Benefit measures for noise abatement: Calculations for road and rail traffic noise*

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Noise is a significant social problem with more than 20% of the EU's population being exposed to higher noise levels than are considered acceptable. The transport sector is a major contributor to society's noise problem. Road traffic constitutes the largest individual noise source in the transport sector, but other transport modes such as aircraft and trains are also substantial sources. Efforts to reduce noise levels come at a cost and policies and projects to reduce noise levels need to be evaluated to secure an efficient resource allocation. Benefit-cost analysis provides a powerful tool for this, but it requires that benefits and costs are measured in monetary terms. This study focus on estimating monetary values which reflect the social value of noise abatement. It is well established that the noise-annoyance varies in accordance with the mode of transport at the same level and there is, therefore, a need to ascertain monetary values for the respective transport modes. This report focuses on road and rail noise. Based on the need to revise the current official Swedish benefit measures for noise abatement we show how to combine monetary values for individuals' preferences with values reflecting health costs to reflect the social value of noise abatement, but also critically discuss whether preference values should be augmented.

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1 Introduction

Noise is a considerable social problem. For example, more than 20% of the EU's population are exposed to higher levels of noise than are deemed acceptable (European Commission, 1996). The transport sector is a principal contributor to society's noise problem, and the combination of increasing traffic volumes and urbanization means that the problem will increase if no measures are taken to curb it (Boverket, 2003; Nijland et al., 2003; Bluhm and Nordling, 2005; Kihlman, 2005). Road traffic is the largest single source of noise in the transport sector, but other modes such as aircraft and trains also contribute substantially to the noise emissions (Kalivoda et al., 2003; Lundström et al., 2003; SOU, 1993).

The negative effects of the noise may be reduced through legislation, e.g. requirement of less noisy technology, investments, such as noise barriers, or infrastructure use charges that punishes more noisy vehicles and lead to a reduction of the noise level. Noise-reducing measures often come at a cost, however, for instance longer traveling times or costs of physical measures such as noise barriers and façade insulation (Oertli, 2000; de Vos, 2003). This, and the fact that society also faces other needs, implies that there has to be some form of prioritization when it comes to resource allocation. Benefit-cost analysis (BCA) is a potential basis for decision making, but it requires, though, both benefits and costs to be measured in a common metric.

Monetary values act as this common metric and in this paper we estimate monetary values for road and rail traffic noise abatement. These two noise sources are of different character and it is well established that the annoyance from noise individuals report differs between the two modes of transport (Miedema and Oudshoorn, 2001). The origin of this paper is the recent revision of the official monetary benefit measures of noise abatement, from now on the "ASEK values"[1], carried out in Sweden (SIKA, 2008). The original values were revised to take into account general price increases and an increase in real growth. It was also decided that the original values should be adjusted upwards since it was assumed they they did not reflect the total social cost, but were missing the social cost of health effects from noise exposure (SIKA, 2008). This revision drew to the attention the potential need for a more comprehensive revision due to the fact that the values for all transport modes were based (after also other sources had been taken into consideration) on estimates from a study in which the effect on property prices from road noise was examined (Wilhelmsson, 1997; SIKA, 1999).

Thus, the aim of this study is to estimate monetary abatement values for road and rail traffic noise that can be used for policy purposes. The main objective is twofold: (i) to estimate monetary values that can be used for BCA in Sweden, and (ii) to examine the magnitude of the difference in estimates

[1] Freely translated from Swedish, the acronym ASEK refers to "Working group for benefit-cost analysis".

between road and railway noise. The former is mainly of policy relevance; both for BCA in which today often the social cost of noise is ignored (Day et al., 2007; Nelson, 2008) and for the pricing of the noise externality in transportation based on the social marginal cost principle (Andersson and Ögren, 2007, 2010). The latter objective has both a policy and research relevance. Benefit measures are usually not elicited for all noise sources and by examining to what extent they differ we test to what extent values for one noise source can be used as a measure for the other source, i.e. a test of the use of benefits transfer (Pearce et al., 2006). Moreover, two additional objectives is to address the question whether derived benefit measures need to be augmented by a health component and to examine the sensitivity of the estimates as a result of assumptions on discount rates. The issue about the health component refers to whether individuals are informed about negative health effects and their costs when they reveal their willingness to pay (WTP) to reduce their noise exposure. If not, this needs to be taken into account when calculating the benefit measures. To estimate the social value of noise abatement we combine measures reflecting individuals' preferences for reducing noise levels from a Swedish hedonic property price study (Andersson et al., 2010) with estimates for the social cost related to health effects from noise exposure.

This paper is structured as follows. The next section briefly discusses the characteristics of noise, depending on the transport mode, and how this affects the individual's annoyance level, assumptions about the social cost of noise exposure, various methods of estimating the social costs, and a review of evaluation studies. We thereafter in section 3 describe and show results of the evaluation of the social cost of noise in this study, i.e. the WTP estimates used and the evaluation of health effects. Section 4 then contains our models on how to combine the different cost components and our estimated benefit measures. In this section we show that the values for road and rail not only differ in levels but also that the relationship between the noise level and the marginal value differ between the two sources. The social value for railway noise reveals a stronger progressive relationship with the noise level compared with road noise. Moreover, we also show that the health cost component added to the WTP estimate will be small but not negligible and that also modest differences in the assumption of the discount rates will have a significant effect on the estimated values. The paper ends with a discussion and conclusions.

2 Noise - characteristics, social costs and evaluation

In the first part of this section we briefly describe how noise is measured and evidence on how individuals are annoyed by different types of traffic noise. We then describe the different social noise cost component, and end this section with a summary of previous findings in the literature.

2.1 Acoustics and annoyance

The A-weighted sound pressure level, measured over 24 hours and usually denoted $L_{\mathrm{AEq,24h}}$, is often used as an indicator of noise levels. It is an energy mean level and correlates well with the general annoyance due to noise at a given place. A more relevant measure for sleep disturbance is the maximum level combined with the number of occurrences (L_{AFmax}).

Different sources have different noise profiles over a 24-hour period. Rail traffic sometimes has a higher proportion of goods trains running at night, and it is therefore not unusual for the equivalent level at night (23-07) to be higher than during the day (07-19). Road traffic reaches its peaks in the morning and afternoon, but there may sometimes be heavy traffic at night. Air traffic resembles road traffic in its distribution but with fewer and noisier events per 24 hours.

Noise sources also differ when seen in a shorter time perspective (variations over minutes). Both rail and air traffic typically have a high maximum level compared to the equivalent level; that is, the individual passages are separated, with silent periods in between. Road traffic tends to have a more even level: that is, smaller differences between maximum and equivalent levels. Note, though, that there are exceptions; for example, low-traffic roads with a high proportion of heavy vehicles may be more like railways in terms of the noise profile.

The EU joint noise indicator L_{DEN} is an attempt to balance the 24-hour effects of traffic. This is, in principle, a weighted equivalent level where passages in the evening and at night are counted as 5 dB and 10 dB noisier, respectively, than is actually the case. Thus, evening and night time noise are "punished" in the sense that they are given more weight in the model. This is also applicable to air traffic.

The evaluation of annoyance due to traffic noise by means of questionnaires is often carried out on a 5-grade scale, in accordance with ISO/TS 15666 (ISO, 2003). One can also predict the number of people on the various annoyance levels according to European Communities (2002), which is based on a metanalysis of many studies (Miedema and Oudshoorn, 2001). Note that the noise indicator used is L_{DEN}, which is why one must know the traffic distribution over a 24-hour period to be able to apply the prediction when only the equivalent level $L_{\mathrm{AEq,24h}}$ is known. Miedema and Oudshoorn (2001) clearly shows that the proportion of people who are annoyed at the same noise level (L_{DEN}) is largest for air traffic and lowest for rail traffic, with road traffic in between.

2.2 Social costs of noise exposure

Noise does not cause any direct environmental damage but incurs costs for society in the form of disturbances for the individual (sleep, conversation, recreation, etc.), worsened health and loss of production. The latter may be due to absence from work or reduced capacity to work, or that not getting a good night's sleep means that the individual is less productive than usual. The social costs of noise exposure may be divided into three groups (e.g. Andersson and Ögren, 2007):

1. *Resource costs* in the form of medical and health care. Includes costs financed by taxes and direct payments by the individual.

2. *Opportunity costs* in the form of loss of production. Includes "non-market services" carried out in the household and lost recreation time.

3. *Dis-utility* in the form of other negative influences resulting from noise exposure. Disturbances in different forms and increased concern about the after effects as a result of exposure are two examples.

Since the three components are not completely separable, an adding up of the three would mean an overestimation of the social cost. While the first two components, *Resource costs* and *Opportunity costs*, the sum of which is usually termed "Cost of illness" (COI) in health economics literature, may be estimated with existing market prices, there are no directly observable market prices for *Dis-utility*. *Dis-utility* is therefore estimated by means of the WTP approach, an approach that is usually divided into two main groups depending on what information is used. Preference estimates based on market data and hypothetical market situations are called "revealed preferences" (RP) and "stated preferences" (SP), where the notations show whether the actual or hypothetical choice is used. If individuals in the WTP studies were fully informed of the total cost of noise exposure and if they themselves bore the costs completely, the values from such studies would reflect the social costs in the form of COI as well. It has been suggested based on the assumption that that individuals are not fully informed of the negative effects of noise that WTP values should be augmented with a health cost component (e.g. Navrud, 2002; Bickel et al., 2006).[2] Whether this is indeed the case have not been proven, though. Hence, the evidence of the necessity of augmenting the WTP values with a health cost component is weak.

2.3 Overview of WTP studies

An overwhelming majority of the WTP studies to elicited preferences for noise abatement have employed the RP-approach using the hedonic price regression technique (Nelson, 2008). By studying how property prices are affected by noise exposure, at the same time as controlling for the effects of prices of other attributes, house owners preferences for noise abatement can be elicited. The "noise sensitivity depreciation index" (NSDI) has evolved as the standard measure of the WTP of this literature. This is a is a measure

[2] The financing of the COI related to noise is not directly linked to the exposed individuals, which could also motive adding a health cost component to the WTP estimate.

of the percentage change in the price as a result of a unit change in the noise level (Nelson, 1980).[3] The strength of HP studies lies in the fact that they are based on individuals' actual choices. A shortcoming is that it can be difficult, and sometimes impossible, to estimate the values of interest, for example the WTP for noise reduction at different times of the day (Carlsson et al., 2004). The methods that use the SP approach offer flexibility, but the hypothetical scenario is their weakness.[4] The SP method most often used to evaluate noise is the "contingent valuation method" (CVM) (Navrud, 2004), in which the respondents directly state their WTP for the good, here a reduction of the noise level. The strength of the CVM and other SP methods, as mentioned above, is that the analyst him/herself constructs the study and may therefore ask questions he/she wants answers to and control for how various factors, such as study design, may have affected the results.

We start this overview of evaluation studies on noise abatement by focusing on Swedish studies.[5] Table 1 contains Swedish WTP studies for traffic noise. As shown in the table, two studies use the hedonic approach and thereby market data (Hammar, 1974; Wilhelmsson, 1997), while four studies use a hypothetical approach, either CVM (Kihlman et al., 1993; Wibe, 1997; Bickel et al., 2006) or "stated choice modelling" (SCM) (Carlsson et al., 2004). The two hedonic studies employ the effects of traffic noise on property prices in Täby (Hammar, 1974) and Ängby (Wilhelmsson, 1997), both outside Stockholm, and estimated only the effect of road noise. Both studies found that the estimated percentage depreciation was progressively increasing with the noise level.

[Table 1 about here.]

The EU project HEATCO (Bickel et al., 2006), carried out in several European countries, was aimed at estimating the WTP to reduce noise from road and railway traffic.[6] Only individuals' WTP for a reduction of road traffic noise was estimated for Sweden. The results of the studies revealed a methodological problem. For example, the proportion that accepted the payment of a certain amount did not decrease monotonically with the level of the offer, and a large proportion stated they were not willing to pay although they admitted that they were disturbed, while others had a positive WTP even though they

[3] Let P and L denote property prices and noise level, respectively, then the NSDI is given by

$$\text{NSDI} = \left| \frac{\partial P}{\partial L} \frac{100}{P} \right|.$$

[4] Weaknesses of the SP-methods, such as insensitivity of the WTP to quantity of the good, hypothetical and strategic bias, anchorage effects and so on, are well known in the literature and may sometimes be a result of to poorly conducted studies (see, e.g., Bateman et al., 2002).

[5] A literature review was conducted of evaluation studies of noise with the WTP approach from 1990 up to October 2008 in both transport databases (ITRD, TRIS) and among economics journals (EconLit). A follow-up among publications in economics journals was conducted in July 2010.

[6] The countries taking part were Norway, Spain, Sweden, Germany, the UK and Hungary. The Hungarian study also estimated values for air traffic noise.

were not disturbed. As a consequence, the validity of the estimations is open to question. Bickel et al. (2006) chose not to use the new results, but to base the recommended calculation values on the results from Navrud (2002) in which an "EU-value" was calculated (Bickel, 2006). The values in Bickel et al. (2006) show a weak progressive relation, two segments with constant marginal costs, which is in contrast to today's official Swedish values.

Turning to the international literature, we find that most WTP studies have concentrated only on one noise source, usually air or road traffic noise effects on property values. In their overview of existing studies Bateman et al. (2001) and Navrud (2002) report NSDI values for road traffic in the interval 0.08-2.22% and 0.08-2.3%. Bateman et al. (2001) also find an average of ca 0.55 for the studies; that is, this mean value implies that a 1 dB increase leads to a reduction of property values by a little over half a percent. Nelson (2004) analyzed 20 hedonic studies on air traffic noise in the USA and Canada in a meta analysis with NSDI in the interval 0.28-1.49 and an average of 0.6. In a new review by Nelson (2008) interquartile means of 0.80% and 0.51% for air and road traffic noise were reported.

Two studies evaluate air, road and rail traffic noise (Dekkers and van der Straaten, 2008; Day et al., 2007). The principal interest in Dekkers and van der Straaten (2008) is the evaluation of air traffic noise, but the analysis is extended to include road and rail traffic noise as well. Various threshold levels are used for the different noise sources; it is assumed that the level at which noise is not annoying varies, depending on whether the noise comes from air or road or rail traffic. For aircraft traffic the limit is set to 45 dB while road traffic is considered annoying at a level over 55 dB and rail traffic over 60 dB. Dekkers and van der Straaten (2008) maintain that the choice of threshold level affects the results of the model and advocates caution when interpreting and using their results in which NSDI for air traffic is estimated at 0.77, while rail traffic has NSDI of 0.67 and road traffic 0.16. Day et al. (2007) is based on estimates from 8 sub-markets, which give NSDI from 0.18% to 0.55% for road traffic noise while the models for rail traffic noise indicate a higher NSDI around 0.67%. For air traffic noise the results are "erratic", which, according to Day et al. (2007) is explained by the very few properties subjected to air traffic noise in the data set. Day et al. (2007) also estimate the theoretically consistent welfare measures for non-marginal changes, the second step in the hedonic method (Rosen, 1974).[7]

Examples of other recent SP studies (besides HEATCO) on the evaluation of traffic noise are Garrod et al. (2002), Bjørner (2004), Arsenio et al. (2006), Galilea and Ortúzar (2005), Nunes and Travisi (2007), and Li et al. (2009).[8] To summarize, the results of these SP studier vary considerably and not always in the

[7]Studies that look explicitly at the evaluation of rail traffic noise are few in number; Brons et al. (2003) lists 4 hedonic and 2 SP studies, of which several are estimated according to distance from the railway rather than the noise level, or have other characteristics that make them less suitable for comparisons.

[8]Nunes and Travisi (2007) in addition to their own SP study also provide and overview of other SP studies.

direction one expects, given the wealth level of the countries. However, for brevity, since your empirical application is on Swedish data and based on a revealed preference approach, we omit a detailed account of these studies.

3 Estimation of the cost components

We start this section by describing the evaluation technique used to derive monetary values for noise annoyance and the empirical study that was conducted. In the second part we describe the estimation of the health cost component.

3.1 Evaluation of annoyance

3.1.1 Hedonic regression

Individuals' WTP for a reduction of their noise exposure is estimated using the hedonic regression method (Rosen, 1974).[9] The estimates are based on price data from the property market and, according to the hedonic method, the price (P) is assumed to be a function of the various attributes that constitute the property,

$$P = P(\mathbf{L}, \mathbf{A}), \tag{1}$$

where $\mathbf{L} = [L_1, L_2]$ and $\mathbf{A} = [a_1, \ldots, a_n]$ denote the noise attributes road (L_1) and railway (L_2) and a vector with other attributes. By studying how the price varies depending on the different levels of the attribute of interest, at the same time as controlling for the effects of other attributes, the individuals' marginal WTP can be estimated. Let $p_i,\ i \in \{1, 2\}$, denote the marginal WTP for a reduction of the noise level from source i, which is given by,

$$p_i = \frac{\partial P(\mathbf{L}, \mathbf{A})}{\partial L_i}. \tag{2}$$

Equation (2) gives the marginal WTP. To estimate the theoretically consistent welfare measure for non-marginal changes, the demand functions should be estimated. This is often referred to as the second stage of the hedonic method and was carried out in Day et al. (2007). An alternative, which assumes that the hedonic price function does not change as a result of the change in noise levels, is to base the individual's welfare change on the price function. The individual's welfare change is then given by the price change and even if it is not theoretically consistent it is a good measure of WTP for small changes in noise levels. This evaluation appraoch is based on an assumption of zero moving costs, making the value

[9] The hedonic method is already well documented and we therefore choose not to describe the method in detail, but to refer the reader to the original source Rosen (1974), alternatively Freeman (2003).

an upper limit for the individual's WTP.[10] In this study we only conduct the first step and, therefore, estimate the monetary values using the hedonic price function.

3.1.2 Econometric analysis

The estimated WTP used in this paper is based on Andersson et al. (2010). Since the data were also analyzed in Andersson et al. (2010) we only provide a terse description of the study and the results that are of interest to our analysis in this article.[11] To conduct their empirical analysis Andersson et al. used a pooled data set for Lerum, a municipality close to Gothenburg, which consisted of two sources; property noise levels from a study on the health effects of traffic noise conducted in Lerum in 2004 (Öhrström et al., 2005) and property prices and other attributes (besides the noise variables) from the *National Land Survey of Sweden.* Descriptive statistics for the different variables are listed and described in Table 5 in the appendix. Prices are reported in 2004 price levels and the explanatory variables used in the regression are *Living space, Quality index, Terraced, Linked* and *Detached* which describe property attributes, whereas the other variables, besides the two noise variables, describe geographical attributes of the properties. Of the latter variables one is a dummy for distance to the motorway, *E20 150m*, a proxy for other negative effects from living close to the road besides road noise, two are measures of the distance to nearest train station and motorway entrance, *Dist. station* and *Dist. entrance*, i.e. measures of positive effects of the railway and motorway, whereas the other geographical variables define different neighborhoods.

The two variables defining the noise indicators are our variables of main interest. These two variables reflect the equivalent noise levels ($L_{\mathrm{AEq,24h}}$) and we have access to the levels of both rail and road noise for each property. This provides us with unusual rich data on noise levels. The noise variables are in the regressions defined by the absolute noise level minus 45, with 0 for levels below 45 dB. The effect from noise on the property price should be zero when no negative effect is observed, and in our study we have chosen to use a lower limit of $L_{\mathrm{AEq,24h}} = 45$ dB. The limit is somewhat arbitrarily determined, but the percentage of persons reporting that they are annoyed by traffic noise is very low below this level (Miedema and Oudshoorn, 2001).

When choosing the functional form of the hedonic price function economic theory leaves us without much guidance (Rosen, 1974). Different forms were tested in Andersson et al. (2010) and based on their results, which revealed the necessity of allowing for a flexible price function, and expectations based on

[10]The current official Swedish monetary noise value is based on estimation with this approach (Wilhelmsson, 1997; SIKA, 2008). For a description of this approach and the property tax effect on the evaluation, see e.g. Freeman (2003).

[11]For a more comprehensive description of the analysis and the results we refer to Andersson et al. (2010).

evidence from the acoustical literature, our preferred hedonic price function has the following form,

$$P_i = \gamma_0 \prod_{j=1}^{2} f(L_{ij}) \prod_{h=1}^{H} a_{ih}^{\gamma_h} + \varepsilon_i, \tag{3}$$

where

$$f(L_{ij}) = 1 + \frac{1 - b_j - (1 - b_j)\, e^{k_j L_{ij}}}{e^{30k_j} - 1}. \tag{4}$$

The noise variables are given by $L_{ij} = L_{\mathrm{AEq,24h}} - 45$ (set to zero for negative values, i.e. if noise levels are below 45 dB) with subscript i and j denoting single properties and road (1) and rail (2), respectively. Other property attributes besides the noise variables are given by a_{ih}, and γ, b, and k are the parameters to be estimated. In Eq. (4) the parameter b corresponds to the maximum effect at the highest noise level 75 dB in the study area and k describes the concavity of the function. In the regression, the parameter k is restricted to be between 0 and 1 and is estimated as,

$$k_j = \frac{e^{c_j}}{1 + e^{c_j}}, \tag{5}$$

thus c is the parameter that is estimated in the regression. Note that b and k are estimated separately for road and rail noise. Hence, Eq. (3) makes it possible to assume not only different maximum effects from road and rail noise, but also different degrees of concavity for the two noise sources. Moreover, to get a more homogeneous sample only properties with a total noise level of at least 50 dB, i.e. the official Swedish threshold value for when noise is assumed to be disturbing, were included.[12]

The regression results and the NSDI estimates based on these results are shown in Table 2.[13] The hedonic regression is based on non-linear estimation and we first focus on the variables not defining noise levels. We find that the property attributes are all statistically significant and with the expected signs. Regarding the neighborhood dummies we find that some are significant compared to the reference group (*Floda 2*). Moreover, we find no evidence that the prices of properties situated within 150 meters from the motorway *E20* are significantly affected by the motorway, given that the noise level is controlled for. Further, *Dist. station* and *Dist. entrance* are both not statistically significant. The coefficients for

[12] The total equivalent noise level was calculated as

$$L_{\mathrm{tot}}(L_1, L_2) = 10 \log(10^{\frac{L_1}{10}} + 10^{\frac{L_2}{10}}),$$

where $L_j,\ j \in \{1, 2\}$, as before represents the equivalent noise level in dB from road (1) and rail (2) traffic noise, respectively.

[13] Andersson et al. (2010) also presented results based on a semi-logarithmic form and conducted tests for spatial dependence, i.e. a violation of the independence between observation. They found strong evidence of a spatial dependence and the reason why the regression in Table 2 has not been tested for spatial dependence is because it is estimated with non-linear estimation and methods for incorporating spatial dependence in non-linear regressions have not been developed. Andersson et al. (2010) also concluded that the spatial dependence would only have a negligible effect on the welfare estimates, hence the OLS and the spatial-lag model revealed similar estimates of marginal WTP. In addition they also examined the effect by using 55 dB as the threshold level, a level often used by authorities as a limit value below which no measures are taken to mitigate the noise (Nijland and Van Wee, 2005). They found that the results were sensitive to the threshold level chosen, a problem usually ignored in the literature where results based on only one level are reported.

the noise variables are our main interest. The relevant hypothesis testing regarding the noise variables is to test whether the b coefficient is equal to one, since $b_j = 1$ suggests that the price is not influenced by the noise level. We find that the coefficient for road noise is statistically significant different from one, but not for railway noise. For the k-parameter, calculated using the coefficient estimate of c_j (see Eq. (5)), a higher value implies a more concave function and a value close to zero implies an almost linear relationship between the noise level and the property price. The results, therefore, suggest a more concave relationship for rail than road noise.

[Table 2 about here.]

The last two columns of Table 2 show NSDI values for 4 different noise levels. The NSDI is given by

$$\mathrm{NSDI}(L_{ij}) = 100 \cdot \frac{f'(L_{ij})}{f(L_{ij})} = 100 \cdot \frac{k_j(1-b_j)e^{k_j L_{ij}}}{e^{30k_j} - b_j - (1-b_j)e^{k_j L_{ij}}} \tag{6}$$

which, since other attributes cancel, only depends on the noise level.[14] The NSDI increases with the noise level and the higher degree of concavity for rail noise leads to lower NSDI values from rail noise than road noise for low noise levels but higher values for very high noise levels. The effect of rail noise on the property prices is lower than the effect of road noise for all noise levels except the highest (70 dB).

3.2 Evaluation of health effects

As described above, if we assume that WTP studies do not capture the total social cost from noise exposure then the values from these studies need to be adjusted such that also the health effects of noise are included. In the first place, noise causes stress that may lead to high blood pressure and a higher risk of cardiovascular diseases over time. To our knowledge two methods have been suggested to include the health effects: (i) the total social cost of noise is calculated and related to estimates from WTP studies, and (ii) the impact pathway approach (IPA). The adjustment of the values in ASEK (SIKA, 2008) is related to the first method where the results of Danish studies are used (Ohm et al., 2003; Ohm and Jensen, 2003).

The health related costs, including loss of production and the health-risk exposure, according to exposure to road traffic noise for the whole of Denmark are estimated in Ohm et al. (2003) and Ohm and Jensen (2003). In these studies the total social cost, including the health cost, is estimated. The total cost is thereafter related to the total cost estimates based on HP studies. In the Danish studies the found relation implies that the hedonic values should be adjusted up by 42% to also include the health effects. Note, however, that this relation only applies to the total cost and it is not certain whether an adjustment

[14] $f'(L'_{ij}) = -\frac{k(1-b)\, e^{kL'_{ij}}}{e^{30k}-1}$

upwards by 42% in a certain area of calculation gives a correct result. Thus, this relationship applies only if the distribution of the number of exposed on different levels is similar in the calculation area to what it is in Denmark as a whole.[15]

The IPA method (Friedrich and Bickel, 2001; Metroeconomica, 2001; Navrud, 2004) uses the "bottom-up approach"; i.e., it starts with the emission source, goes on to estimations of the distribution and then the final effects of the emission. The final effects are thereafter given monetary values and the social cost can be established. The method builds on the fact that the emission, distribution and final effects can be measured with precision. However, monetary evaluation is necessary for the IPA as well, which together with the uncertainty in calculating the dose response relationship leads to great uncertainty in the estimations (Metroeconomica, 2001). The latter uncertainty comes both from determining the exposure in a research area with noise calculation methods and from the inherent uncertainty in the underlying epidemiological study. A dose response relationship is establishes by relating the exposure to a certain noise level to an end effect (such as hypertension or myocardial infarction) via a relative risk or odds ratio. The confidence intervals are usually rather wide and often include the "no influence" outcome, and it is vital to control for other important factors such as smoking and diet habits. Finally there is an uncertainty in the estimations of the costs of a certain end effect for a part of the population. HEATCO (Bickel et al., 2006), e.g., uses IPA to include the effect of health costs. Unlike ASEK, which assumes a positive health cost from 50 dB, HEATCO assumes that a health cost starts at L_{DEN} =70 dB (that is, the health cost is zero at lower levels), which increases linearly thereafter (Bickel, 2006).

The two approaches have their weaknesses but we argue that the IPA is the preferred approach. More evidence is needed before the approach where estimated total costs are related to WTP estimates can be used, and the IPA has been suggested to be used on the EU level as described above. Based on the IPA the following expression starts from evaluations of the health effects of road traffic noise in a recent Swedish study (Lindberg, 2007) which estimated a linear cost function between 70 and 80 dB expressed as L_{DEN}. For road traffic with a normal 24-hour distribution, the difference between L_{DEN} and $L_{\mathrm{AEq,24h}}$ is 3 dB according to Jonasson (2005). Thus, there are no health effects under the equivalent level of 67 dB, but this can be an expression of the fact that the health studies on which it builds do not comprise enough people to distinguish the small effects at lower noise levels. It seems hardly likely that the health cost would be approximately SEK 1,000 per year and person at 67 dB but SEK 0 at

[15] Angelov (2008) evaluates the health cost of road traffic noise with similar methods using data for the Swedish relationship from Kjellström et al. (2008). The risk of high blood pressure and coronary artery disease is calculated for the whole population. The results of the study should be "regarded as a calculation exercise" (Angelov, 2008, p. 2, freely translated from Swedish) but imply that previous ASEK values should be adjusted upwards by 60% instead of the 42% ASEK-choice based on the Danish studies.

66 dB.[16] We therefore suggest that the health cost should be extrapolated downwards with the same linear trend as over 67 dB. The health cost would then be zero just under 53 dB, and the marginal cost can be calculated as

$$H(L_i) = \begin{cases} 74.2 & \text{if} \quad L_i \geq (L_h - 45) \\ 0 & \text{if} \quad L_i < (L_h - 45) \end{cases}, \tag{7}$$

which should be added to the WTP according to Eq. (14) below. The limit L_h is 52.74 dB, the term -45 comes from the fact that the noise attribute L_i is defined as the 24-hour equivalent minus 45 dB. The curve corresponds to SEK 72.4 per person and year for every dB of increased noise level. A lack of direct studies of the health effects of noise from rail traffic means that the health costs are set at the same level as for road traffic, which is probably an overestimation. The same approach is used in HEATCO.

4 Estimation of the social cost of noise

In the following section we first describe how the estimates from the HP study are converted to values that can be implemented in policy evaluations and how the WTP estimates of annoyance and health effects can be combined to reflect the total social cost (under the assumption that the WTP does not already reflect to total social cost). We then examine the sensitivity of the annual benefit estimates to the discount rate chosen. Finally we relate our results to other findings of the literature that we consider are of main relevance to our study, i.e. current official Swedish monetary values (SIKA, 2008), a recent European multinational study (Bickel et al., 2006), and a recent hedonic pricing study estimating WTP for both road and rail traffic (Day et al., 2007).

4.1 Noise evaluation model

The estimated hedonic price functions in Table 2 provide the price change; that is, a present value as a result of a noise change. Moreover, the revealed WTP for most households, i.e. those with more than one household member, does not define individual WTP. The estimated price change should therefore be: (i) recalculated into an annuity and, assuming eternal life of the property, the estimated value is multiplied by the real discount factor, r, (Sydsæter et al., 2000) and (ii) divided by the number of household members that the WTP refers to, n. Moreover, Eq. (2) ignores the effect of a property tax. Not taking into account the property-tax effect on the property price means that the welfare effect is underestimated (Niskanen and Hanke, 1977). Since the property tax (t) is not based on the actual value but on the taxable value, only a part of the actual price is taxed. Let $\lambda = [0, 1]$ denote the proportion of

[16]

the price that is taxed, and the annualized marginal individual WTP is given by,

$$V_i = \frac{(r + t\lambda)p_i}{n}. \tag{8}$$

The marginal effects on property prices using the hedonic price function in Eq. (3) is given by:

$$p_i = \gamma_0 \, f'(L_i) f(L_j) \prod_{h=1}^{H} a_h^{\gamma_h}, \tag{9}$$

where

$$f(L_j) = 1 + \frac{1 - b_j - (1 - b_j)\, e^{k_j L_j}}{e^{30k_j} - 1}, \tag{10}$$

$$f'(L_i) = -\frac{k_i(1 - b_i)\, e^{k_i L_i}}{e^{30k_i} - 1}, \tag{11}$$

and where prim denotes the first derivative. For each noise sources the marginal WTP is evaluated based on the mean value of the other variables. Equation (9) may therefore be written as:

$$p_i = \beta_j \, f'(L_i), \tag{12}$$

where β_j is a constant. Index j denotes that this constant is dependent on the level of the other noise sources; that is, the constant varies between the noise variables.

As the estimates are to be used on a national level, the income differences between Lerum and the rest of the country should be taken into account. Corrections are based on the difference in income and empirically estimated income elasticities for the WTP for a noise reduction (θ). Let Y_S and Y_L denote mean income for Sweden and Lerum, respectively, and the calculation model for the WTP then becomes:

$$B(L_i) = -V_i \left(\frac{Y_S}{Y_L}\right)^{\theta} = -\frac{(r + t\lambda)\beta_j \, f'(L_i)}{n} \left(\frac{Y_S}{Y_L}\right)^{\theta}, \tag{13}$$

where the equation is multiplied by -1 to give a positive value.

As the threshold value for when noise is regarded as disturbing is set at 50 dB in Sweden (SIKA, 2008), our estimations are based on this value. The estimation of the annual social cost per person in the case of a change in level from l' to l'' can then be estimated by:

$$S(L_i) = \int_{l'}^{l''} [B(L_i) + H(L_i)] \, \mathrm{d}L, \qquad l' \geq 50, \tag{14}$$

where $B(L_i)$ and $H(L_i)$ are given by Eqs. (13) and (7). Hence, Eq. (14) assumes that the estimated WTP do not reflect the total social cost, since it includes $H(L_i)$ to estimate $S(L_i)$. Therefore, by simply dropping $H(L_i)$ from the equation we can estimate the total social cost under the assumption that WTP do indeed reflect the total social cost.

4.2 Abatement values for transport noise

As mentioned above, Eq. (12) should be turned into an annuity and the property tax and number of occupants of the property should be taken into consideration. Since estimations are sensitive to the choice of the discount factor, we choose to report calculations on three levels. As a discount factor we choose the level 4% as suggested by ASEK (SIKA, 2008) for BCA in Sweden and for the sensitivity of our estimations we also show the results for 2% and 6%. The property tax in 2004 was 1.0%, which is applied to 50% of the value, and the number of occupants for our sample was, on average, 2.8 (Öhrström et al., 2005).[17]

In order to transfer the results from Lerum to the rest of Sweden, differences in income between regions ought to be considered. Empirical estimations of income elasticity vary between 0.5-1.6 (Palmquist, 1992; Bjørner, 2004; Wardman and Bristow, 2004; Arsenio et al., 2006; Nellthorp et al., 2007). Since most of the estimations lie in the lower interval and nearer 1, we choose, like Nellthorp et al. (2007), to set $\theta = 1$. This means that the values from Lerum should be multiplied by the actual quotient between the average incomes for Sweden and Lerum, which, for the age group 20 and over, was 0.875 during the data period (www.ssd.scb.se, 2008-11-19).

As explained, Eq. (14) should be used for "smaller changes", i.e. for individuals' preferences for marginal changes in noise level (not elimination). Table 3 contains some examples where the change is 1 dB. The choice of levels is based on the presentation in Day et al. (2007) and the estimates have been calculated without the health cost component to facilitate a comparison with other WTP studies . The constants reported in Table 3 are from Eqs. (11) and (13). This table reports the results of the sensitivity analysis for choice of discount factor, and as shown by the table, the values are sensitive to the choice of the discount factor.

[Table 3 about here.]

Table 4 provides the comparison between the benefit measures of this paper and the three studies of interest described above. The benefit measures of this study (REBUS) which are estimated with Eq. (14) are shown with and without the health premium. As mentioned above, it was assumed that the earlier ASEK values, based on hedonic evaluation, did not reflect the whole social cost (SIKA, 2008, p. 119-120), and therefore, based on Danish findings (Ohm et al., 2003; Ohm and Jensen, 2003) the WTP estimates were adjusted upwards by 42%. Since we suggest a constant marginal cost per dB for the health premium the percentage adjustment varies over the levels with the highest adjustment for lower levels, and by

[17] The property tax is based on the property's taxable value. The point of departure is that this value should constitute 75% of the market value (SFS, 2001). Data material from Lerum showed, though, a taxable value corresponding to 50% of the market value.

mode. Whereas the highest adjustment for road noise is 20% it is considerably higher for rail noise, 300%. Due to the convexity of cost function for rail noise it rapidly drops to 6% and 2% for the two highest levels in Table 4, though. Hence, the constant percentage adjustment is not supported by our findings, and the health premium is most significant at lower levels, especially for rail noise.

[Table 4 about here.]

Our estimates (REBUS) are generally higher than the estimates from HEATCO and higher for road-traffic noise compared to Day et al. (2007). Note that Day et al. (2007) finds a higher WTP to reduce railway noise than road noise. A suggested explanation for their results is too few observations for railway noise (Nellthorp et al., 2007, p. 334). A direct comparison between REBUS and ASEK is only relevant for road noise since there is no corresponding ASEK value for railway noise. Therefore, since the ASEK values include the health component we compare these values to the REBUS values also including the health component. These REBUS values indicate that the social cost of noise is underestimated at low levels with the current values, and substantially overestimated at higher levels. The strongly progressive relation in ASEK is not found in either HEATCO or in Day et al. (2007).[18] The relation between cost and noise level for railways in REBUS is similar to that in ASEK.

5 Discussion and conclusions

We have in this estimated the social cost of road and rail traffic noise and described how WTP estimates from hedonic pricing studies can be combined with cost estimates for health effects in order for the estimates to reflect to total social cost. Based on estimates where the two cost components have been combined we find that the largest part of the social cost from noise exposure is the annoyance component, which is reflected in the individuals' WTP. The WTP estimates in this study correspond with the annoyance relationship found in the acoustics literature; that is, the WTP is generally higher for reducing road than rail traffic noise. This is important from a validity perspective and our results show that there is a need to revise the Swedish benefit measure of transport noise abatement. Today's values for all transport modes are based on WTP estimates for road noise. By using an unusually rich data set, in which we are able to estimate individuals' WTP separately for road and railway noise, we show that WTP to reduce road and rail traffic noise not only differ in absolute but also marginal levels.

Our finding also show the importance of the chosen social discount rate. Even modest differences in the discount rates result in significant differences in the welfare estimates. Hence, the chosen discount

[18] A reason for this difference between ASEK and our results may be Wilhelmsson's function-form combined with the fact that the study did not control for other negative effects of proximity to roads; that is, the marginal relation between costs and noise is probably overestimated at high noise levels using the current calculation values, a problem also noticed in Wilhelmsson (2000, p. 808).

rate by analysts or policy makers needs to be well motivated, to avoid values being tampered with. Regarding the second cost component, the unintended health effect, even if it is small relative to the WTP for most noise levels it is not negligible. For rail traffic and noise levels within the lower end of the interval it is either the dominant component or a relatively large share of it since here the welfare estimate of annoyance is low. Since most individuals live at the lower noise levels this has implications for a welfare analysis, such as BCA. For Swedish policy purposes we suggest that health costs should be evaluated by means of the impact pathway approach (IPA) since it is sanctioned in the EU, but more importantly, is superior to the alternative approach of relating the total estimated social cost to WTP estimates (as suggested by ASEK). More importantly, though, we believe that more research is necessary not only to study the relationship between annoyance and health effects, but also what effects are unintended and thereby not part of the individual's WTP. The approach of adjusting the WTP estimates assumes that house owners are indeed unaware of the negative health effects and do not bear the full cost of noise related health effects. This is, as outlined above, in line with the general view among analysts and policy makers. However, as also pointed out, the empirical evidence is limited. Therefore, if evidence instead suggests that house owners are well informed, and that the COI related to these health effects is negligible, the estimates from the hedonic price regression analysis should be left unadjusted.

Estimated WTP in this study is based on actual decisions by house buyers and the estimates are adjusted based on income differences between the study area and the national level. The latter refers to the use of benefits transfer (BT) and it has been argued that evaluation in respect of BT should be based on annoyance, not dB, and done with SP-studies (Navrud, 2004, p. 30). The annoyance measure rather than dB is preferred simply because it can be evaluated with SP methods. The advantage of using an SP method is that it possible to establish the effect of the study design and individual characteristics on the results. However, the difficulty of estimating individuals' preferences in SP studies is well established (Carson et al., 2001; Bateman et al., 2002) and we, therefore, do not agree that there is support that SP are superior to RP methods. Revealed preference methods are based on actual decision, not hypothetical ones, and if, for instance, data is available to conduct the second step of the HP method (Day et al., 2007; Nellthorp et al., 2007) it is possible to adjust the welfare measures based on results on individual characteristics from a RP study.

Today's traffic noise policies are often influenced by guidelines for acceptable noise levels rather than welfare efficiency.[19] To gain acceptance for the use of BCA among policy makers it is therefore important not only to provide monetary welfare measures but also to inform about strengths and weaknesses, policy

[19]This is, for instance, the case in Sweden (e.g., SIKA, 2005) even though the Swedish legislation requires that transport policy should be formulated in terms of welfare efficiency (Prop. 2008/09:35, 2001).

implications of using different values, and potential use. For instance, the belief that WTP estimates should be augmented with the health cost is widely held among many policy makers. We have shown how this can be done, but we have also highlighted that the evidence for doing so is weak. The policy implication of using today's official values instead of the estimates of this study is the risk that social benefits from a road-noise-reducing measures will be underestimated, due to the strong progressive relationship between costs and noise levels and the fact that most people are subjected to low noise levels. Moreover, the choice of the recommended welfare measures is also important for the potential use of infrastructure user charges based on the marginal cost principle, since it has been shown these charges are determined largely by the individuals subjected to low noise levels (Andersson and Ögren, 2007, 2010). Evidence found in this and other studies need to be communicated by analysts to policy makers to secure a more efficient resource allocation. This study aims of doing this and has with a critical discussion of its results contributed to the estimation of the total social cost related to road and rail noise. There is, though, as has been highlighted need and room for further research.

Appendix: Descriptive statistics

[Table 5 about here.]

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Table 1: Swedish noise WTP studies

Study	Noise type	
Hedonic regression		NSDI (%)
Hammar (1974)[a]	Road and Railway	0.8-1.7
Wilhelmsson (1997)[b]	Road	0.5-5.0
Stated preferences		Description of monetary values[c]
Kihlman et al. (1993)	Transport	112-250 SEK/month/household for a quiet traffic environment.
Wibe (1997)	Living	212 SEK/month/household for a quiet living environment.
Carlsson et al. (2004)	Aircraft	5-25 SEK/takeoff depending on more or less takeoffs and when the takeoff occurs, i.e. day of week and time during the day. The results indicate a higher WTP for reducing the noise level in the evenings and mornings.
Bickel et al. (2006)	Road	122-211 SEK/year/person per dB(A).[d]
	Railway	122-211 SEK/year/person per dB(A) with a 5 dB(A) "rail bonus".[d]
	Aircraft	188-277 SEK/year/person per dB(A).[d]

a: Results from Hammar (1974) based on presentation in Hansson (1995).
b: Also published as Wilhelmsson (2000) in which the interval for NSDI is 0.3-3.0%.
c: Values shown in the price level of each study.
d: Values adjusted from EUR to SEK (EUR 1 = SEK 9.16, www.riksbank.se, 2008-11-28) and from factor costs, where the factor 1.21 has been used (SIKA, 2008). As from 70 dB(A) the values also include a health cost component which lead to a higher EUR per dB(A) in the interval 70-71 dB(A) than the values reported here (Bickel et al., 2006, Table 6.9).

Table 2: Regression and noise sensitivity depreciation index (NSDI) results

Regression results			NSDI		
Variable	**Coefficient**	**(Std. Err.)**	**Level**	**Road**	**Rail**
Living space	0.485***	(0.049)	55 dB	1.35	0.08
Quality index	0.310***	(0.062)	60 dB	1.70	0.28
Terraced	-0.315***	(0.025)	65 dB	2.19	1.03
Linked	-0.174***	(0.026)	70 dB	2.90	4.09
Aspen1	0.274***	(0.058)			
Aspen2	0.218***	(0.055)			
Aspedalen1	0.219***	(0.051)			
Aspedalen2	0.312***	(0.029)			
Lerum1	0.187***	(0.038)			
Lerum2	0.153***	(0.027)			
Country side	0.063	(0.044)			
Stenkullen1	0.079	(0.100)			
Stenkullen2	-0.012	(0.079)			
Floda1	0.080	(0.057)			
E20 150m	-0.012	(0.034)			
Dist. station	-0.004	(0.029)			
Dist. entrance	0.039	(0.029)			
b_1	0.560***	(0.117)			
c_1	-3.448**	(1.396)			
b_2	0.506	(0.712)			
c_2	-1.078	(2.094)			
Constant	62.848***	(14.536)			
k_1	0.031	(0.417)			
k_2	0.254	(0.397)			
N	1034				
R^2	0.949				

Robust standard errors in brackets.
Significance levels: * 10%, ** 5%, *** 1%
Subscript $j = \{1, 2\}$ denotes road (1) and rail (2).
$k_j = e^{c_j}/(1 + e^{c_j})$
NSDI $= |(\partial P/\partial L)(100/P)|$

Table 3: Constants and sensitivity analysis: SEK/person/year in 2004 price level[a]

	Constants[b]			**Change**		**Sensitivity analysis**		
	β	k	b	High	Low	$r = 2\%$	$r = 4\%$	$r = 6\%$
Road	1 938 866	0.031	0.560	56	55	202	363	525
				66	65	275	495	715
				75	74	364	654	945
Railway	2 097 665	0.254	0.506	56	55	13	24	35
				66	65	171	308	444
				75	74	1681	3027	4372

a: Calculated with Eq. 14 with $H(L_i) = 0$, i.e. without the health component.
b: k and b from Table 2. β-value for road is the value that is used in Eq. (13) when calculating for railway. and vice versa

Table 4: Welfare estimates: SEK/person/year in 2004 price level

Change		REBUS w/o health		REBUS w/ health		ASEK 4[a]	HEATCO[b]		Day et al. (2007)[c]	
High	Low	Road	Railway	Road	Railway	Road	Road	Railway	Road	Railway
56	55	363	24	437	98	280	125	0	218	578
61	60	424	86	498	161	336	125	125	300	656
66	65	495	308	569	382	616	125	125	383	733
71	70	578	1096	652	1170	2371	817	817	465	811
75	74	654	3027	729	3101	3621	204	204	548	888

Adjusted to 2004 price level using the CPI from SIKA (2008).
a: Adjusted based on real GDP (SIKA, 2008).
b: Adjusted from factor costs. factor= 1.21 (SIKA, 2008). EUR 1 = SEK 9.16 (www.riksbank.se, 2008-11-28)
c: Adjusted from household to person based on 2.36 household members (Nellthorp et al., 2007) and purchasing power parity (stats.oecd.org, 2007-09-02). The reported value for the highest level is for the interval $75 - 76$ dB. Since REBUS is based on $L_{\text{DEN}} \leq 75$ dB, we have chosen $74 - 75$ dB as the highest interval.

Table 5: Descriptive statistics

Variable	**Description**	**Mean value**
Price	Property price in thousand SEK and 2004 price level	1917.913 (675.549)
Living space	Living space in square meters	130.144 (47.606)
Quality index	Index of indoor-quality	29.016 (5.517)
Terraced	Dummy equals one if terraced house	0.063
Linked	- " - if house linked by a garage	0.093
Detached	- " - if detached house	0.844
Aspen 1	- " - if <1 km from nearest stn Aspen	0.026
Aspen 2	- " - if 1-2 km from nearest stn Aspen	0.043
Aspedalen 1	- " - if <1 km from nearest stn Aspedalen	0.049
Aspedalen 2	- " - if 1-2 km from nearest stn Aspedalen	0.088
Lerum 1	- " - if <1 km from nearest stn Lerum	0.063
Lerum 2	- " - if 1-2 km from nearest stn Lerum	0.252
Countryside	- " - if >2 km from nearest station	0.112
Stenkullen 1	- " - if <1 km from nearest stn Stenkullen	0.019
Stenkullen 2	- " - if 1-2 km from nearest stn Stenkullen	0.067
Floda 1	- " - if <1 km from nearest stn Floda	0.035
Floda 2	- " - if 1-2 km from nearest stn Floda	0.246
E20 150m	- " - if within 150 m from motorway	0.136
Dist. station	Distance to nearest railway station in km	1.672 (1.320)
Dist. entrance	Distance to nearest motorway entrance in km	1.960 (1.005)
Road noise	Road noise in dB exceeding 45 dB	7.566 (4.17)
Rail noise	Rail noise in dB exceeding 45 dB	3.005 (4.888)

$N = 1034$

Standard deviations in brackets below mean values. For dummies, std.dev.$(x) = \sqrt{\bar{x}(1-\bar{x})}$.

EUR 1 = SEK 9.13, www.riksbank.se, 9/16/2008