Sofia Ahlroth
Avd för mijöstrategisk forskning – fms
KTH

Calculating damage values for ecosystem effects in Sweden

Introduction
Valuation of environmental effects is used in many impact assessments today, to make it easier to assess total effects from different projects and products. The valuations can be both monetary values and non-monetary weights. In cost-benefit analysis (CBA), cost and benefits from a project are compared by way of monetizing both costs and benefits. Effects on e.g. the environment, time use and health are monetized, thus making them comparable to the monetary costs and revenues of the project. Also in other systems analysis tools, such as life cycle analysis (LCA), Environmental Management Systems (EMS), Life Cycle Costing (LCC) and Strategic Environmental Assessment (SEA), valuing is used to assess the total impact on the environment from e.g. a product or a project, and to compare with the impact from alternative projects or similar products.

In the environmental systems analysis tools mentioned above, sets of generic values are frequently used. In Sweden, values for pollutants from transport are calculated regularly on the governments commission in the so-called ASEK projects (SIKA 2004). These generic values are routinely used in CBA:s in infrastructure planning in Sweden, a practice which has parallels in many countries (Navrud 2000). On the EU level, the ExternE (Externalities of Energy) project has been going on for several years calculating values for pollutants from energy generation and transports (www.externe.info). Some examples of other sets of generic values, both monetary and non-monetary, are Ecotax (Finnveden et al. 2006), Eco-indicator 99 (Goedkoop and Spriensmaa 2000) and the Ecoscarcity method (Ahbe et al. 1990). The available methods have been evaluated e.g. in (Finnveden 1997, Pennington et al. 2004). Each of the available sets of weights/values is found to have flaws, most often by way of lacking consistency, transparency and/or sufficient comprehensiveness.

The logic behind the weighting can vary with the purpose of the analysis. The two most usual criteria are 1) how damaging the impact from the pollutant is to the state of ecosystems and humans and 2) what impacts are regarded as the most important by the relevant population. The second alternative involves a subjective valuation similar to the forming of market prices. The first criterium can be defined fairly objectively within the same type of damage, e.g. global warming potential or eutrophication potential of a certain substance. But when it comes to weighing between two different types of effects, e.g. between climate change and eutrophication, some kind of subjective criterion is introduced, whether it is expert opinions, multiple choice methods or some other type of decision method that is used.

Willingness-to-pay studies are used to determine a kind of market price for non-marketed goods. What this kind of prices show is the current preferences of the survey population, given their awareness of the subject and the information given to them. The resulting values do not necessarily mirror the importance of the environmental impact from a scientist's point of view, but rather the current attitudes and preferences of the affected people. The likeness to market prices makes it an attractive method, since this makes it possible to compare to the values of marketed goods. In cost-benefit analysis, this kind of assessments is focus of the analysis. But weights expressing society's preferences is often relevant criteria when using the other decision tools mentioned above.

The ExternE project provides an extensive database of damage values for different pollutants, based on the criteria 1) above. However, health values and ecosystem values are estimated differently. Health effects are calculated with the impact-pathway approach, where the health impact of different pollutants are valued with willingness to pay (WTP) studies (see e.g.(Friedrich and Bickel 2001). Values for ecosystem effects are based on the so-called standard price method, where the costs for reducing the pollutants to a sustainable level are used as a proxy for damage values. This means that they are not comparable to the health values, and that no real welfare values for the ecosystem effects are estimated.

The purpose of this project is to calculate values for the ecosystem effects of nitrogen, sulphur and phosphorus for Sweden based on WTP studies. This will make it possible to compare willingness-to-pay estimates with existing values derived with avoidance costs for these pollutants, and see how large differences the two methods come up with. The ambition has been to use only Swedish studies, to capture the Swedish preferences for the environment and avoid the ambiguities with transfers between countries, that is shown in many benefit transfer studies (e.g. (Loomis 1992; Barton 2002; Ready et al. 2004)). To extrapolate values from valuation studies for a specific area to other regions and to the national level, a benefit transfer method called preference calibration is tested for eutrophication.

Valuation functions for acidification and eutrophication in Sweden are estimated. A survey of valuation studies was done using the EVRI database, the Swedish Value Base^{SWE}. Also, a literature search in EconLit, an economic journals database, was done. As mentioned above, the aim was to use primarily Swedish valuation studies, with supplement of suitable studies from similar countries in Europe if necessary. To derive actual valuation functions was found to be possible only for the case of eutrophication of the Baltic, since this was the only area where suitable valuation studies could be found. For eutrophication of freshwater, valuation studies for local eutrophication of the sea were used as a proxy. In the case of nitrate in groundwater and acidification, only a point estimate was possible to derive.

Method description

Benefit transfer of values from willingness-to pay studies is today very much an accepted practice when valuing health effects (e.g. in ExternE). For ecosystem effects, the situation is different. Dose-response functions are not as developed, and not only preferences about environmental issues but also the ecosystems themselves differ between countries, whereas the human body reacts much the same on exposure to pollutants no matter where people live (though the attitudes towards health risks differs quite much between countries, as was shown in (Ready et al. 2004)). However, it is possible to value endpoints like changes in flora and fauna, sight depth and water quality.

The both theoretically and intuitively most appealing way to value the environmental effects is to establish a direct damage value that can be set against the costs for reducing the damage in a cost-benefit analysis. This involves estimating the value individuals put on enhancing (or maintaining) environmental quality. This can be done through revealed preference studies, such as travel cost studies or hedonic pricing, or through stated preference studies, where contingent valuation (CV) is the method most often used.

Here, we will explore using a method labelled "structural benefit transfer" or "preference calibration" by the originators (Smith et al. 2002). The method allows using different types of valuation studies to produce a valuation function for a specific amenity. It produces benefit transfer values that are consistent with economic theory, by relating them to utility functions with well-known economic properties like declining marginal utility of quantity/quality and binding budget constraints. The starting point is to set up a consistent framework for consumer preferences by defining a utility function. The entities estimated in valuation studies, such as consumer's surplus and compensating variation, are assumed to be derived from the same utility function. The parameters in the utility function are calibrated using data from studies valuing the same object. Thus, instead of estimating a function using the full data set from the original studies, the analyst chooses functional forms and calibrates them against the benefit values from the studies, very much like in CGE modelling.

The method is attractive in that it provides a framework for the benefit transfer based on economic theory, producing theoretically consistent estimates. This meets the transparency criterion, since assumptions are explicitly defined. Double-counting risks are also eliminated. From a practical point of view, an additional advantage is that the analyst does not need access to the full data set of the valuation study.

In the following, we will briefly describe the logic behind the method. The description is largely taken from (Smith et al. 2000), where a more detailed description can be found.

The basic idea is to define a utility function and derive the measures estimated in valuation studies to this function. Travel cost (TC) studies, hedonic pricing (HP) studies and contingent valuation (CV) studies all give estimates of measures that can be derived from a utility function. Here, we will deal with measures from TC and CV studies. By linking the total values from a CV study to perceived changes in price for using the amenity, estimated in a travel cost study.

For the utility function, it seems natural to use one of the frequently used forms like CES (constant elasticity of substitution) or Cobb-Douglas. The utility function chosen here is of the Cobb-Douglas type. This is an obvious simplification, which easily can be relaxed by introducing some other, more realistic function. To capture the recreational values, a cross-product repackaging form (Willig 1978; Hanemann 1984) is used. Here, the value of the quality change is expressed as a reduced cost, i.e. the site visitor experiences that the effective cost of the trip is lower. The indirect utility function is written as

$$V = ((P-r(q))-\alpha m)K \qquad (2)$$

where P is a relative price (the price of the aggregate good normalised to 1) that represents the travel costs , r is a valuation function which describes how the environmental quality affects the effective price of a trip, q is an index for environmental quality (e.g. sight depth, pH value, fish catch), m is income and α, K are parameters.

Using Roy's identity, we can derive the demand for trips, X:

$$X = -\frac{V_P}{V_m} = \frac{\alpha m}{P - r(q)} \qquad (3)$$

From travel cost studies, we obtain an estimate of the marginal consumer surplus (MCS) for an environmental improvement. The MCS associated with the chosen utility function takes the following form:

$$\frac{\partial CS}{\partial q} = \frac{\partial}{\partial q}\int_{P_0}^{P_c} XdP = \alpha m \frac{r'(q)}{P_0 - r(q)} \qquad (4)$$

From (4), we can solve for r'(q), which shows how much the effective price of one trip is perceived to be reduced with a quality change:

$$r'(q) = \frac{\frac{\partial CS}{\partial q}}{\frac{\alpha m}{(P_0 - r(q))}} = \frac{\frac{\partial CS}{\partial q}}{X} \qquad (5)$$

The willingness to pay (WTP) for obtaining a certain improvement is defined as

$$V(m, P, Q0, \alpha) = v(m\text{-}WTP, Q1, \alpha) \qquad (6)$$

From the specification of the indirect utility function, WTP can be written as

$$WTP = m - \left(\frac{P - r(q_1)}{P - r(q_0)}\right)^{\alpha} m \qquad (7)$$

The WTP is thus linked to the experienced change in effective price for using the amenity, which is elicited from the travel cost study. Values of r'(q) and WTP is obtained from travel cost and willingness-to-pay studies, and with these data at hand, it is possible to calibrate the parameters needed. α can be solved from eq.(7) and calibrated using the WTP value. r(q) can be expressed as a function the quality index q and some parameter β, which is calibrated by inserting the marginal value per trip from a travel cost study in eq(5). For a derivation of the functional form for r(q) and comparisons to functions derived in the original studies, see (Ahlroth, forthcoming).

Case study: Eutrophication of the Baltic Sea

The Baltic Sea has for a long time been affected by eutrophication, due to high levels of nitrogen and phosphorus. The emissions come from all the countries around the Baltic, and also via air from other European countries. Among the effects of eutrophication are turbidity, smaller sight depth, more algae blooms in spring and summer and less biodiversity (Naturvårdsverket 2005). The blooms of the toxic blue-green algae is a growing problem, that was at its most severe ever in the summer of 2005 (Länsstyrelsen 2006).

Eutrophication is one of the areas that are best researched in terms of valuation studies in Sweden. Reduction of the eutrophication of the entire Baltic Sea was valued in a contingent valuation (CV) study, (Söderqvist 1996) and a travel cost (TC) study, (Sandström 1996), which were done in the same context. Both use sight depth as the variable with which to describe the eutrophication level, and value the reduction of nutrients to a level that is perceived as sustainable. Turbidity is an immediately visible effect of eutrophication, which is also linked to eutrophication levels in different quality class systems (see e.g. Swedish EPA, www.naturvardsverket.se), which is why sight depth is often used as the describing variable. (Söderqvist and Scharin 2000) and (Soutokorva 2005) valued a halving of the emissions to the area, corresponding to a sight depth improvement from 2 to 3 meters. A similar CV study was done for the Laholm Bay, (Frykblom 1998). The studies have been tested against the Swedish EPA:s quality criteria (Naturvårdsverket 2005), and they have all been found to be of good quality. The two sets of studies value different amenities, since the first set encompasses a much larger area, with a statistical selection of the whole Swedish population, whereas the second set values the local environment for the inhabitants of the two counties adjacent to the Stockholm archipelago, Uppsala and Stockholm, and to the Laholm Bay. Having these valuations studies, the preference calibration method described above can be used to derive values for reducing eutrophication.

The eutrophication levels vary along the Swedish coast. To attach as correct local values as possible, willingness-to-pay (WTP) functions across different quality levels have been calculated for both sets of valuation studies. For each WTP function, both a travel cost study and a contingent valuation study has been used.

The calibrations constants α and β are calculated for two sets of studies, one for the Stockholm archipelago and one for the whole of the Baltic Sea. The values for α and β for the Stockholm Archipelago case is calibrated given the data from the travel cost study (Soutokorva 2005)and the CV study (Söderqvist and Scharin 2000). The valuation function becomes

$$WTP = m - \left(\frac{P(1-0{,}8\left(1-e^{-0{,}2q1}\right)}{P(1-0{,}8\left(1-e^{-0{,}2q0}\right)} \right)^{0{,}03} m$$

with P= 300 and m = 189 000. The values of α and β are thus 0.03 and 0.2, respectively. The values of the calibrations constants α and β for reducing the eutrophication level in the entire Baltic Sea are 0.09 and 0.12, given P = 1100 and m = 178 000.

The functions are shown in figure 1. As expected, the value for improving sight depth from 2 to 3 meters is higher for the entire Baltic than for only the Stockholm archipelago; for Stockholm, it is about SEK 600 per person and year[1] (discounted present value) and for the Baltic, it is about SEK 1500.

[1] All values are in 2005 SEK, discounted present value for a period of 20 years and a discount rate of 4 %.

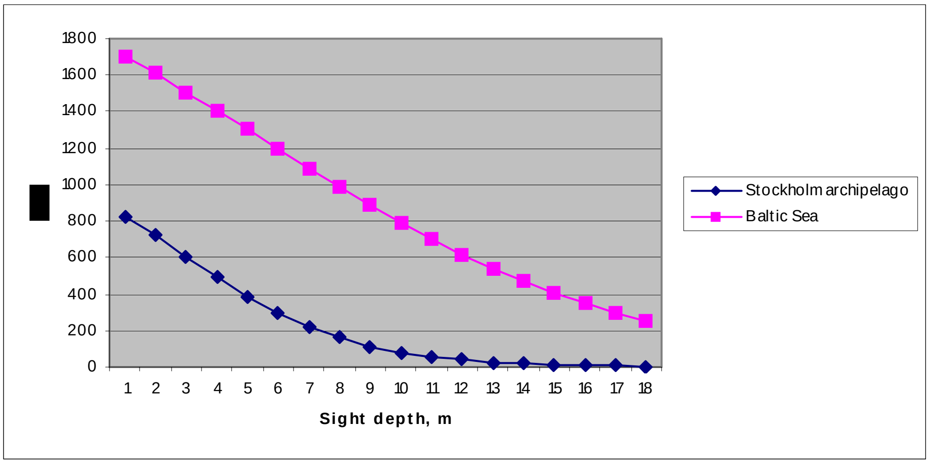

Figure 1. Marginal willingness to pay for reducing nutrient concentrations, represented as different sight depths.

Also, the value for improving sight depth with one metre is higher when the quality is lower, implying diminishing marginal utility of quality. For example, water quality and thus average sight depth is larger in the county of Kalmar than in the Stockholm archipelago; it averaged about 5 meter in 1996 (SMHI 1996). An improvement from 4 to 5 meters sight depth would yield a value of SEK 470 per person, which is about two thirds of the value estimated for Stockholm archipelago. In *Table 1*, the sight depth improvements of the coastal zones to the level that is considered to be approximately a "natural" state in the different basins (www.naturvardsverket.se) are shown, as well as the resulting value per person and total value per county. The values are estimates from the Stockholm studies, which were directed only to inhabitants in the nearby counties. Consequently this estimate is only used for coastal counties. In the northern parts of the Baltic, the Bothnian Bay and the Bothnian Sea, sight depths are not affected by eutrophication but by other factors. Therefore the value is set to zero for these counties as well.

Table 1. Regional values for improving sight depth to a "natural" level

	Inhabitants Age 18-64	Sight depth, meters	Improvement to	Value per person	Total per county
00Riket	6 626 350				
01Stockholms län	1 388 953	2	5	1370	1 902 865 610
03Uppsala län	223 573	2	5	1370	306 295 010
04Södermanlands län	190 189	2,5	5	1370	260 558 930
05Östergötlands län	304 404	3,4	5	850	258 743 400
06Jönköpings län	236 379				0
07Kronobergs län	130 067				0
08Kalmar län	170 989	4	5	470	80 596 396
09Gotlands län	42 187				0
10Blekinge län	111 099	2,9	5	1130	125 541 870
12Skåne län	858 619	2,9	5	1130	970 239 470
13Hallands län	205 162	4,5	6	723	148 332 126
14Västra Götalands län	1 118 539	4	6	990	1 107 353 610
17Värmlands län	200 757				0
18Örebro län	199 761				0
19Västmanlands län	191 211				0
20Dalarnas län	200 915				0
21Gävleborgs län	202 905	5	0		0
22Västernorrlands län	179 372	5	0		0
23Jämtlands län	92 873				0
24Västerbottens län	190 738	5	0		0
25Norrbottens län	187 658	5	0		0
				Totalt	**5 160 526 422**
				Average per capita	780

= inland counties

The aggregate value for reducing eutrophication along the Swedish coast was estimated to SEK 5 billion per year. To reduce total eutrophication of the Baltic, using the Baltic studies, was estimated to an additional 14 billion.

These values can be compared with the value of the original studies aggregated to national values. If the mean value per metre improvement was used, i.e. no differentiation for different quality improvements were made, a national value for reducing eutrophication along the coast would become SEK 7 billion, and the value for the entire Baltic SEK 26 billion. In total the estimate from the valuation function is 14 billion smaller than the simple unit transfer, or about 40 % smaller.

Calculating values per pollutant

What is needed in systems analysis tools like CBA and LCA and for modelling exercises with economic models, is a value per kg pollutant. Ideally, a dose-response function would be needed to allocate the total values estimated above to the polluting substances. At the Department of Systems Ecology at the Stockholm University, such a function has been estimated for sight depth. Phosphorus was not found to be significant in their estimations. Using this function would consequently allocate the whole value estimated above to nitrogen.

However, for the eutrophication of the Baltic, it is clear that phosphorus has an important role. The last few years there has been a vivid debate on the influence of nitrogen and phosphorus on the eutrophication of the Baltic, where some experts have claimed that too much attention has been put on reducing nitrogen, resulting in too little efforts to reduce phosphorus, which is the limiting factor for the increasingly frequent blue-green algae blooms in the summer (Naturvårdsverket 2006) However, for the blooming of other algae species in the springtime, nitrogen is the limiting factor (Naturvardsverket 2005 a) . Allocating a damage value to both nitrogen and phosphorus thus seems reasonable.

One way of allocating a total value between nitrogen and phosphorus is to use the characterisation factors developed e.g for life-cycle analysis. Characterization factors are used to quantify the potential of different substances to contribute to various environmental impact categories, the most well-known example being Global Warming Potential for greenhouse gases. According to the standard set in Handbook on Life Cycle assessment (Guinée 2002), 1 kg phosphorus has 7 times more eutrophying potential than 1 kg nitrogen. These characterization factors have been used for scaling the amount of phosphorus and nitrogen emitted in Sweden, to find a relationship with which to divide the total damage value between the two pollutants. The resulting weights put on each pollutant are shown in *Table 2*. A sensitivity analysis using other weights is shown at the end of this section.

Table 2. Characterisation factors

	N	**P**	**NOx**	**NH3**
Phosphate = 1	0,42	3,06	0,16	0,3
N = 1	1,00	7,29	0,37	0,71
Swedish emissions per year, kg	85 000 000	3 550 000	56 538 462	28 000 000
N-equivalents, kg	85 000 000	25 864 286	21 000 000	20 000 000
Relative weights	0,56	0,17	0,14	0,13

To find a value per kg pollutant, there is one more choice to be done: should the damage value be divided by total emissions, i.e. an average value, or by the damaging part of the emissions (the amount that is above the amount that nature can sustain without damaging effects), i.e. a marginal value. When using the abatement cost method, average cost per kg pollutant for reducing emissions to a level that is seen a sustainable, is used as a generic value. No weighting is undertaken to account for the fact that only part of the emitted substances need to be cut down. This corresponds to ascribing the marginal damage value to all of the polluting substances emitted, i.e. to divide the total damage value by the amount above the critical load.

Either a critical load concept or a political target decision must be used both when valuing with abatement costs and with damage costs, since some decision must be made as to how much the emissions need to be cut down. In the CV studies used here, the need for reduction was specified in the valuation survey instruments. For the local values, we use these amounts, since they represent the critical load envisaged by the experts consulted when the surveys were constructed.

To calculate a generic value on the national level is more complex. The Swedish coastal waters is likely to be more affected by reduced emissions from Sweden than reductions in a country on the other side of the Baltic, since the Swedish emissions to water land right at the coast, whereas foreign emissions are diluted before arriving at the Swedish coast.[2] Therefore reducing a kg nitrogen of Swedish emissions has a higher value than reducing e.g. a Polish kg nitrogen. On the other hand, Swedish coastal waters can not be restored to a "natural" state (taken to be the state of the fifties) if other countries do not reduce their emissions too, so using only the local value for reducing Swedish emissions would also be misleading. This dilemma will be approached by calculating different values for local waters and for the entire Baltic Sea.

The value estimated for the Stockholm archipelago could reasonably be seen as a subset of the total value for reducing eutrophication in the Baltic Sea as a whole. The nutrient emission reduction valued in the Stockholm study is just one percent of the reduction required to achieve the enhancement in the Baltic study. In both sets of studies, the amount of emissions that will be reduced is specified in the survey instrument. In the Stockholm study only local emissions will be reduced. Mean WTP per kg pollutant is therefore much higher in the Stockholm study than in the Baltic study, even though the sight depth improvement is smaller. The Stockholm study as well as the similar Laholm study was distributed only to the inhabitants of the area. The elicited values can therefore be expected to have a higher use value than the values in the Baltic CV study, where the non-use value is probably a larger part of the value than in the local studies. To deal with this, and also avoid double counting from using both the local and Baltic studies, a regional division is done when aggregating to a total value. The population along the coast is ascribed both a value for improvement of local areas and a value for better water quality in the entire Baltic Sea, the latter reduced by the amount stated for the local environmental quality. The population in the inland counties is ascribed only the total Baltic value.

Table 3. Generic values per kg pollutant emitted in Sweden . SEK/kg.

	N	P	NOx	NH3
Local value for pop. in coastal counties in Sweden	24	248	2	3
Baltic value for pop. in coastal counties in Sweden	7	84	0	0
Baltic value for pop. in inland counties and Northern Sweden	3	33	0	0
Baltic value for pop. in other countries around the Baltic	13	104	3	3
Total value	**47**	**469**	**5**	**6**

These values are generic values for a kg pollutant emitted anywhere in Sweden, calculated on the assumption that 70 % of the N emissions and 12 % of the NOx emissions end up in the Baltic (data from HELCOM(2005) and retention calculations from the TRK project (TRK 2007)). For phosphorus, no retention estimations are available, so here the gross emission is used. Given the location, a more precise calculation could be done using both regional values and site-specific assumptions on the percentage of the emissions that end up in the sea.

In the table below, values for a kg pollutant deposited in the sea is shown, both national average and values for pollutants deposited in two regions of different sizes: the south drainage basins and the Stockholm archipelago. The regional values are calculated using site-specific values (see appendix). As can be seen, the values differ quite much. The differences are due to different quality levels in the areas and in the amount of people that are affected.

[2] This notion has been disputed by e.g. …. . If all emissions to the Baltic are treated as equally eutrophying also in the inner archipelago, the resulting values per kg pollutant change quite significantly.

Table 4. Generic value per kg pollutant deposited in the Baltic. SEK/kg

	N	P	NOx	NH3
Average for all drainage basins around Sweden	61	469	19	31
South drainage basins	70	356	26	50
Stockholm archipelago	128	729	47	91

Sensitivity analysis

How to allocate the damage value to different pollutants is a delicate question. In recent years there has been a vivid debate on whether it is phosphorus or nitrogen that is primarily to blame for the eutrophication effects in the Baltic. In the above calculations, the damage value is allocated to the eutrophying pollutants using characterisation factors. In the sensitivity analysis shown in *Table 5*, the difference for the value per kg N and P is shown for four different weighting choices, with a weight of 100 to 33 percent allocated to nitrogen. The values for NOx (NO2) and NH3 are calculated using atom weights as conversion factor in the first four columns. The characterization factors give the eutrophying potential of all forms of nitrogen, so in that case those factors are used.

The calculations show that although the difference is large in percentage points between the largest and the smallest values for a certain pollutant, they still are in the same order of magnitude, except for phosphorus. The case where the total value is allocated to nitrogen and none to phosphorus is quite extreme and is probably not a likely choice unless pure sight depth is the object of valuation, not eutrophication. Setting that option aside, we can see that the largest values for the various forms of nitrogen are about twice as high as the smallest. It may influence the results of an impact assessment or cost-benefit analysis quite significantly and should therefore be explicitly shown and the choice commented upon.

Table 5. Values per kg pollutant using different weights

	N 1/3, P 2/3	N ½, P ½	N 2/3, P 1/3	N 100 %	Using characterisation factors*
P	2042	1532	1021	0	469
N	34	50	67	92	61
NO2	10	15	20	28	19
NH3	23	35	47	64	31

** Characterisation weights: P 0.17, N 0.56, NO_2 0.14 and NH_3 0.13*

The values in the previous section are all calculated with equal value per kg substance, i.e. an average damage value is ascribed to each kg pollutant. A different approach would be to divide the total damage value on only the amount of pollutant that is above the critical load. Once the emissions are lower than the critical load, the value is zero, but up till then each unit is ascribed the marginal damage value. As can be seen in *Table 6*, the difference is about a factor 2.5, except for phosphorus where the value is about four time larger. The conclusion is the same as in the previous case. The choice could be decisive for the results and should thus be explicitly shown and the choice commented upon.

Table 6. Average and marginal value: total damage divided on total deposition and deposition above critical load

	N	P	NOx	NH3
Average value	47	469	5	6
Marginal value	115	1874	12	15

Freshwater

There are no Swedish valuation studies for eutrofication of freshwater available. As mentioned above, there are several studies for the WTP for fish catches in recreational angling. However, a eutrophied lake is affected in many ways that is not attractive to humans: turbid water, algae blooms, overgrowth of seaweed and less biodiversity, including fish. To include just the effects on fish would be a serious underestimate.

A second-best solution would be to use studies from other Nordic countries, since both the nature and people's relation to it are similar in these countries. There is a Danish study for eutrophication of the Randers Fjord in Denmark (Atkins and Burdon 2005). They valued increased transparency of about 2.5 to 3 metres to about SEK 1650 per capita and year. In Norway, (Bergland et al. 1995) in a study of two selected rivers estimated values of SEK 2600 and 3500 per year and capita respectively. The quality change was specified as improving water quality by one quality class, which corresponds approximately to an improvement of about 2 metres, according to the Norwegian classification criteria followed in the study (NIVA 2003).

The local value for Stockholm archipelago was SEK 650 for a 1 metre improvement. The valuation study of the Laholm Bay mentioned above yield a value of SEK 800 for improving the water quality one step on the quality ladder, which approximately corresponds to a sight depth improvement of 1.5 metres (www.ma.slu.se/Miljotillst/Eutrofiering/Trofiklassning.ssi).

These Scandinavian valuation studies thus give values that ranges from SEK 650 to1700 per metre sight depth improvement. The Norwegian values are much higher than both the Danish and the Swedish values, though the values in the cited study are much lower than earlier, similar studies in Norway, e.g. (Magnussen 1992). They are even higher than the Swedish value for reducing eutrophication in the entire Baltic Sea, which was about SEK1500 per metre.

When deciding which value to use for lakes and rivers, two things are important to note. Firstly, quality varies a lot between different lakes and rivers, so that substitutes are easier to find for freshwater sites. Secondly, in Sweden the archipelago areas are usually more highly valued than inland waters; for instance, real estate values in Sweden are much higher at the seaside than inland (www.ssd.scb.se). We will therefore not use the Norwegian and Danish estimates, since they appear to be too high in comparison to the Swedish value for the Baltic. Instead, as relatively conservative estimate, we will use the valuation function estimated from the Stockholm Archipelago studies.

The computations are done for each of Sweden's 21 counties, using eutrophication mappings for each county (Naturvårdsverket 1995). The southern part of Sweden has higher levels of eutrophication, while there are only few eutrophied lakes in the northern parts. The national aggregate amounts to SEK 8,670 million.

Lakes and rivers in Sweden are P regulated, i.e. phosphorus is the limiting substance. Nitrogen deposition thus does not have any impact on eutrophication levels (Naturvårdsverket 2003). The eutrophication values for freshwater is therefore allocated only to phosphorus. The estimate per kg P becomes SEK 2400.

Nitrate in groundwater

High nitrate (NO_3) content in drinking water is carcinogenic and causes health problems to infants. These effects have been valued by (Silvander 1991) and (NIER 1998). Both studies give a value of around SEK 400 per person and year to avoid nitrate levels in groundwater above recommended limits. This is equal to a total value of SEK 2700 million per year for the population between 18 and 64 years of age. Divided on annual nitrate deposition, it amounts to SEK 40 per kg nitrate. This corresponds to SEK 33 per kg nitrogen and SEK 10 per kg nitrogen dioxide (NO_2), using the atom weights for conversion.

Acidification

For acidification, few valuation studies exist in Europe. There are no studies where the willingness to pay is related to a quality measure. The valuation study that encompasses most of the effects of acidification in Sweden is (NIER 1998). In that study, respondents were asked to state their willingness to pay to eliminate acidification of lakes and forests through emissions reductions and prudent liming of some heavily acidified lakes and forest areas. The estimated average value was SEK 890 per person for eliminating acidification of freshwater lakes and watercourses, and SEK 420 per person for eliminating acidification of forests. In total, this amounts to SEK 7,9 billion for the population between 18 and 64 years of age.

The main acidifying substances are sulphur dioxide (SO_2), nitrogen oxides (NO_x) and ammonium (NH_3). Like in the eutrophication case, the allocation of the total value for acidification to the affecting substances is not self-evident. Chemically, the acidifying potential of sulphur and nitrogen is the same, which cold be an argument for allocating the damage costs equally between them. However, in practice sulphur has a stronger acidifying impact than nitrogen on the ecosystems. In (NIER 1998), two thirds were allocated to sulphur and the remaining third to nitrogen. This assumption was based on expert assessments, with no written source given. In life cycle analysis, so-called characterisation factors are used, which gives weights to substances for their contribution to different impact categories, e.g. acidification, eutrophication and the forming of ozone. In the Handbook on Life Cycle Assessment, operational guide to the ISO standards (Guinée 2002), best estimates of the acidification potential of SO_2, NO_x and NH_3 are provided. They are set to 1.2, 0.5 and 1.6, respectively. In *Table 7*, values per kg pollutant are shown using different assumptions about their acidifying potential.

Table 7. Generic values for SO_2 , NH_3 and NO_x using different weights for acidifying potential. Values in 2005 SEK

	Equal acidifying potential		NIER 1998		LCA characterisation factors	
	Weight	Value per kg	Weight	Value per kg	Weight	Value per kg
SO_2	0.33	15	0.43	30	0,30	15
NO_x	0.33	35	0.17	17	0,35	16
NH_3	0.33	41	0.40	20	0,35	29

As can be seen, the resulting values differ by a factor two. There is really no objective way to discriminate between the weights. In the following, we choose the same method as in the case of eutrophication and use the values computed with the characterisation factors, being an alternative that has gone through a scrutinizing process by several experts when earning its place in the handbook.

Comparison with estimates in other studies

Adding up the damage values for acidification and eutrophication gives total damage values for each of the selected pollutants. They are given in *Table 8*.

Table 8. Damage values for sulphur, nitrogen and phosphorus per kg pollutant. Site independent average. 2005 SEK

	Eutrophication of the sea	Eutrophication of freshwater	Nitrate in groundwater	Acidification	Sum
N	60		30		90
NO_3			40		40
NO_x	5		10	15	30
NH_3	4			30	34
SO_2				15	15
P	470	2400			2870

In *Table 9* below, the standard values for NO_x and SO_2 in Sweden, as recommended by the ASEK working group, are shown. The local values include primarily health effects. The regional values are proxies for both health effects, ecosystem effects and other effects (e.g. corrosion, crop yields), calculated with mitigation costs.

Table 9. Current ASEK values for local and regional effects.

	Local value, Stockholm	Local value, Falun	Regional value
NOx	30	11	62
SO2	275	96	21

Source: SIKA (2005).

To compare the values estimated in this study to the existing values shown above, health effects should be added. In the table below, values for health effects and damages to crops from two different studies are added to the values for ecosystem values computed in this study. The two studies are:
1) the BeTa database, produced in the EU project MethodEx, an offspring from the ExternE project (http://www.methodex.org/introduction.htm)
2) the so-called all-modes study, where values for Sweden were calculated using the ExternE method (Nerhagen and Johansson 2003).

Table 10. Combining values for health, crop yields and ecosystem effects.

	Ecosystem values	BeTa database version 1.07 (Health+crop effect) SEK/kg	All-modes study SEK/kg	**Sum, BeTa + ecosystem**	**Sum, all-modes + ecosystem**
NH3	*34*	102		**136**	
NOx	*30*	35	16	**65**	**46**
PM2.5		205	226	**205**	**226**
SO2	*15*	49	23	**54**	**38**
VOC		5		**5**	
N	*90*			**90**	
P	*2870*			**2870**	

We can conclude that the value for NOx happen to coincide exactly with the abatement cost proxy in ASEK, when the BeTa values are used. The value for SO2, on the other hand, is more than twice as large than the avoidance cost estimate of ASEK. The values estimated in the all-modes study are generally lower than the BeTa values (version 1.07, 2006). Adding those health values to our ecosystem values, we see that the total value per kg NOx is lower than the ASEK value, and the SO2 value is less than double the ASEK value. The ASEK values being average abatement costs, we can conclude that which health effect values that are used is decisive for if the benefit values of reducing a kg pollutant is cost-effective or not.

To these values can be added the values for chemicals and heavy metals from the BeTa study, thus getting a more comprehensive set of weights showing willingness-to-pay values for reducing these substances. For consistency and comprehensiveness, the BeTa health and crops values for the other pollutants are chosen before the all-modes study. The resulting set of values/weights is listed in *Table 11*.

Table 11. Values based on willingness to pay for health, crops, eutrophication and acidification.

	SEK/kg
NH3	136
NOx	65
PM2.5	205
SO2	54
VOC	5
N	90
P	2870
Arsenic	720
Dioxin	333 000 000
Lead	5 400
Mercury	54 000
Cd	153
Cr (typical mix of Cr species)	126
CrVI	990
Formaldehyde	0.5
Ni	15

The ExternE values are taken from the latest version (version 1.07, 26th July 2006) of the BeTa study from the MethodEx project(http://www.methodex.org/introduction.htm). The values for chemicals and heavy metals are reference values for Europe, while the other health and crop effects are computed for Sweden.

Damage values by vehicle

The damage values can be recalculated by vehicle by using the specific emission rates of different vehicles. Using the emission factors from the Swedish ExternE project (Johansson and Ek 2003; Nerhagen et al. 2005), we get the following generic values per vehicle kilometre (km).

Table 12. Damage values for different vehicles. Site independent average. 2005 SEK

	SEK/g	SEK/km Gasoline cars	Diesel cars	HDV > 3,5 ton	Buses	Two-wheelers
NOx	0,027	0,025	0,033	0,26	0,28	0,0022
NH3	0,034	0,0019	0,00003	0,00010	0,00010	0,00007
SO2	0,015	0,00014	0,00006	0,00027	0,00026	0,00004

Abatement costs for NOx and SO2 per vehicle type from the All-modes study (Nerhagen and Johansson 2003) are shown in the table below.

Table13. Abatement cost values for different vehicles in the all-modes study. SEK/kg

	Gasoline cars	Diesel cars	HDV > 3,5 ton	Buses	Two-wheelers
NOx	0,015	0,032	0,19	0,23	0
Acidifying substances	0,03	0	0,23	0	0

The reason that the value for acidifying substances is zero for three of the vehicle types is that abatement costs only arise after a certain level, and this limit is only passed by gasoline cars and heavy duty vehicles. The results for NOx are very similar to the values estimated in this study, while the values for acidifying substances, when not zero, are much higher, due to the high mitigation costs. It would seem that abating NOx is cost-effective even when considering only eutrophication effects, while abating SO2 and NH3 is not motivated by the acidification of ecosystems alone.

Site-specific values for Stockholm are shown in *Table 14.* Although trucks and buses have larger emission factors, the largest part of emissions come from private cars. In Swedish cities, they account for 80 percent of the road traffic. The total distance covered by road vehicles in Stockholm in a year is 3963 kilometres, amounting to about 350 million SEK per year, of which private cars account for 290 million SEK.

Table 14. Damage values for different vehicles in Stockholm. 2005 SEK

	SEK/g	SEK/vkm Personbil bensin	diesel	Lastbilar > 3,5 ton	Bussar	Motorcyklar
NOx	0,047	0,043	0,058	0,46	0,49	0,0038
NH3	0,091	0,00500	0,00009	0,0003	0,0003	0,0002

Concluding comments

The willingness-to-pay values estimated in this study are generally in the same order of magnitude as the abatement cost values from other studies (with the exception of the costs and benefits of abating acidifying substances from gasoline cars and heavy duty vehicles). The exception is phosphorus, where the value is much higher than the values in other value sets, e.g. Ecotax (Finnveden et al, 2006) and EPS (Steen, 1999). Here the difference is several orders of magnitude higher, which may influence the outcome of a cost-benefit analysis.

In percentage points, the difference can be rather large also for the other pollutants, and this may be decisive in cost-benefit analyses of abatement measures. The sensitivity analysis for eutrophication of the sea showed that the chosen way to allocate the values to different pollutants that contribute to the same ecosystem effects is decisive, as well as the choice of average or marginal value. Which value that is appropriate to use depends on the context. The average value shows the average ecosystem effect of a kg nitrogen that is deposited on Swedish soil or in the sea around Sweden. This value can be used for site independent analyses. The marginal value on the other hand shows the value of the damage of a kg nitrogen ending up in the Baltic that is above the critical load. This value could be used e.g. for site specific analyses of the benefit or reducing the deposition of nitrogen and phosphorus.

It has not been possible to use the preferred benefit transfer method, relating benefits to quality levels, for other ecosystem effects than eutrophication of water. The values for sulphur and nitrate in groundwater are point estimates from rather simple willingness-to-pay studies. They are national values that are not differentiated for different quality levels, regions or sites, and can thus not be adjusted for a changing situation. This is due to lack of valuation studies. Even when extending the search to Europe and the US, no appropriate studies were found.

Another issue is to find updated data. The most up-to-date willingness-to-pay studies used in this study are about ten years old. It would naturally be preferable to have newer data. Even though the numbers have been converted to the price level of 2005, the situation is so different, with more frequent and more severe algae blooms, a vivid environmental debate, and higher income levels in society, that the values would probably be different were the surveys done today. It is also clear that more valuation studies needs to be done for other environmental effects, such as eutrophication of soil and acidification. Also, with the development of methods for valuations studies, the quality of the data is usually better the newer the studies are.

Since there are no exact dose-response function for ecosystem effects similar to the exposure-response functions for health effects, the ecosystem damage from a kg pollutant must be calculated "backwards" from the aggregate value of eutrophication related to the total deposition in the water area in question. This invokes two difficulties. Firstly, the critical load concept is not undisputed and it is not unproblematic to determine which deposition levels that are sustainable. Secondly, and related to the first issue: how to allocate e.g. the eutrophication effect to the impacting pollutants is also a matter of dispute. The viewpoint taken here is that the calculations should be very clear on how they treat these matters, and that it should be easy to recalculate the values using other assumptions.

Finally, an important issue when valuing environmental and other vital good sand services is the use of willingness-to-pay studies in itself. This is a long discussion and it would lead too far to go into it in depth here. Suffice it to say that it is important to recognize that the values elicited in this kind of study is not the absolute value of nature, but a marginal antropocentric value, i.e. showing the value to humans given the current scarcity situation. It is a true market value in the sense that it shows how much the consumers are willing to pay in relation to other goods and services given current supply and level of information.

1. References

Ahbe, S., A. Braunschweig and R. Müller-Wenk, 1990. Methodik für Oekobalnzen auf der Basis Ökologisher Optimierung. Schriftenreihe Umwelt Nr.133, BUWAL, Bern.

Atkins, J. and D. Burdon, 2005. An inital economic evalutaion of water quality improvements in the Randers Fjord, Denmark. Marine Pollution Bulletin Article in Press.

Barton, D. N., 2002. The transferability of benefit transfer: contingent valuation of water quality improvements in Costa Rica. Ecological Economics 42: 147-164.

Bergland, O., K. Magnussen and S. Navrud, 1995. Benefit transfer: Testing for Accuracy and Reliability. Comparative Environmental Economic assessment: Meta Analysis and Benfit Transfer. R. Florax, N. Nijkamp and K. Willis, Kluwer Academic Publishers.

Finnveden, G., Eldh, P. and Johansson, J., 2006. Weighting in LCA based on ecotaxes. Development of a mid point method and experiences from a case study. International journal of LCA Special issue honouring Udo de Haes.

Friedrich, R. and P. Bickel, 2001. Environmental External Costs of Transport. Springer Verlag, Berlin Heidelberg, Germany.

Frykblom, P., 1998. Halved emissions of nutrients - what are the benefits? A contingent valuation method survey applied to Laholm Bay. Question in the Contingent Valuation Method - Five Essays. PhD Thesis. Uppsala, Swedish University of Agricultural Sciences.

Goedkoop, M. and R. Spriensmaa, 2000. The Eco-indicator 99. A damage oriented method for Life Cycle Impact Assessment. Methodology Report. Report no 1999/36A, Ministry of Housing, Spatial Planning and the Environment, The Hague.

Guinée, J. B., Ed. 2002. Handbook on Life Cycle Assessment. Operational Guide to the ISO Standards. Eco-efficiency in Industry and Science. Kluwer Academic Publishers, Dordrecht/Boston/London.

Hanemann, M., 1984. Discrete/Continuous Models of Consumer Demand. Econometrica 52(3): 785-797.

Johansson, H. and M. Ek, 2003. Emissions from transport in Sweden. TFK rapport 2003:5E,

Loomis, J., 1992. The evolution of a more rigorous approach to benefit transfer: benefit function transfer. Water Resources Research 28: 701-705.

Länsstyrelsen, S. L., 2006. Informationscentralen för Egentliga Östersjön. Årsrapport 2005., Stockholm.

Magnussen, K., 1992. Valuation of Reduced Water Pollution Using the Contingent Valuation Method: Testing for Mental Accounts and Amenity Misspecification. Pricing the European Environment. S. Navrud. Oxford, Oxford University Press.

Naturvårdsverket, 1995. Riksinventering 1995. Report no. 4813, Swedish EPA,

Naturvårdsverket, 2003. Ingen övergödning. Underlag för fördjupad utvärdering av miljömålsarbetet. (No eutrophication). Report nr 5319, Stockholm.

Naturvårdsverket, 2005 a. Förändring under ytan (Changes under the surface). Monitor 19, Naturvårdsverket, Stockholm.

Naturvårdsverket, 2005 b. Kvalitetskriterier för ekonomiska miljövärderingsstudier (Qulity criteria for economic valuation studies. In Swedish). Naturvårdsverket, Stockholm.

Naturvårdsverket, 2006. Eutrophication of Swedish seas. Report 5509. Stockholm.

Navrud, S., 2000. Strengths, weaknesses and policy utility of valuation techniques and benefit transfer methods. Proceedings from OECD-USDA workshop: "The Value of Rural Amenities: Dealing with Public Goods, Non-market Goods and Externalities", Washington D.C., June 5-6, 2000., Nerhagen, L., B. Forsberg, C. Johanssson and B. Lövenheim, 2005. Luftföroreningars externa kostnader. Förslag på beräkningsmetod för trafiken utifrån granskning av ExternE-beräkningar för Stockholm och Sverige. VTI Rapport 517,

Nerhagen, L. and H. Johansson, 2003. Variations in the external cost of transport air pollution - the case of Sweden. VTI Notat 36A-2003, Väg- och Transportforskningsinstitutet,

NIER, N. l. o. E. R., 1998. Swedish environmental accounts for sulphur and nitrogen. (In Swedish). Environmental accounts 1998:7, Stockholm.

NIVA, 2003. Common Procedure for Identification of the Eutrophication Status of Maritime Area of the Oslo and Paris Conventions. NIVA Report 4654-2003 (SFT TA-1927-2003), Norsk Institutt for Vannforskning, Oslo.

Ready, R., S. Navrud, R. Ready, B. Day, R. Dubourg, F. Machado, S. Mourato, F. Spanninks and M. X. Vazquez Rodriquez, 2004. Benefit transfer in Europe. How reliable are transfers between countries? Environmental and Resource Economics 29(1): 67-82.

Sandström, M., 1996. Recreational benefits from improved water quality: A random utility model of Swedish seaside recreation. Working paper series in economics and finance No 121, Stockholm.

SIKA, 2004. The external costs of transport. SIKA report 2004:4, Stockholm.

SIKA, 2005. Arbete med att utveckla värderingar för trafikens avgasutsläpp (The work with developing valuations for air emissions from traffic).In Swedish. SIKA Report 2005:9.

Silvander, U., 1991. Betalningsvillighetsstudier för sportfiske och grundvatten i Sverige/ The willingness to pay for angling and ground water in Sweden. Institutionen för ekonomi, Sveriges Lantbruksuniversitet, Uppsala.

SMHI, 1996. Samordnad kustvattenkontroll för Kalmar Län. Årsrapport 1996. (Coordinated control of coastal water areas in the county of Kalmar. Annual report 1996. In Swedish). SMHI, Fiskeriverket och Statens Geologiska undersökning, Kalmar.

Smith, K. V., G. Van Houtven and S. Pattanayak, 2000. Improving the Practice of Benefit Transfer: A Preference Calibration Approach. Paper prepared for the US Environmental Protection Agency, Washington, D.C.

Smith, V. K., G. Van Houtven and S. K. Pattanayak, 2002. Benefit Transfer via Preference Calibration: "Prudential Algebra" for Policy. Land Economics 78(1): 132-152.

Soutokorva, Å. 2005. The Value of Improved Water Quality. A Random Utility Model of Recreation in the Stockholm Archipelago.

Steen B. 1999. A systematic approach to Environmental Priority Strategies in product development (EPS), Version 2000—models and data of the default method. CPM report. Göteborg, Sweden: Chalmers University;

Söderqvist, T., 1996. Contingent Valuation of a Less Eutrophicated Baltic Sea. Beijer Discussion papers 88, Beijer Int. Institute of Ecological Economics,

Söderqvist, T. and H. Scharin, 2000. The regional willingness to pay for a reduced eutrophication in the Stockholm archipelago. Beijer Discussion paper No.128, Beijer Int. Institute of Ecological Economics,

Vermoote, S. and L. De Nocker, 2003. Valuation of Environmental Impacts of Acidification and Eutrophication Based on the Standard Price Method. VITO NV - Integral Environmental Studies,

Willig, R., 1978. Incremental Consumer's Surplus and Hedonic Price Adjustment. Journal of Economic Theory 17(2): 227-253.