Charging the polluters: A pricing model for road and railway noise

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Abstract

This study outlines a method to estimate the short run marginal cost (SRMC) for road and railway noise. It is based on standardized calculation methods for total noise levels and monetary cost estimates from well established evaluation methods. Here official calculation methods and monetary values are used for Sweden, but the estimation method for the SRMC outlined can be directly applied using other standardized noise calculation methods and monetary values. This implies that the current knowledge regarding the calculation of total noise levels and the evaluation of the social cost of noise can be extended to estimate the marginal effect as well. This is an important finding since it enables policy makers to price noise externalities in an appropriate way. Several sensitivity tests run for the SRMC show that: (i) increasing the total traffic on the infrastructure has only a minor influence, (ii) estimates are quite sensitive to the number of exposed individuals, and (iii) to the monetary values used. Hence, benefits transfer, i.e. using monetary values elicited based on road noise for railway noise, should be done with caution or not at all. Results also show that the use of quiet technology can have a significant effect on the SRMC. The fact that this model is able to differentiate not only modes of transport, but also vehicles and even technologies is an important finding. It is essential that the noise charges give the operators the right incentives to choose their optimal allocation.

Keywords: Externalities; Marginal cost, Noise; Railway; Road

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1 Introduction

The transportation sector provides many benefits to the society. For instance it is crucial both for business to facilitate geographical specialization and physical access to markets, and for individuals to access labor markets, to consume many of the goods and services produced, and to socialize. Hence, the transportation sector is important for public welfare, but many of its activities are accompanied by negative social effects, such as the risk of fatalities and injuries, air pollution, loss of valuable time (i.e. the problem of congestion), and noise nuisance. These negative impacts are not marginal from a social perspective, but are significant and can also be the main source of these bads. For example, the transport sector is a major source of air pollution and a dominating one in urban areas, accounting for more than 20% of greenhouse gas emission in the USA and the European Union (EU) (OECD, 2006). In 2000 the total external costs (excluding congestion costs) of transport for the 15 European Union (EU) member at the time and Switzerland and Norway was estimated to be 650 billion euros, which corresponded to 7.3% of their total GDP (OECD, 2006).

From a policy perspective these negative effects would not be problematic in the absence of market failures. With no market failures individuals would allocate their resources such that their individual welfare, and thus the total social welfare was maximized. The transportation sector faces several market failures, though. For instance, regarding the negative effects mentioned above, individuals may not be well informed about their actual exposure level of, for instance air pollution, or what the potential effect of such a pollution level is on their own health or the environment. Besides other market failures such as monopoly powers and political intervention, the existence of externalities is probably the major market failure in the transportation sector.

This study focuses on the problem of noise nuisance from road and railway traffic. Noise related to transportation is a considerable social problem with estimates suggesting that more than 20% of the population within the EU are exposed to higher noise levels than are deemed acceptable (European Commission, 1996). The largest single source is road traffic, but air and rail traffic also contribute substantially (Kalivoda et al., 2003; Lundström et al., 2003; SOU, 1993). The externality of noise exposure is an obvious market failure. Holding other aspects of the decision alternatives constant, train passengers or car drivers will base their decision on when and where to travel on the noise level they are exposed to inside the vehicle. They are likely to ignore the effect of their decision on others' exposure to the noise they emit.

The EU has decided that infrastructure-use charges should be based on the short run marginal cost (SRMC) principle (European Commission, 1998) and that different externalities should be internalized in these charges. This has the potential of mitigating the negative effects of transportation use and making the resource allocation of the sector more efficient, which is an important policy for all externalities within the sector, but particularly so for noise pollution. Since noise is less "visible" than congestion, air pollution or accident risk, and since there is a relatively long time period between exposure and negative health effects, the incentives for policy makers to address noise problems may be lower than for other problems. Moreover, users of transportation networks, such as train companies or individual car drivers, have no incentives to try to influence the policy makers to address the noise externality problem, since it will increase their costs, but may not provide any benefits. Thus, there is a risk that noise is ignored. The SRMC principle forces policy makers to deal with the noise problem as well, which is particularly important since noise annoyance is likely to increase over the years due the to combined effect of urbanization and increased traffic (Nijland et al., 2003).

The aim of this study is to design pricing models for road and railway noise based on the marginal cost principle that not only deal with the externality problem, but are also sophisticated enough to give operators of the infrastructure incentives to contribute to a more efficient resource allocation. Regarding the latter, we develop models which estimate the marginal social cost on the vehicle level. This article draws on previous findings in Andersson and Ögren (2007, 2010) on the development of the pricing model but improves the analysis in several aspects. One important aspect is that the differences in how to estimate the marginal acoustical effects of road and rail traffic is thoroughly developed and described. A major contribution of this paper is the sensitivity analysis in which several important parameters of the models are varied, for instance total traffic volume. This analysis provides important insight into when the estimates of the SRMC need to be adjusted based on changes of the levels of the parameters Moreover, previous studies were based on benefit measures of noise abatement, which were missing for rail noise and have been questioned for road noise (Andersson et al., 2010a). As a consequence, benefit measures based on road noise were used, with or without adjustments, for rail traffic as well. It is, however, well known that individuals' annoyance with road and rail traffic differs and so does their WTP to reduce their exposure to the two sources (Miedema and Oudshoorn, 2001; Day et al., 2007; Andersson et al., 2010b). Benefit measures are not often available for all noise sources and we, therefore, also conduct a sensitivity analysis to examine the effect of using road traffic noise benefit measures for rail traffic. Hence, the objectives of the article are threefold: (i) to design noise pricing models based on the marginal cost principle, (ii) to outline how to calculate the marginal acoustical effects of road and rail traffic noise, and (iii) conduct several sensitivity tests. In order to conduct the empirical analysis we use Swedish data.

The article is organized as follows. In section 2 we briefly describe the motivation of the marginal cost principle and derive our theoretical model. We thereafter explain and present the most common noise indicators and describe how the marginal acoustical effects of road and rail traffic are estimated. Sections 4 and 5 contain our empirical results and sensitivity analyses. We then discuss our results and relate them to other findings in the literature, and offer some concluding remarks in section 6.

2 Internalizing the external cost

The EU decision to base infrastructure user charges on the social marginal cost principle (European Commission, 1998) is based on Pigou (1920)'s work on externalities. Since many of the activities within the transportation sector produce negative externalities, these externalities should be priced to prevent social excessive consumption. This is illustrated in Figure 1 where Q, D, and $MC_i,\ i = \{p, s\}$, denote traffic volume, the demand function, which reflects the marginal benefit (MB), and marginal cost functions, respectively. In the absence of externalities, Q_p would be the optimal market equilibrium. However, the presence of a negative externality will mean that the true social marginal cost (MC_s) is higher than the one that individuals face and base their decisions on (MC_p). The difference between the two curves reflects the size of the externality measured in monetary terms, MEC, and to reach the optimal equilibrium, Q_s^*, a user charge of τ is required; that is, the difference in optimum between MC_s and MC_p. This internalization will result in the maximization of social surplus in the market in which it is applied.

[Figure 1 about here.]

Implementation of a pricing scheme for infrastructure use may have several objectives. Above, we have shown that internalizing the external costs results in an efficient allocation of resources in the transportation sector. Other objectives could be the generation of revenues to finance infrastructure investments, fairness, i.e. a polluter pays principle, or broader welfare efficiency. As the aim of this study is to developed a model for estimating the SRMC from road and rail traffic, we do not address and discuss the different motives of "infrastructure pricing" or the conflict between the marginal social cost principle and long-run incremental costs (see, e.g., Nash, 2005; Rothengatter, 2003; Sansom et al., 2001).

We define the SRMC of being exposed to road noise as the marginal social cost of one extra vehicle. Our model makes a distinction between types of vehicles, for instance cars or trucks, and between different technologies, for instance different types of breaks. However, in this section we outline our model in general form and keep other effects besides noise from changes in traffic volumes as constant. Let $L(Q,r,X)$ denote the noise level which is assumed to be a function of the traffic volume (Q), distance to the noise emission source (r), and a vector of other factors assumed to influence the noise level (X), e.g. traffic composition, presence of barriers, meteorological effects and ground properties. Moreover, let $C(L(Q,r,X))$ and $n(r)$ denote the individual cost function from noise exposure and the density of exposed individuals at different distances, and we can estimate the total social noise cost of traffic (S) as

$$S = \int_0^\infty C(L(Q,r,X))n(r)\mathrm{d}r. \tag{1}$$

By differentiating Eq. (1) with respect to traffic volume we obtain the expression for the short-run marginal cost,

$$M = \frac{\partial S}{\partial Q} = \int_0^\infty \frac{\partial C\,(L(\cdot))}{\partial L}\frac{\partial L(\cdot)}{\partial Q}n(r)\mathrm{d}r. \tag{2}$$

Equation (2) shows the theoretically correct expression for the SRMC. However, as discussed in Andersson and Ögren (2007, 2010) a first step to transform the model from a theoretical one to one that can be implemented is to acknowledge that data are usually available in discrete and not continuous form. That is, available data on noise levels and exposed individuals are constant within specific intervals (i). Moreover, Eq. (2) can be seen as the change in the total social cost for a unity change in the traffic volume, i.e. $\Delta Q = 1$. Let the change in noise level ΔL caused by the marginal vehicle be expressed as $\Delta L = \partial L(\cdot)/\partial Q$ and M can be written as

$$M = \sum_i c(L(\cdot))n(r)\Delta r \Delta L, \tag{3}$$

where $c(L(\cdot)) = \partial C(L(\cdot))/\partial L$. The final step is to define the number of exposed to the noise level L. Let $N(L)$ denote this number, corresponding to $n(r)\Delta r$ in Eq. (3), and the noise charge based on the SRMC principle is estimated as

$$T = \sum_L c(L(\cdot))N(L)\Delta L. \tag{4}$$

To distinguish this equation from other cost measures in this paper we refer to T in Eq. (4) as the *noise tariff*.

3 Noise indicators and emitters

Noise is normally measured as an A-weighted equivalent sound pressure level during a 24 hour (h) period with the unit dB, here denoted $L_{\text{AEq,24h}}$. For sleep disturbance the maximum level and number of events during the night is more relevant, but it is a difficult indicator to use for marginal cost calculations since only the loudest events contribute and the measure is two dimensional (both level and number of occurrences). The equivalent level may also be weighted according to when the noise event occurred, and in the EU a common noise indicator is L_{DEN} (from level day-evening-night), where nighttime events are treated as 10 dB louder and evening events as 5 dB louder than what they actually are.[1] The L_{DEN} is always higher than the $L_{\text{AEq,24h}}$ except if there are no events during the evening or night. If so, there would be no events to punish with 5 or 10 dB, and $L_{\text{AEq,24h}}$ and L_{DEN} become equal.

Swedish authorities usually use the standardized Nordic prediction methods for noise from road and railway traffic (Jonasson and Nielsen, 1996; Ringheim, 1996), for urban and infrastructure planning, noise mitigation measure planning, etc. Similar official methods exist for many countries, and there are also international methods such as HARMONOISE (de Vos et al., 2005) available. Since the Nordic method is used by the Swedish authorities, we also use it in this study. We have used it to calculate the acoustic source strength as the Sound Exposure Level (SEL). When comparing noise events and their contribution to the equivalent level during 24 h, it is convenient to use the SEL of the event, denoted L_{AE}. Short and loud events such as a car passing by at high speed can then be compared to more elongated events such as a slow and long freight train passing by. If the SEL of the two example events are the same, then their contribution to the overall equivalent level is the same.[2]

4 Empirical estimates

The empirical application of our model is based on data from Lerum, a municipality close to Gothenburg in the south-west of Sweden, located along the motorway (E20) and the main railway line (Västra stambanan) that connect Gothenburg and Stockholm. The data on Lerum that we use originate from two sources; the main source is Öhrström et al. (2005) in which health effects and annoyance from noise exposure were examined, and the other source is data from the *National Land Survey of Sweden*. The latter source contains property prices

[1] $L_{den} = 10 \log \left(\frac{12}{24} 10^{0.1 L_d} + \frac{4}{24} 10^{0.1 (L_e + 5)} + \frac{8}{24} 10^{0.1 (L_n + 10)} \right)$

[2] For a more comprehensive description of the SEL, see e.g. Fahy and Walker (1998).

and attributes and is used for the monetary evaluation of noise presented in section 4.3. It is the official registry used by the Swedish authorities for property taxation. The main source, i.e. Öhrström et al. (2005), is the one that provides us with information about noise levels and number of exposed individuals.

Based on calculations using the *Nordic method* of road and rail traffic noise levels for more than 24,000 inhabitants, Öhrström et al. (2005) chose a subset of the municipality around the railway and motorway for further investigation. This area is illustrated in Figure 2. Öhrström et al. distributed 2,751 questionnaires in this area with a return rate of 71%, i.e. 1,953 households answered the survey. In our study we use information from the survey about household size and the total number of exposed dwellings to estimate the total number of exposed. For those who answered the questionnaire, a refined set of calculations of $L_{\mathrm{AEq,24h}}$ and L_{den} noise levels of the road and railway traffic was carried out, and it is these values that are used in our study. Thus, road and railway noise levels are calculated on property level, which means that we have unusually rich data on noise levels. It should be noted, though, that non-responding households have been excluded from our analysis, since less information on noise exposure is available for them, and therefore the number of exposed is underestimated. This has an effect on the level of the tariffs estimated in this study, but not on the main objectives, i.e. developing and describing the model of noise tariffs and the sensitivity analysis.

4.1 Traffic situation

The traffic is concentrated to the two main transport routes through the area, the motorway E20 and the railway. Traffic flows were around 19,000 road vehicles (9% heavy vehicles) and 190 train passages per day at the time of the survey (Öhrström et al., 2005). The railway line transports both passenger and freight, 42 of the total are freight train passages. Approximately half of the freight traffic occurs at night (22–06). In a Swedish context the traffic volumes on both road and railway are high but not extreme.

The two transport routes are illustrated in Figure 2. In some parts of the area the railway and motorway are located close together, and in other areas they are more distant from each other. Also note that the railway route is slightly longer. The areas marked as urbanized in Figure 2 contain a majority of the exposed buildings, but there are buildings throughout the whole research area.

[Figure 2 about here.]

4.2 Acoustic differences

Comparing the SEL of two noise sources shows their relative contributions to the total noise level from all sources. In Table 1 we show the SEL calculated at a distance of 10 meters (m) from the emission source, denoted $L_{\mathrm{AE,10m}}$, for different vehicle types. The SEL at 10 meters is also given per metric ton or passenger by evenly distributing the noise over the full freight or passenger count.[3]

The vehicles types shown in Table 1 are the ones used for the SRMC calculations, and the vehicles that we have chosen are for passenger transport; a car with four passengers and a bus with 50 passengers. These are compared to three train sets, one high speed train set denoted X2 (200 km/h) for regional transport and two electric multiple units (EMU) for commuter service, X14 and X60. The train set X60 is a newer construction which has been in service just a few years and has lower acoustic emissions.[4] Note that the rail vehicles are limited to speeds below 135 km/h in the research area, but can travel faster. Increasing speed to the maximum increases the SEL per vehicle and passenger by about 1.5 dB for the commuter trains and about 3.5 dB for the high speed train.

For freight transport we have chosen a fully loaded 60 ton truck, with approximately 42 metric tons as payload, and a 500 m long freight train with 1500 tons of freight (using an electric locomotive of the Swedish Rc type). To examine the effect of using quieter technology we have also introduced low noise versions of the vehicles by equipping them with of-the-shelf low-noise technology. For the truck we assume an overall reduction of 5 dB, which is achievable mainly by using low-noise tires (Sandberg and Ejsmont, 2002). In the railway example we assume a retrofitting of the brakes from traditional cast iron to K-blocks, which on average lower the sound level by 8 dB according to the *International Union of Railways* (UIC) (Oertli and Hübner, 2010). Note that the noise reduction is for constant speed, i.e. it is not related to the noise radiated while braking. Using K-blocks causes less wheel corrugation compared to traditional brake blocks while braking, which in turn lowers the emissions during normal rolling conditions.

[Table 1 about here.]

Table 1 shows that the railway vehicles are typically more noisy than the road vehicles. This is an effect of larger vehicles and that the railway vehicles in some cases also travel faster than

[3]For a short discussion on typical load factors and occupancy rates see Section 4.4.

[4]For a description of the different train sets mentioned in this article see, e.g., Diehl and Nilsson (2009)

the road vehicles. The railway vehicles also transport more passengers and freight per vehicle, and by examining the SEL per unit of transported passenger or freight, we find that, whereas it varies for passenger traffic, the freight trains are still more noisy than the trucks. It is important to remember though that these calculations are for typical vehicles, individual vehicles (especially trucks and freight trains) may vary a lot compared to these averages. Moreover, Table 1 also reveals that the potential for improvement in noise levels is higher for railway than for road vehicles, since as much as 8 dB reduction is available by retrofitting brake systems.

4.3 Monetary values

As our measure of individuals' preferences for reducing noise levels we use estimates from a Swedish hedonic price study (Andersson et al., 2010b). In Andersson et al. (2010b) house-owners' willingness to pay (WTP) for a quieter living was estimated using the pooled data set consisting of Öhrström et al. (2005), which provides the noise levels of each property, and the *National Land Survey of Sweden*, which provides property prices and attributes. Since the results of the hedonic price study and conversion of its values to benefit measures for policy use have been reported in Andersson et al. (2010b) and Andersson et al. (2010a), we only provide a terse summary of the results in this paper.

The hedonic regression technique assumes that property prices are a function of the different utility-bearing attributes of the property, and, by studying the price variation due to variation in the composition of attributes implicit prices of the different attributes can be estimated (Rosen, 1974).[5] The hedonic price equation used to estimate marginal WTP in this study is based on the following functional form,

$$P_i = \gamma_0 \prod_{j=1}^{2} f(L_{ij}) \prod_{h=1}^{H} a_{ih}^{\gamma_h} + \varepsilon_i, \tag{5}$$

where

$$f(L_{ij}) = 1 + \frac{1 - b_j - (1 - b_j)\, e^{k_j L_{ij}}}{e^{30k_j} - 1}. \tag{6}$$

The noise variables are given by L_{ij} 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

[5] Formally, let P and $\mathbf{A} = [a_1, \ldots, a_n]$ denote the price and the vector of attributes of a property; then the hedonic price method suggests that

$$P = P(\mathbf{A}),$$

and the implicit price by

$$p_i = \frac{\partial P(\mathbf{A})}{\partial a_i}.$$

The hedonic method is already well documented and well known to many readers of this journal. We therefore choose not to describe the method in detail, but instead refer the readers not well acquainted with the method to the original source Rosen (1974), or Freeman (2003).

a_{ih}, and γ, b, and k are the parameters to be estimated. Descriptions and summary statistics of the different variables can be found in Table 8 in the appendix. In Eq. (6) 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 being between 0 and 1 and is estimated as,

$$k_j = \frac{e^{c_j}}{1+e^{c_j}}, \tag{7}$$

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. (5) makes it possible to assume not only different maximum effects of 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, are included.[6]

The regression results are shown in Table 2. We have also included the estimates of the "noise sensitivity depreciation index" (NSDI), which has evolved as the standard measure of the WTP in the hedonic noise literature.[7] Focusing first on the hedonic regression, which is based on non-linear estimation, we find that: (i) other property attributes besides the noise attributes are all statistically significant and with the expected signs, (ii) some of the neighborhood dummies are significant compared to the reference group (*Floda 2*), (iii) after controlling for the noise levels, the prices of properties within 150 meters from the motorway *E20* are not significantly affected by the motorway, and (iv) the distance variables to train stations and motorway entrance, *Dist. station* and *Dist. entrance*, are not statistically significant in the regression. Now, focusing on the variables of our main interest, the two noise variables, the relevant hypothesis-testing 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 significantly different from one, but not for railway noise. For the k-parameter, calculated using the coefficient estimate of c_j (see Eq. (7)), a higher value implies a more concave function, and a value close to zero implies an almost linear relationship between the noise level

[6]The total equivalent noise level is calculated as

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

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.

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

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

Thus, the NSDI is a measure of the percentage change in the price as a result of a unit change in the noise level (Nelson, 1980).

and the property price. The results, therefore, suggest a more concave relationship for rail than road noise. This is also reflected in the NSDI values where the effect of rail noise on property prices is lower than the effect of road noise for all noise levels except the highest (70 dB). These estimates lie in the upper end of the interval found for NSDI values in the literature, and for the higher noise levels our estimates exceed those values (e.g. Bateman et al., 2001; Nelson, 2008). Note, though, that most studies, as a result of the functional form chosen for the hedonic price function (i.e. a semi-logarithmic function) in their studies, have estimated a constant NSDI. Hence, those studies have ignored that the NSDI may vary with the baseline noise level.[8]

[Table 2 about here.]

The marginal implicit price based on the hedonic price equation in Eq. (5) is given by,

$$p_i = \gamma_0 f'(L_i) f(L_j) \prod_{h=1}^{H} a_h^{\gamma_h}, \tag{8}$$

where

$$\begin{aligned} f(L_j) &= 1 + \frac{1 - b_j - (1 - b_j)\, e^{k_j L_j}}{e^{30k_j} - 1}, & (9) \\ f'(L_i) &= -\frac{k_i (1 - b_i)\, e^{k_i L_i}}{e^{30k_i} - 1}, & (10) \end{aligned}$$

and where prim denotes the first derivative. Marginal WTP is evaluated for each noise source based on the mean value of the other variables and Eq. (8) may therefore be written as:

$$p_i = \beta_j f'(L_i), \tag{11}$$

where β_j is a constant and where the subscript j denotes that β varies between the noise variables.

The hedonic price regression provides us with the present value of the price variation. This should be converted into an annual benefit measure, both for the purpose of this paper and for use in benefit cost analysis (BCA). We also need to take into account tax effects, since failing to do so would underestimate the welfare effect (Niskanen and Hanke, 1977). Moreover, we also need to consider that the revealed WTP represents household WTP, not individual WTP. Equation (12) takes into account these three aspects,

$$V_i = \frac{(r + t\lambda) p_i}{n}, \tag{12}$$

[8] The preferred functional form in this study was chosen based on the results in Andersson et al. (2010b) in combination with evidence from the acoustical literature. However, the semi-logarithmic functional form was also used in Andersson et al. (2010b) and their results suggested that road and railway noise both had a significant negative effect on the property prices, with the effect from road noise being stronger. Estimated NSDI was 1.15 and 0.34 for road and railway noise, respectively.

where r, t, and n denote the real discount rate, the property tax, and the number of household members, respectively. The equation assumes eternal life of the property and the property tax is multiplied by λ, which denotes the proportion of the property value that is taxed in Sweden. The property tax at the time of the survey was 1% in Sweden and data from Lerum showed that the tax, on average was based on 50% of the value of the property, i.e. $t = 0.01$ and $\lambda = 0.5$. The Swedish property tax at the time of the survey was 1% which on average was based on 50% of the market value of the properties in Lerum, i.e. $t = 0.01$ and $\lambda = 0.5$.[9] The discount rate is based on the official Swedish rate proposed for BCA in the transport sector (SIKA, 2008), i.e. $r = 0.04$, whereas number of household members is based on the information in Öhrström et al. (2005), $n = 2.8$.[10]

Since the wealth level in Lerum is higher compared with the average level in Sweden, Andersson et al. (2010a) also suggested that the monetary estimates should be adjusted according to this difference before being used for Swedish policy purposes. Let Y_S and Y_L denote mean income for Sweden and Lerum, respectively, and θ the income elasticities for the WTP for a noise reduction, and our benefit measure 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. Based on empirical evidence, we follow Nellthorp et al. (2007) and let the income elasticity be equal to $\theta = 1$. Hence, the marginal WTP is given by multiplying Eq. (12) 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).

Another issue is whether estimated WTP from hedonic property studies reflects the total social cost of noise exposure. It has been argued that WTP estimates from these kinds of studies should be augmented by a value reflecting health costs, which are assumed not be known and/or considered by the property owners. The evidence is weak and we therefore base our estimations on the values derived from the hedonic price function without a health cost component. However, in our sensitivity analysis in the next section we show the effect on the tariffs from including the health component. The health cost component ($H(L_i)$) was estimated to be SEK 74.2 per dB for levels above 53 $L_{\text{AEq,24h}}$ and 0 below that level in Andersson et al. (2010a). The estimation of the annual social cost per person in the case of a change in level

[9]The point of departure is that the taxation should be based on 75% of the market value (SFS, 2001).

[10]Andersson et al. (2010a) examined the effect of discount rate chosen and showed that the difference in estimates was not negligible.

from l' to l'' can then be estimated by,

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

where $B(L_i)$ is given by Eq. (13). As the threshold value for when noise is regarded as disturbing is set at 50 dB in Sweden (SIKA, 2008), our estimations are also based on this value. Equation (14) includes $H(L_i)$ to show how the two cost component should be combined to estimate the total social cost, if we assume that the estimated WTP does not reflect the total social cost. Therefore, by simply dropping $H(L_i)$ from the equation, we can estimate the total social cost under the assumption that WTP does indeed reflect the total social cost. Table 3 shows the constants from Eqs. (10) and (13) and examples of welfare estimates. In the next section, as described, we have $H(L_i) = 0 \; \forall \; L_{\mathrm{AEq,24h}}$.

[Table 3 about here.]

4.4 Estimation of noise tariffs for road and railway traffic

The model for estimating the SRMC of a single vehicle passage is outlined in Eq. (4) and the empirical application along a certain route can be seen as a three-step process. First a noise map is created where the noise is predicted in the surrounding landscape using the traffic volume as input ($L(\cdot)$ in Eq. (4)). Then, for each inhabitant in the exposed area the marginal acoustical contribution of the vehicle under study must be calculated (ΔL in Eq. (4)). Finally all contributions must be summed up (over $N(L)$ exposed inhabitants) as the product of the marginal cost function $c\,(L(\cdot))$ and the marginal acoustical contribution ΔL.

For the noise maps we use, as previously explained, the values from Öhrström et al. (2005), which were calculated using the standardized Nordic methods (introduced in section 3) for road and railway noise prediction. The same methods are also used to calculate the acoustical contribution of the marginal vehicle, i.e. ΔL. The calculation of the noise maps for railway noise is straightforward since we usually, as is the case in this study, only have one source of emission, i.e. one railway line, contributing to the level. Noise maps for road traffic, however, are usually calculated based on the traffic of not only one route but all roads in a certain area. This is the relevant noise map for the estimation of the total social cost of road noise in the whole area, but it is not what we need for the estimation of the marginal effect of following a single route. The best approach would be to make a new noise map using only the infrastructure under study as a noise source, i.e. a noise map for the primary source along the path the marginal vehicle will follow. If that is not possible, it may be feasible to start from a normal noise map with all roads present, and exclude areas that are influenced by secondary roads.

The effect of secondary roads is illustrated in Figure 3, where the noise is displayed as contours of equal noise level for a simple flat landscape with one primary and one secondary road with less traffic. If we increase the traffic at the primary road, the sound level at position A will increase (as illustrated by the moving of the contour line), but at position B it will hardly increase at all. Therefore, it is important to calculate the marginal change including the effects of secondary sources for receiver positions where both the main and secondary roads make a substantial contribution to the sound level. A rule of thumb is that if the sound level from the secondary source is within 3 dB of the total level, then the secondary source can be regarded as dominant. In the Lerum example it was possible to identify and remove the approximately 10% of the population who were primarily exposed to noise from secondary roads.[11] Removing those inhabitants exposed to secondary sources leads to an underestimation of the SRMC, since the presence of a secondary source does not completely eliminate the WTP. Even when the noise from the secondary source is 10 dB louder than the noise from the primary source, the primary source is still audible. However, it is a reasonable assumption since a renewed noise calculation effort would be expensive and there is already data available for the total noise situation.

[Figure 3 about here.]

Regarding the calculation of the acoustical marginal effect Andersson and Ögren (2010) showed that ΔL is to a close approximation constant over the large area if one noise source is dominant. Table 4 shows the estimates from their calculation, which where based on the HARMONOISE (de Vos et al., 2005). Their estimates were based on road traffic and the results showed that whereas the total noise level depended on the total traffic volume, the increase was close to 1 dB over the estimated range. Thus, the change in noise level does not depend on distance, and this simplifies the calculation of the marginal change. However, the total noise level must still be calculated for the total traffic in all interesting receiver positions in order to calculate $c\left(L(\cdot)\right)$.

[Table 4 about here.]

To calculate the acoustical contribution of the marginal vehicle, i.e. ΔL, for the different vehicle types we use the standardized Nordic methods to calculate the SEL of a single vehicle

[11] For details see Andersson and Ögren (2010).

passage at a reference distance from the source of 10 meters (denoted $L_{\mathrm{AE,10m}}$), which is a standard part of the methods and different values for different vehicles and speeds are listed in tables. The acoustical marginal effect can then be calculated as

$$\Delta L = 10\log\left(10^{0.1L_{\mathrm{AE,10m}}} + 10^{0.1L_{\mathrm{AE,tot,10m}}}\right) - L_{\mathrm{AE,tot,10m}}, \tag{15}$$

where $L_{\mathrm{AE,tot,10m}}$ is the SEL of the total noise from all traffic at a distance of 10 m. This in turn can be calculated from the equivalent level $L_{\mathrm{AEq,24h}}$(calculated at a receiver point 10 meters from the source) using

$$L_{\mathrm{AE,tot,10m}} = L_{\mathrm{AEq,24h}} + 10\log(86400), \tag{16}$$

where the constant 86,400 is the number of seconds during 24 hours. The marginal effect ΔL is typically very small, in the range of 1–0.001 dB. It is not negligible, however, since it is multiplied by the valuation in SEK per person and year and then summed up over all exposed inhabitants.

In order to compare the SRMC of the road and railway traffic it is assumed that both occupy the same corridor through the area and expose an equal amount of inhabitants to the same noise level. In reality it is also important ascertain if there is a systematic difference in the population distribution around railways as opposed to roads, and possibly also systematic differences in acoustical properties such as screening by terrain and buildings, but this is not investigated further here. It is also assumed that the total noise load over the area is the same, which would correspond to a slightly increased railway traffic volume compared to the real situation.

As can be seen in Figure 2, the railway and motorway partly occupy different corridors through Lerum, and in order not to make the comparison between them cluttered by the details of the geography, we use the motorway corridor for both transport modes when calculating the number of exposed inhabitants. Using the railway corridor reduce the number of exposed by about 17%, but not the shape of the distribution shown below, i.e. a function estimated from the population exposed to railway noise would have much the same shape as the function in Figure 4, which is explained below.

In order to simplify the sensitivity analysis in the following section the number of inhabitants exposed at different equivalent levels along the road corridor is sorted into 1 dB intervals, and a simple function is fitted to the data using the least squares method. The functional form of the population distribution then becomes

$$N(L) = 10^{-0.102L+8.20},\ L \geq 50, \tag{17}$$

where N is the predicted number of exposed inhabitants and L the equivalent noise level ($L_{\mathrm{AEq,24h}}$). This function is then divided into 1 dB intervals, following the discussion in section 2 about discrete information on the number of exposed, and is plotted together with the original data in Figure 4. Using this simplified population distribution gives an error of less than 5% on the calculated SRMC. Note that it is important not to use this function to estimate the number of inhabitants at very low levels since it grows very rapidly as the level decreases, hence the limitation to equivalent levels of 50 dB or higher.

[Figure 4 about here.]

The estimates of the noise tariffs that reflect the SRMC are shown in Table 5. Tariffs are estimated for the vehicles presented in section 3 and the upper and lower parts of the table show the estimates for the passenger and freight traffic, respectively.[12] In order to calculate the SRMC per passenger and per metric ton (1000 kg) of cargo, it is assumed that the vehicles are filled to their maximum capacity. In the EU the actual passenger occupancy rate is about 20–40% for long distance bus travel and 25–50% for railway travel, and the typical load factor for both road and railway freight is 40–60% (EEA, 2010, TERM 029 and 030 indicators). Our assumption facilitates calculations and has negligible effects on the relationships shown in Table 5.

[Table 5 about here.]

For passenger traffic we find that the SRMC for railway traffic, with the exception of *X60 EMU*, is higher than for road traffic. This would suggest higher noise tariffs for railway than for road traffic if they were to reflect the SRMC. However, when comparing the SRMC per passenger, it is in general lower for rail transport. This is explained partly by lower noise emissions per passenger for the rail vehicles (as discussed in section 4.2) and partly by the lower valuation for the cost of noise from railways. As an example, a comparison between bus transport and the X14 commuter train shows that they have approximately the same noise emissions per passenger, but lower SRMC per passenger for the X14 due to the differences in valuation function for road and rail.

We find similar results for freight traffic, i.e. higher on vehicle level and lower per metric ton for railway compared with road traffic. To illustrate the effect of using low-noise technology the same two examples as in section 3 are included. The low noise truck is assumed to be

[12] Swedish trucks are allowed a maximum total weight of approximately 60 tons over much of the Swedish road network, which is why the mass of the freight seems high in an EU perspective.

5 dB quieter than the average truck, which can be achieved mainly by using low noise tires and effective noise mufflers (Sandberg and Ejsmont, 2002). For the rail example, an identical train as the reference is assumed, except that the brakes have been retrofitted with K-blocks, a measure that on average reduces the noise by 8 dB according to UIC (Oertli and Hübner, 2010) by making the wheels smoother and thereby reducing the rolling noise. In both cases the economic incentive to reduce the noise emission is strong, if it is available within a hypothetical noise charging system.

5 Sensitivity of tariffs due to changes in parameter levels

In order to explain which parameters influence the SRMC, results from a sensitivity analysis are presented in Tables 6 and 7. By examining which parameters influence the SRMC, we obtain information about whether adjustments of the tariffs are necessary when the corresponding parameter varies. To illustrate the effect, we have chosen the truck and the freight train with the same parameters as in Table 5 in the previous section as our reference cases, and then varied the parameters within $\pm 50\%$ or ± 1.8dB. The results in Table 6 are for freight traffic, but the results are close to identical for passenger transport (which therefore are omitted here but available upon request from the authors).

The first part of Table 5 shows the effect of a change in traffic volume. Estimating tariffs that should reflect the SRMC, a change of $\pm 10\%$ in the traffic volume is of main interest (since larger changes may require changes to the infrastructure). The results show that changes in the total traffic volume only have a minor influence on the SRMC. This is a result of two opposite effects; the monetary values and the number of inhabitants exposed to high noise levels increase if the total traffic volume increases, but the marginal acoustical contribution of a single vehicle decreases. This insensitivity is not only the case for the smaller changes in the total traffic volume, but also for larger ones; an increase in the total traffic volume of 50% will only increase the SRMC by about 1%.

[Table 6 about here.]

The second and the third parts of Table 5 show the effect of changing the noise level of the marginal vehicle and of the whole fleet (including the marginal vehicle). Changing the noise level of only the marginal vehicle itself has the expected effect; i.e. it is in line with the results in Table 5 of using "quiet technology".[13] Manipulating the noise emissions of the whole fleet

[13] A change of +0.4 dB corresponds to increasing the radiated sound power measured in Watts by 10% (+1.0 dB to 25% and +1.8 dB to

is slightly more complex, since this also affects the number of exposed at different levels. The effect of changing the noise level of the whole fleet will therefore be larger compared with changing it only for the marginal vehicle. Finally, it is evident that variation in population density has a strong influence on the SRMC.

In Table 7 we show the results of a sensitivity analysis on the assumption about the monetary values used to calculate the SRMC. Again we use the truck and freight train as our examples and the reference case is the values for these vehicles from Table 5. We start by investigating the effect of including the health component in Eq. (14). The effect on the estimates is substantially higher for the railway estimates; railway estimates increase by 87% compared with 11% for the road estimates. This is explained by the fact that the health component has a larger influence at low noise levels for railway noise as shown in Table 3. Since most individuals are exposed to low noise levels, this will influence the estimates.

The second analysis is done by interchanging the monetary values of the two noise sources, i.e. road is treated as railway and the other way around. Due to the difference in the effect on the property prices of the two noise sources, interchanging the preferences also has large effects.

Finally, we also examine the effect of using the official Swedish monetary noise values for benefit-cost analysis (BCA), denoted ASEK 4 (SIKA, 2008). The official Swedish values are based on individuals' preferences for reducing road noise and were used in Andersson and Ögren (2007, 2010). Thus, the estimates in Andersson and Ögren (2007) for railway noise were based on preferences for road noise. They also estimated the SRMC using the 5 dB railway bonus, a bonus commonly used to take into account the fact that individuals are less annoyed by railway than by road noise. The results suggest that using the monetary values for road noise when estimating the SRMC for railway noise will overestimate the SRMC compared with when monetary values for railway noise are used. The estimates are more than seven times higher SRMC for the freight train and even if the introduction of the railway bonus of 5 dB lowers the SRMC, it is still about twice as high as the values shown in Table 5.

[Table 7 about here.]

50%).

6 Discussion

In this study we have described how to estimate the SRMC for road and railway noise. The estimation method that has been outlined is based on standardized calculation methods for total noise levels and monetary cost estimates from well established evaluation methods. We have used official calculation methods and monetary values for Sweden, but the estimation method for the SRMC can be directly applied using other standardized noise calculation methods and monetary values. Hence, this study has shown that the method already available for the calculation of total noise levels and the evaluation of the social cost of noise can be extended to estimate the marginal effect as well. This is an important finding since it enables policy makers to price noise externalities in an appropriate way. The EU has decided that infrastructure-user charges should be based on the SRMC principle (European Commission, 1998). It is important that these estimates are based on solid calculation methods and that the estimation of the SRMC is transparent. The estimation method in this study takes both of these into account.

For the objective of this study, the absolute level of the estimated SRMC is of limited interest. As described, the data on number of exposed from Öhrström et al. (2005) provide an underestimation of the actual number, and we also choose not to include a health cost component in our estimates, since the evidence that this is not already included in the WTP estimates is weak (Andersson et al., 2010a). Thus, our estimates can be seen as a conservative estimate of the true social marginal cost for an area like Lerum. Moreover, the levels of the estimates are only relevant for areas such as Lerum since they depend on the actual traffic situation and the distribution of the number exposed. For instance, Lerum is relative densely populated for Sweden and the SRMC would, therefore, be lower for many other areas in Sweden (Andersson and Ögren, 2010). Previous research has also shown the sensitivity of the chosen threshold level (Andersson and Ögren, 2007, 2010). In this study we have chosen the Swedish official level of 50 dB, whereas many other countries use 55 dB as their threshold level.

The objective of this study is, however, to outline a model to estimate the SRMC of road and railway noise and to examine its sensitivity to different aspects concerning traffic situation, quiet technology and preferences for quiet (i.e. monetary values used). Regarding the sensitivity, we find that increasing the total traffic volume has only a minor influence on the estimated marginal cost, since the total noise exposure increases and the marginal WTP is higher, but the marginal acoustical contribution of each vehicle decreases. Therefore, charges based on the SRMC are relatively stable when the traffic volume changes. The SRMC is, however, quite sensitive to the number exposed and to the monetary values used. The latter suggest that benefits transfer, i.e. using monetary values that are based on road noise for railway noise, should be done with caution or not at all. We have also shown that the use of quiet technology can have a significant effect on the SRMC; the SRMC was reduced to one third and one sixth for the truck and freight train, respectively, in our example.

The fact that our model is able to differentiate between different vehicles, not only modes of transport, and even technologies is an important finding. It is also important that the noise charges give the operators the right incentives to choose their optimal allocation of noise-reducing activities. A noise charge that differentiates between not only vehicle types but also technologies gives the operators incentives to reduce their noise emission to the point when it is no longer optimal for them to reduce their emission.[14] The model presented in this study differentiates type and technology, area (number and distribution of individuals exposed), and traffic situation. It can also easily be extended to differentiate day, evening and nighttime emissions in accordance with the L_{DEN}. A more diversified model is preferred since it better reflects the true SRMC, but it may be too costly to implement since it will require advanced technical solutions and monitoring to make it work. In the end BCA should decide on the optimal level of diversification of the charges.

Even if the levels of the SRMC are higher for railway than for road vehicles for most of our vehicle types, the SRMC per passenger and ton is lower. This is mainly explained by the lower WTP expressed in the valuation functions for rail; the difference in acoustic emissions per ton/passenger compared to road transport is less important. Note, however, that this paper assumes that both transport modes on average have a similar degree of utilization (i.e. similar percentage of empty seats or unloaded freight wagons). The comparisons between total levels and per passenger and ton of freight show the importance of providing policy makers and operators with both levels, since it mitigates the risk of different groups trying to influence the charges based on non-efficient arguments.

A further issue regarding noise charges in transport is acceptability. Since there are no direct benefits for the road and railway users from noise charges, their acceptability of such charges may be low. For instance, congestion charges can reduce travel time, which is a benefit for those who pay these charges, or charges for maintenance may be accepted if the users believe that they will be used for actual maintenance. Regarding noise charges, there may not be, or the users may not believe that there will be, any benefits from paying the charge. To increase

[14] An alternative for the operators is to use a different route where the SRMC is lower.

acceptability we believe that differentiated charges according to the above, but also transparent charges, are necessary. Our model can obtain this and further research should be focused on determining rules of thumbs for the number of exposed in different areas.

Appendix: Descriptive statistics

[Table 8 about here.]

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Figures

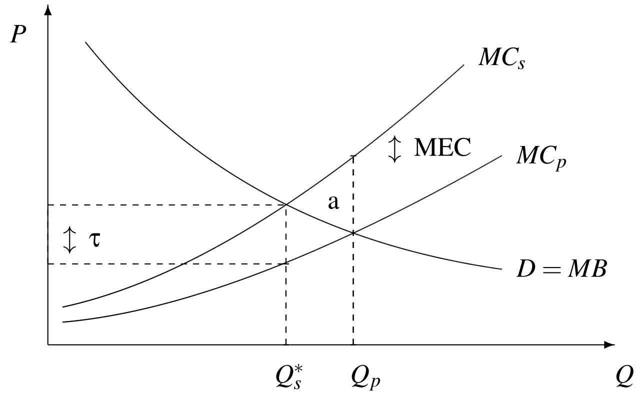

Figure 1: Marginal cost pricing and economic efficiency

Figure 2: Sketched map over the research area

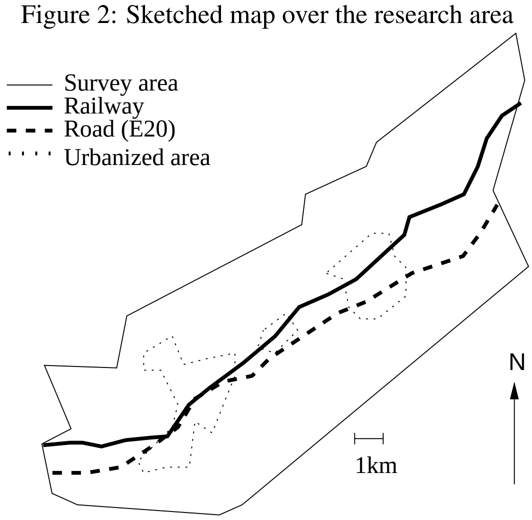

Figure 3: The effect on the noise level contours when the traffic is increased on the main road. (Source: Andersson and Ögren (2010))

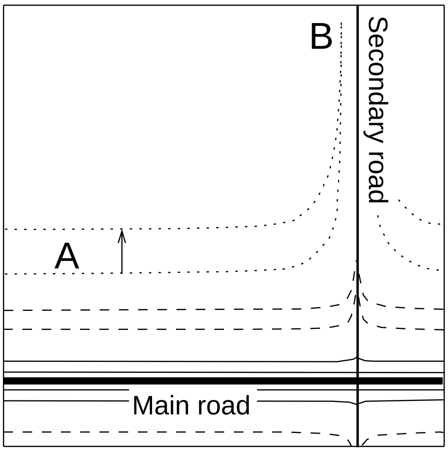

Figure 4: Estimation of number of exposed inhabitants as a function of $L_{\mathrm{AEq,24h}}$in 1 dB intervals.

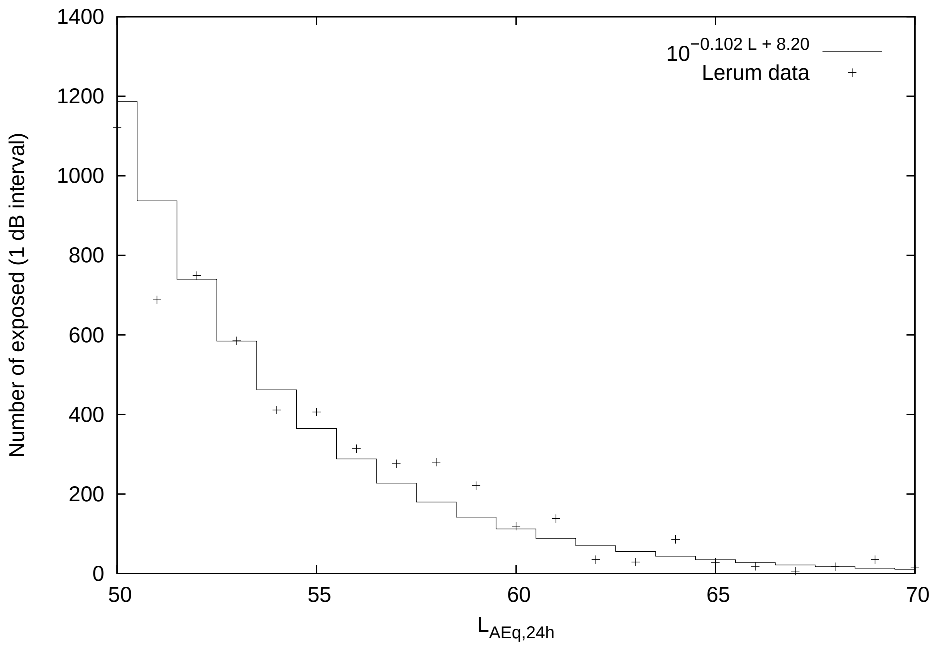

Table 1: Sound Exposure Level (SEL) calculated per vehicle and unit

Vehicle	Speed km/h	Passengers/ Freight[a]	SEL at 10 m, dB per vehicle	per unit[b]
Passenger traffic				
Car	110	4	82.1	76.0
Bus	90	50	88.2	71.2
X2 high speed	135	310	98.0	73.1
X14 EMU	135	350	97.0	71.5
X60 EMU	135	370	90.6	64.9
Freight traffic				
Truck	90	42	88.2	71.9
Truck (low noise)	90	42	83.1	66.9
Freight train	90	1500	106.9	75.1
F. tr. (K-blocks)	90	1500	98.9	67.1

SEL at a distance of 10 m from emission source ($L_{\mathrm{AE,10m}}$).
a: Metric ton (1,000 kg)
b: Per passenger and metric ton for passenger and freight traffic, respectively.

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).
$k_j = e^{c_j}/(1+e^{c_j})$
$\mathrm{NSDI} = |(\partial P/\partial L)(100/P)|$

Table 3: Welfare estimates: SEK/person/year in 2004 price level

	Constants[a]			Change			
	β	*k*	*b*	High	Low	w/o health	w/ health
Road	1 938 866	0.031	0.560	56	55	363	437
				66	65	495	569
				75	74	654	729
Railway	2 097 665	0.254	0.506	56	55	24	98
				66	65	308	382
				75	74	3027	3101

a: *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.
Average exchange rate 2004: USD 1 = SEK 7.35 and EUR 1 = SEK 9.13 (www.riksbank.se, 1/27/11)

Table 4: Marginal change in noise level as a function of distance

	Traffic	20 m	50 m	100 m	200 m
Flat	20,000	66.9	56.6	49.0	43.2
ground	16,000	65.9	55.6	48.0	42.2
	Diff.	1.0	1.0	1.0	1.0
Screening	20,000	53.1	50.3	46.9	43.5
	16,000	52.1	49.4	45.9	42.6
	Diff.	1.0	0.9	1.0	0.9

Source: Andersson and Ögren (2010)

Table 5: Noise tariffs calculated per vehicle and unit

	Speed	Passengers/	Tariff, SEK/km	
Vehicle	km/h	Freight[a]	per vehicle	per unit[b]
Passenger traffic				
Car	110	4	0.06	0.0148
Bus	90	50	0.24	0.0048
X2 high speed	135	310	0.37	0.0012
X14 EMU	135	350	0.29	0.0008
X60 EMU	135	370	0.07	0.0002
Freight traffic				
Truck	90	42	0.24	0.0057
Truck (low noise)	90	42	0.08	0.0018
Freight train	90	1500	2.82	0.0019
F. tr. (K-blocks)	90	1500	0.45	0.0003

SEK price level 2004.
a: Number of passenger and metric ton of freight, respectively.
b: Per passenger and metric ton for passenger and freight traffic, respectively.

Table 6: Noise tariffs and parameter variability

	Changes as percent and dB						
	-50%	-25%	-10%	±0	+10%	+25%	+50%
Parameter	-1.8dB	-1.0dB	-0.4dB	±0	+0.4dB	+1.0dB	+1.8dB
Total traffic volume							
Railway	0.988	0.994	0.997	1.000	1.003	1.006	1.011
Road	0.992	0.996	0.998	1.000	1.002	1.004	1.008
Noise level of vehicle							
Railway	0.668	0.801	0.910	1.000	1.099	1.248	1.494
Road	0.667	0.800	0.909	1.000	1.100	1.250	1.500
Noise level of fleet							
Railway	0.661	0.796	0.907	1.000	1.102	1.256	1.512
Road	0.661	0.796	0.907	1.000	1.102	1.256	1.512
Number of exposed							
Railway	0.667	0.800	0.909	1.000	1.100	1.250	1.500
Road	0.667	0.800	0.909	1.000	1.100	1.250	1.500

Railway and Road refer to a 1,500 and a 60 metric ton vehicle, respectively.

Table 7: Noise tariffs and monetary preference estimates

Parameter	Reference	Railway	Road
Including health comp.	1.00	1.87	1.11
Switch val. road/rail	1.00	8.28	0.12
ASEK 4[a] val.	1.00	7.51	0.91
ASEK 4[a] (5 dB rail bonus)	1.00	2.21	0.91

Railway and Road refer to a 1,500 and a 60 metric ton vehicle, respectively.
a: ASEK 4 refers to the official Swedish monetary noise values (SIKA, 2008).

Table 8: Descriptive statistics

Variable	**Description**	**Mean value**
Price	Property price in thousand SEK and 2004	1917.913
	price level	(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