climate change economics. However, since the mid-1990s a productive dialogue has started between the proponents of the two approaches (Hourcade, Jaccard et al. 2006). Observers have noticed some convergence to a middle ground that they have labelled "hybrid modelling" (Hourcade, Jaccard et al. 2006). Nevertheless, there are still differences between the approaches that might affect the assessment of abatement costs. The dummy variable CGE takes a value of 1 when the model is CGE ("top-down") and 0 otherwise. 33 observations were derived from a CGE model.

A different issue concerns the treatment of intertemporal dynamics within the models. Some models assume the existence of long-lived decision-makers that optimize the timing of consumption, investments and abatement over the entire planning period (intertemporal optimization), while other models optimize only period for period (recursive dynamic). It might be that different dynamics lead to different emissions profiles over time, thereby affecting MAC in any particular year. For further reference we notice, however, that the more stringent the stabilization targets, the less flexibility there is for alternative emissions pathways (Fisher, Nakicenovic et al. 2007). The dummy variable IDO takes the value 1 when the model solves by intertemporal dynamic optimization (IDO), and 0 otherwise. 23 observations are from models that used IDO as the solution concept.

In some models, MAC are bound by some "backstop technology".[19] A backstop technology with respect to energy-related CO_2 emissions is a (hypothetical) technology that can

[19] In other models, the upper bound on the MAC is implicit and may be rather complex.

produce any amount of CO_2-free energy at constant (high) cost. Marginal CO_2 abatement costs in a model with a backstop technology can never rise above a level for which the backstop technology would be the least-cost option. Some recent models have included the technology "carbon capture and storage" (CCS) as a sort of quasi-backstop technology. It is not a real backstop technology because the economic model still endogenously determines the price of CCS, but it nevertheless puts some quasi-cap on MAC.

We have constructed the dummy variable CCS that has value 1 if the model includes CCS or some undefined backstop technology, and 0 otherwise. Among the 49 observations, 26 have explicitly considered CCS or a backstop technology.

Finally, we have constructed dummy variables for the different modelling fora. The dummy variables IMCP and USCCSP have been introduced to check whether there are significant, but otherwise unexplained differences between the three modelling fora. This is all the more interesting as the IMCP models have been accused of making overly optimistic assumptions on technological progress and the costs of emissions abatement (Tol 2006).

The dummy variables IMCP and USCCSP have values 1 if the observation was presented in this forum, and 0 otherwise (some models participated in multiple fora). The forum EMF-21 does not have a dummy; the results of this forum are included in the constant of the regression in the following section. Among the 49 observations, 14 are from IMCP and 6 from USCCSP.

4.3.4 Results

We present the results of two meta-regression models in Table 4.5. The first model (model 1) includes all variables that were described above. The second model (model 2) was derived by stepwise regression and only includes variables that are at least significant at the 10% level. Results of models 1 and 2 are presented for the years 2025 and 2050.

Model 1 explains more than 50 percent of the variance in the projected MAC across the studies, and it explains the MAC for 2025 a little better than those for 2050. The unexplained share of the variance is due to unobserved differences in, e.g., model structure and parameterization. The sign and significance of the independent variables are as follows.

The stabilization target and the baseline emissions have a significant effect on MAC, as we would expect. The signs of the coefficients are also as expected: an increase in the stabilization target reduces MAC, and an increase in baseline emissions increases MAC.

A multigas policy, offering "what" flexibility in climate policy, reduces MAC in 2025. This result is as expected, as we argued above. While the coefficient of multigas is still negative in 2050, it is no longer significant. Overall, the difference between single gas and multigas models appears smaller than might be expected. As argued in Tol (2006), the option to mitigate other greenhouse gases than carbon dioxide increases flexibility and so reduces costs. However, the only way to stabilize climate change is to reduce carbon dioxide emissions to zero. Therefore, the other greenhouse gas emissions have a substantial effect on costs only in the medium term. Furthermore, differences between models in the treatment of non-CO_2 greenhouse gases are large, and some models have many such gases while other models have few. The distance between baseline emissions and target emissions depends on the gases included in the analysis. Therefore, the multi-gas results are noisier than the single gas results, and this also explains the lack of significance in 2050.

The dummy variable indicating the assumption of induced technical change is significant for 2025, but not for 2050. What is remarkable, however, is that the 2025 coefficient seems to have an unexpected sign. The assumption of ITC seems to increase MAC rather than reduce it. This is not completely contrary to the conclusions of the IMCP forum, that emphasized the large differences among models regarding assumptions on, for example, existing market distortions, long-term investment behaviour and the nature of the technological options considered (Edenhofer, Lessmann et al. 2006). In general we might conclude that the inclusion of ITC in IAMs is still, to a large extent, in an experimental phase.

Contrary to the results of Fisher and Morgenstern (2005) we do not find significant effects for aggregation characteristics across our models. There is no significant difference between MAC from models with a high level of detail in terms of primary energy sources and regions and those with low levels of detail.

Table 4.5 Results of meta-analysis

	MAC2025				MAC2050			
	Model 1		Model 2		Model 1		Model 2	
	b	t	b	T	b	t	b	t
CONSTANT	10.623	5.218 ***	8.922	6.523 ***	10.594	5.480 ***	8.217	7.355 ***
TARGET	–.016	–4.565 ***	–.015	– 5.487 ***	–.013	–3.902 ***	–.010	–4.204 ***
BASELINE	.667	3.524 ***	.677	4.202 ***	.267	1.487	.308	2.124 **
MULTIGAS	–.643	–1.787 *			–.499	–1.459		
ITC	.586	1.908 *	.607	2.115 **	–.021	–.073		
REGIONS	.000	.030			.006	.593		
ENERGIES	–.061	–.869			–.017	–.247		
CGE	–.444	–.977			–.009	–.022		
IDO	–.791	–2.229 **	–.790	–2.930 **	–.822	–2.441 **	–.852	–3.433 ***
CCS	–.511	–1.302	–.636	–2.153 **	–.250	–.670		
IMCP	–.727	–1.338			–.764	–1.481		
USCCSP	.415	.785			.146	.290		
R2	.650		.602		.536		.477	
R2 (adjusted)	.533		.551		.381		.439	

* significant at 10% level

** significant at 5% level

*** significant at 1% level

There are also no significant differences between CGE and other models. The absence of a significant difference might be interpreted as a confirmation of the suggestion of Hourcade, Jaccard et al. (2006) suggestion of a convergence of the modelling approaches – or at least the results.

Intertemporal dynamic optimisation is significant and has the expected sign. Flexibility in choosing the optimal reduction path (“when” flexibility) seems to matter a great deal.

The difference between models that include backstop technologies and CCS is significant for 2025 (for Model 2), but not for 2050. The sign of the coefficients is negative, suggesting that backstop technologies and CCS reduce MAC in comparison to models that do not include these options. The signs seem to make sense, and are contrary to the results of Fisher and Morgenstern (2005).

Compared to the EMF-21 modelling forum, the models in the IMCP forum tend to report lower MAC, and the models in USCCSP tend to report higher values. The coefficients, however, are not significant. The lower values for MAC in IMCP are not due to their inclusion of ITC. In the first place, the data include IMCP both results computed with and without ITC. Furthermore, in the regression model ITC is already accounted for in a separate variable.

In sum, the meta-analysis suggests that differences between MAC across studies can to some extent be explained by differences in target and baseline, intertemporal optimization, the inclusion of non-CO_2 gases, and the inclusion of CCS or a backstop technology. Other technical features of models, such as type (CGE or not), ITC, and aggregation issues, have random effects on MAC. There is some influence of the Modelling Forum on the MAC results: the more “experimental” models of the IMCP forum tend to report lower values than the more mature and standard models that participated in the USCCSP forum. This gives some support to the critique of Tol (2006) on IMCP. The EMF-21 forum takes a middle position.

The estimated meta-regression functions can be used to predict MAC given specific values for the significant explanatory variables. We have tried to examine the association between MAC and TARGET based on the estimated meta-models (model 2). Figure 4.3 shows the association between MAC and TARGET for the full range of TARGET values in our database (350-550 ppmv). Figure 4.3 also shows the 95% prediction interval around the central prediction. The prediction interval quickly increases if we leave the range 450-550 ppmv where the bulk of our observations are. According the Figure 4.3, MAC for a stringent long-term target of 350 ppmv CO_2, which is more or less consistent with the EU’s 2°C target (see Table 4.3), could be between € 74 and € 227 in 2025 and between € 132 and € 381 in 2050.

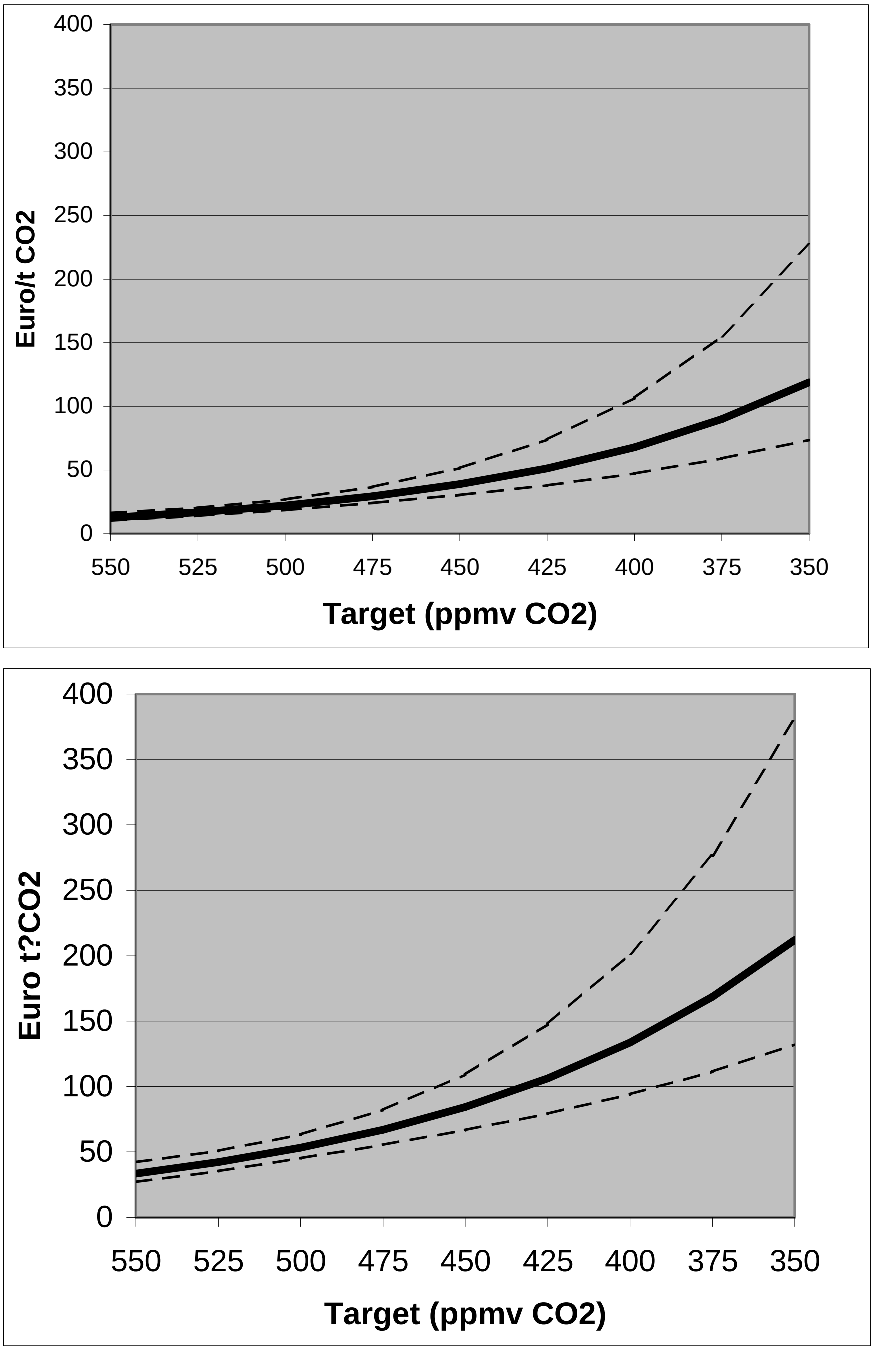

Figure 4.3 MAC as a function of Target (Upper panel 2025; Lower panel 2050)

As explained in the Introduction, our MAC estimate can be viewed as the carbon permit price in an idealized global emissions trading system. We can compare this "ideal global" MAC with two recent policy estimates at country and EU levels. The first relates to the United Kingdom's (UK) target of achieving a 60% reduction in greenhouse gas emissions by 2050[20], and the second to the recently announced policy targets of the EU

[20] The Energy White Paper "Our Energy Future – Creating a Low Carbon Economy" (2003).

(EC 2007; Elzen, Lucas et al. 2007).[21] Table 4.6 presents estimated MAC for the UK and EU targets in 2020 and 2050 (2050 for the UK only). Table 4.6 shows that estimated MAC (across the target range 550-350 ppmv) span the full range of the national estimates for both years.

Table 4.6 National and regional MAC (€^{2005}/tCO_2)

	2020	2050	source
UK	15 – 60	142 – 193	(Watkiss 2005)
EU27	23 – 93		(Elzen, Lucas et al. 2007)
MAC*	13 – 119**	34 – 212	This report

* MAC across the target range 550-350 ppmv CO_2.
** These values refer to 2025.

4.3.5 Conclusions and discussion

We have analysed information on MAC from 62 recent studies that assessed the economic impacts of meeting long-term stabilization targets of greenhouse gases in the atmosphere. All the studies computed a least-cost trajectory of global abatement efforts to meet such a target. The MAC assessed by these studies were shown to depend on the level of the stabilization target, the assumed emissions baseline, intertemporal optimisation, the choice of control variable (CO_2 only versus multigas), assumptions on future technological options (backstop and CCS), and, to a lesser degree, on the scientific “forum” in which the study was developed.

The estimated MAC can be considered as “idealized global MAC”: they assume a perfectly rational, efficient and global policy that would equate MAC across all sources of emissions at each point in time and would also result in an optimal trajectory of MAC over time. In less “ideal” settings, the MAC may well be substantially higher. We compared our “ideal global MAC” with MAC that were assessed in the context of real policy proposals in the UK and the EU and found that the policy-specific estimates and our central estimates are of the same order of magnitude. We also found, however, that the uncertainty of the estimates increases quickly if we move in the direction of more stringent targets.

[21] In January 2007, the European Commission proposed that the EU should (in the context of international negotiations) pursue the objective of a reduction of 30 percent in greenhouse gas emissions by 2020 (compared to 1990). Without international cooperation the EU should unilaterally commit to a reduction target of 20 percent in 2020 (EC, 2007).

Annex I Database of GHG stabilization studies

No.	Model	Platform
1	AIM	EMF-21, IMCP
2	AMIGA	EMF-21
3	GTEM	EMF-21
4	GEMINI	EMF-21
5	PACE	EMF-21
6	EDGE	EMF-21
7	EPPA	EMF-21
8	IPAC	EMF-21
9	SGM	EMF-21
10	WIAGEM	EMF-21
11	COMBAT	EMF-21
12	FUND	EMF-21
13	GRAPE	EMF-21
14	MERGE	EMF-21, USCCSP
15	IMAGE	EMF-21
16	MESSAGE	EMF-21, IMCP
17	MiniCAM	EMF-21, USCCSP
18	POLES	EMF-21
19	DEMETER-ICCS	IMCP
20	DNE21+	IMCP
21	E3MG	IMCP
22	ENTICE-BR	IMCP
23	FEEM-RICE	IMCP
24	GET-LFL	IMCP
25	IMACLIM-R	IMCP
26	IGSM	USCCSP

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5. Note on discounting for CASES

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In the context of global climate change, the practice of discounting has been and continues to be questioned by many economists and other policy analysts. It has raised a huge literature on discounting that we will not present here. The reader who is interested in this debate is referred to the book *Discounting and Intergenerational Equity* edited by Paul Portney and John Weyant (1999) or the survey article published in the *Journal of Economic Literature* by Frederick, Loewenstein and O'Donoghue (2002).

The first question to address is to know whether standard practice of discounting is relevant to the CASES project. As mentioned in the Stern review (2006), "*standard treatments of discounting are valuable for analyzing marginal projects but are inappropriate for non-marginal comparisons of paths*". As far as CASES concerns the energy sector, we can assume that the project's impacts will induce only marginal variations of the economic paths. Therefore, the general approach of discounting in social cost-benefit analysis can be applied to the project. From a theoretical perspective we can differentiate between two approaches of discounting in social cost-benefit analysis. The first approach is the social rate of time preference, which expresses the collective choice with regard to the appreciation of the future. The second approach is the opportunity cost of capital, which is, as we shall argue below, problematic for long-term investments. The two approaches can be combined in an approach called step-declining discounting in which the rate of return on capital is applied as discount rate for the first thirty year while the social rate of time preference is applied in the years thereafter..

5.1 Discounting around the world

The starting point of our note will be to present the applied discount rates for public projects around the world. It has to be noted that the practice of discounting varies with respect to countries according to the sector concerned and the planning horizon considered. So, one must be prudent in making comparisons between the different discount rates applied by different countries.

Some countries like Spain apply different discount rate according to the sector considered: 6% for the transport sector and 4% for projects related to water resources.

Great Britain recently revised its discount rate from 6% to 3.5% for projects with a planning horizon of 30 years. It has also considered applying a declining discount rate for very long-term projects with the discount rate falling to 1% beyond 300 years.

In Italy, different discount rates were applied in the north of the country (8%) and the south (5%) to take into account differences in economic performance. Actually, the discount rate is 5%.

The table below gives some picture of the rates applied in different countries.

	Discount rate	Planning horizon (years)
EU Commission	5%	
World Bank -Developing Country	10-12%	
Australia	6-7%	20-30
Canada	5-10%	20-50
Czech Republic	7%	20-30
Denmark	6-7%	30
France	8%	30
Germany	3%	variable
Hungary	6%	30
Italy	5%	
Japan	4%	40
Mexico	12%	30
Netherlands	4%	30
New-Zealand	10%	25
Norwich	5%	25
Portugal	3%	20-30
South Africa	8%	20-40
Sweden	4%	15-60
United Kingdom	3.5%	30
United States of America	3-7%	variable

Source: Rapport “Le prix du temps et la décision publique: Révision du taux d’actualisation public”. Daniél Lebègue, La Documentation Française, 2005.

5.2 Social rate of time preference

We must clarify, in the beginning, whether it is discounting utility or consumption that is at stake. If a social planner chooses consumption as the common unit then the consumption discount rate becomes the social discount rate. If utility is chosen as the common unit then the utility discount rate is the social discount rate. The rate of return on the common unit is in general the social discount rate: that is the rate at which the implicit value of the common unit varies over time. As mentioned by Heal (1998), “*utility discounting is appropriate when we are working with a general equilibrium model and general equilibrium consequences will follow from the choice under consideration. By* *contrast, discounting consumption is appropriate when we are working in a partial equilibrium context and the underlying growth path and resource allocation of the economy can be taken as given*".

Assuming the consumption as common unit, the social rate of time preference, δ, is written as:

$$\delta = \rho + \eta g \quad (1)$$

where ρ is the social rate of pure time preference, η is the elasticity of marginal utility of consumption, and g is the growth rate of per capita consumption. The social discount rate δ is the rate at which future consumption (or cash flows) is discounted. The social rate of pure time preference is the equivalent to the representative individual's rate of pure time preference. At individual level, it measures the degree of impatience (utility today is perceived as being better than utility tomorrow). Several economists criticize the existence or the fairness of the rate of pure time preference (Ramsey, 1928; Harrod, 1948; Pigou, 1924). For ethical reasons, many economists prefer to set the rate of pure time preference to zero for social cost-benefit analysis, finding it unfair to discount future consumption solely because of impatience (Cline, 1999). The second term at the right hand side of equation 1 is called the wealth effect. It supposes that people are expected to be better off in the future as far as η and g are positive and gives a basis for discounting. The elasticity of marginal utility of the consumption is the percentage at which the incremental value from consumption falls off. From empirical estimates, the elasticity of marginal utility is suggested to be in the range from one to two. Although a survey of Stern (1977) raised the possibility of a wide range of elasticities running from 0 to 10, with elasticities around 2 most supported, Cowell and Gardiner (1999) estimate 0.5 to 4 as the reasonable range, and Pearce and Ulph (1999) range η from 0.7 to 1.5. We can expect a positive growth rate of consumption per capita over the next two or three centuries. Therefore we can define a range of social discount rates depending on the assumptions made about its different components. The table below gives the suggested range of social discount rates.

We assume three levels of rate of growth of per capita consumption per year: 1, 1.5, 2. At the lowest level, one may think that we are already too optimistic about future growth rates and that the earth cannot sustain such economic expansion. Elasticities of marginal utility are assumed to range in the interval described above: 1 to 2.

Pure rate of time preference ρ	Elasticity of marginal utility η	Per capita cons. growth rate g	Social discount rate δ
0	1	1	1
0.5	1	1.5	2
0.75	1.5	1.5	3
1.5	1.5	1.5	3.75
2	2	2	6

For the pure rate of time preference, the different perspectives in the literature are considered with values ranging from 0 to 2. It is generally supposed that the impact of the impatience motive on the interest rate required by households for their saving returns is low, between 0 to 2% (Cline, 1993).

These assumptions on the values of the components of the social discount rate lead to a range of discount rates from 1% to 6%. From the theoretical literature, the lowest value of 1% seems to give the best chance for the distant future to be taken into account in project evaluation. The highest value at 6% reaches the claim of those who prefer the use of the opportunity cost of capital with real rates of return to capital of 6% to 8% or higher.

5.3 The opportunity cost of capital

Some economists suggest that for global efficiency the rate of return to investment in the public sector should be the same as the rate of return in the private sector. Therefore, the return of public investment should be calculated on the basis of the cost of capital in the private sector so that there will not be a crowding out of private investment by public investment. The social opportunity cost of capital has been defined to measure the best alternative use of funds in a public project. In a perfectly competitive world, the opportunity cost of capital can be confounded with the market interest rate. But under imperfect competition, there exists no unique interest rate or return rate that can account for the opportunity cost of capital. In a second best world, the appropriate discount rate depends on the type of imperfections prevailing on the market and on the distortions that cause the bad allowance of resources. Cline (1999) summarizes this approach by relating the social discount rate, δ, to the rate of return on capital, r:

$$r = \delta + w \qquad (2)$$

where w is the wedge caused by tax and other obstacles to complete clearing of the market for capital.

Diminishing returns to capital may guide the selection of real rates of return on capital on a hundred years long time scale. This will lead to consider lower discount rates than those currently applied. The problem with discounting the far future at today's rate of return on capital is that it assumes that the present generation and all intervening generations will keep intact an investment fund that is capable of continued real returns at today's level to generate a payment that will compensate future generation for damage inflicted. This commitment seems to be hardly credible (Arrow et al., 1996; Lind and Schuler, 1996). We do believe as stated by Cline (1999) that: "*because this commitment cannot be counted on, because the rates of return cannot be counted on, and because it is not even clear that there is a meaningful vehicle for storing this physical investment in a way that will generate goods and services relevant to the generation two centuries from now, discounting at today's rate of return on capital seems highly likely to stack the cards against future generations. This leaves the social rate of time preference as a far superior approach*".

5.4 Step-declining discount rate

The idea of this approach is to consider different arguments that are defended by the tenants of the two main approaches presented above. Step-declining discounting means to apply different discount rates at a declining space for a divided horizon. For example, for the first thirty years, we apply the opportunity cost of capital approach leading to a rate of return on capital of 6%-8%. This will coincide with the limited horizon that actual financial markets reach and can be seen also as a generational break point. After the first thirty year, the social rate of time preference can be applied. It is also possible to divide this part in two periods, a first period for which the highest value of discount rate is applied and a second period for after one hundred years for which the lowest value is applied. We can summarize this proposition in the following table.

	First 30 years	From 30-100 years	After 100 years
Rate of return on capital	6-8%		
Social rate of time preference		3%	1%

This approach is in the line of the compromise proposed by Cline (1999) to reconcile the tenants of the "prescriptive" (social time preference) and the "descriptive" (market interest rate) approaches. It is also in line of the report on discounting made in 2005 by the Commissariat Général du Plan (Lebègue, 2005), the former French government body for long-term policy.

The step-declining approach can be considered as a sensitive scenario.

5.5 Recommmendations and conclusions

1. We believe that private costs and external costs should be discounted at the same rate if these costs are subsumed in the same block of costs. Therefore, based on the arguments developed above we recommend the use of the social rate of time preference as the discount rate as far as external and public costs or benefits are considered. This will respect the principle of unity of discounting. If these costs

are separated into two different sets of analysis, one can argue the use of rate of return on capital for private costs and the use of the social rate of time preference for external costs. Although dual-discounting is one of the prominent debate on discounting in project evaluation. Dual-rate discounting is the use of different discount rates to evaluate consumption and environmental quality impacts of a project (Weikard and Zhu, 2005). The argument for the use of dual-discounting is that different trends of utility derived from different goods should impact project evaluation. However, the use of dual discount rates lacks a rigorous theoretical foundation. In fact, it is hard to see a reason why rates of pure time preference should be different for different types of goods. Weikard and Zhu (2005) explore the issue of dual-discounting and conclude that "*dual-rate discounting can serve as a pragmatic device to evaluate consumption and environmental quality aspects of a project or policy separately. In this case, a constant accounting price is used instead of current accounting prices which reflect the marginal rate of substitution between consumption and environmental quality at each point in time. To proceed in this way can simplify calculations, but it cannot change results. We have shown that dual-rate discounting with a constant accounting price is equivalent to discounting at a uniform rate using current accounting prices. Secondly, we have argued that non-substitutability is a strong case for true dual-rate discounting. If goods are not substitutable, then an accounting price cannot exist and each good must be evaluated separately and will be discounted at different rates. However, strictly speaking, this will not lead to dual-rate discounting as only one rate, the rate for the limiting good, is relevant for the decision-maker*".

2. We believe also that it would make sense to at least make two scenarios for discounting. One scenario will be concerned by the application of the social rate of time preference approach with a discount rate of 3%, which is consistent with most of the valuation studies of climate change. The second scenario will consider the step-declining approach presented above, which is becoming a widely used approach proposed in the literature in different contexts.

3. We also believe that the public decision-maker cannot have different discount rates per sector. This would mean that the public decider has different views of the future. Also, the allocation of resources must be consistent among sectors of the economy and should not discriminate between sectors. The social discount rate must be unique and applied uniformly to all public projects and to all sectors of the economy.

4. A risk premium should not be added to the discount rate. Risk and uncertainty must be treated separately in the assessment of costs and benefits for each project as far as for quantity and prices.

5. As far as possible, long-term environmental values of benefits and costs have to be calculated and integrated into the assessment of a project. Of course, the non-environmental costs and benefits have to be fully assessed, but they are not the ones that are most problematic.

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