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Charging the Polluters

A Pricing Model for Road and Railway Noise

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Abstract

By combining standardised calculation methods for total noise levels and monetary estimates from well-established evaluation methods, this study outlines a model to estimate the short-run marginal cost (SRMC) for road and railway noise that is able to differentiate not only modes of transport, but also vehicles and technologies. Several sensitivity tests run for the SRMC show that estimates are insensitive to traffic volume, sensitive to the number of exposed individuals, and sensitive to the monetary values used. Results also show that the use of quiet technology can have a significant effect on the SRMC.

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1.0 Introduction

The transportation sector provides many benefits to society. For instance, it is crucial for business to facilitate geographical specialisation and physical access to markets, and for individuals to access labour markets, to consume many of the goods and services produced, and to socialise. 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 (that is, the problem of congestion), and noise nuisance. These negative impacts are significant from a social perspective, and transportation 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 per cent 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 fifteen European Union (EU) members at the time, and Switzerland and Norway, were estimated to be EUR 650 billion, which corresponded to 7.3 per cent 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 maximised. 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 to 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 per cent of the population within the EU is 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 (SOU, 1993; Kalivoda *et al.*, 2003; Lundström *et al.*, 2003). 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 about 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 the exposure of others 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 internalised 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. For instance, since transport noise has fewer negative effects from a transport planning perspective than, for instance, congestion, and since there is a relatively long time period between exposure and negative health effects (Jarup *et al.*, 2008; Barregard *et al.*, 2009), 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 will be 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 to the combined effect of urbanisation and increased traffic (Nijland *et al.*, 2003).

Other noise mitigation measures that address the propagation of noise, such as noise barriers, window insulation, and so on, are less cost-efficient compared to measures taken at the source (Oertli, 2000; de Vos, 2003). Propagation measures are also usually not under the influence of, for instance, train or truck operators. However, there are some measures that the infrastructure manager can take to reduce the noise at the source, such as low-noise road pavements and acoustic rail grinding to remove rail roughness. These measures will lower the SRMC of all passing vehicles, but are again outside the influence of the vehicle operators.

The aim of this study is to present 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 and are able to differentiate the estimates based on low-noise technology. As one example of low-noise technology, we use the possibility of retrofitting of brakes on train sets. A pricing model that is able to differentiate charges based on noise costs related to different technologies is of great policy relevance; the European Commission is proposing to make it compulsory to differentiate access charges for railway infrastructure according to the noise costs of different types of brakes (European Commission, 2008).

This article draws on previous findings in Andersson and Ögren (2007, 2011) 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. Another important contribution is the comparison between the empirical estimates of the SRMC for railway and road noise. Since we create an environment where other factors besides traffic-related ones are constant, the direct comparison highlights actual differences in estimates between the two models. 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 has huge policy relevance since it provides important insight into when the estimates of the SRMC need to be adjusted, based on changes of the levels of the parameters. Moreover, studies by Andersson and Ögren (2007, 2011) were based on benefit measures of road-noise abatement, since useful measures were missing for rail noise. As a consequence, the 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 noise differs, and so does their WTP (willingness to pay) to reduce their exposure to the two sources (Miedema and Oudshoorn, 2001; Day *et al.*, 2007; Andersson *et al.*, 2010). 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:

1. to estimate and compare the SRMC for road and railway noise of several vehicle types and technologies;
2. to outline how to calculate the marginal acoustical effects of road and rail traffic noise; and
3. to conduct several sensitivity tests.

In order to conduct the empirical analysis, we use Swedish data.

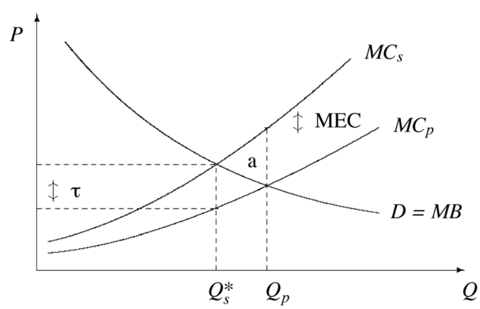
The article is organised as follows. In Section 2 we briefly describe the motivation of the marginal cost principle and derive our theoretical model. We thereafter explain and present the most common noise indicators, and describe how the marginal acoustical effects of road and rail traffic are estimated. 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.

2.0 Internalising the External Cost

The EU decision to base infrastructure user charges on the social marginal cost principle (European Commission, 1998) is based on Pigou's (1920) 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 internalisation will result in the maximisation of social surplus in the market in which it is applied.

Implementation of a pricing scheme for infrastructure use may have several objectives. Above, we have shown that internalising the external costs results in an efficient allocation of resources in the transportation sector. Other objectives could be the

Figure 1
Marginal Cost Pricing and Economic Efficiency



generation of revenues to finance infrastructure investments, fairness — that is, a 'polluter pays' principle — or broader welfare efficiency. As the aim of this study is to outline 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, for instance, Sansom *et al.*, 2001; Rothengatter, 2003; Nash, 2005).

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 brakes). 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); for instance, 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 equation (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, 2011), 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. The integral over distance in equation (2) is therefore divided into i discrete sections. Within each i-interval the noise level is equal for all individuals in that particular interval, which is in line with available data on noise levels and exposed individuals. Moreover, equation (2) can be seen as the change in the total social cost for a unity change in the traffic volume — that is, $\Delta Q = 1$. 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 individuals to the noise level L. Let $N(L)$ denote this number, corresponding to $n(r)\Delta r$ in equation (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 equation (4) as the *noise tariff*.

3.0 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 $L_{\mathrm{AEq,24h}}$. 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 night-time events are treated as 10 dB louder and evening events as 5 dB louder than they actually are.[1] The L_{DEN} is always higher than the $L_{\mathrm{AEq,24h}}$ 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 $L_{\mathrm{AEq,24h}}$ and L_{DEN} become equal. It is possible to differentiate the SRMC for the three time periods using the L_{DEN} indicator (Andersson and Ögren, 2007).

Swedish authorities usually employ the standardised Nordic prediction methods for noise from road and railway traffic (Jonasson and Nielsen, 1996; Ringheim, 1996). The methods describe how to calculate the noise level at a receiver point using data on the road pavement or rail roughness, together with traffic flow and speed as input (number of vehicles per 24 hours, vehicle type distribution, and so on). Both methods then correct for distance from the source, height above ground, and screening by terrain, buildings or sound barriers, in order to calculate the maximum and equivalent noise level at the receiver. Other parameters than those mentioned above can influence the noise level, such as the meteorological conditions (Embleton, 1996) and the driving behaviour (Sandberg and Ejsmont, 2002), but they are not part of the standardised methods and are not studied further here.

The main field of use for the Nordic methods is in urban development and infrastructure planning, but the methods are also used by the authorities for planning noise mitigation measures such as noise barriers or speed reductions. 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 hours, 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 prolonged events such as a slow and long freight train passing by. If the SELs of the two example events are the same, then their contribution to the overall equivalent level is the same.[2]

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

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

4.0 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 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 (Öhrström *et al.*, 2005) is the one that provides us with information about noise levels and the 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.* (2005) distributed 2,751 questionnaires in this area with a return rate of 71 per cent — that is, 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 individuals. 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 individuals is underestimated. This has an effect on the level of the tariffs estimated in this study, but not on the main objectives — that is, developing and describing the model of noise tariffs, and the sensitivity analysis.

4.1 Traffic situation

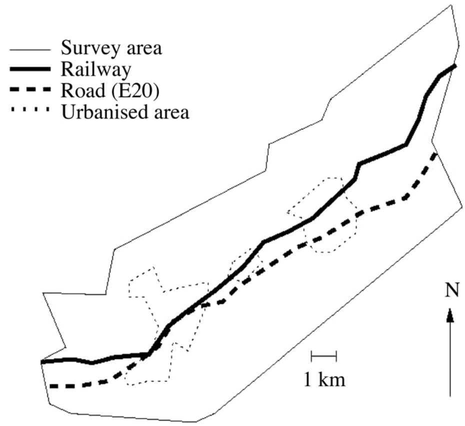
The traffic is concentrated on the two main transport routes through the area: the motorway E20 and the railway. Traffic flows were around 19,000 road vehicles (9 per cent heavy vehicles) and 190 train passages per day at the time of the survey (Öhrström *et al.*, 2005). The railway line transports both passenger and freight; forty-two of the total are freight train passages. Approximately half of the freight traffic occurs at night (22:00 to 06:00). 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 urbanised in Figure 2 contain a majority of the exposed buildings, but there are buildings throughout the whole research area.

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 m (metres) from the emission source, denoted $L_{\mathrm{AE,10m}}$, for different vehicle types. The SEL

Figure 2
Sketched Map Over the Research Area



at 10 m is also given per metric ton or passenger by evenly distributing the noise over the full freight or passenger count.[3]

The vehicle 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 fifty 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 1,500 tons of freight (using an electric locomotive of the Swedish Rc type).[5] To examine the effect of using quieter technology, we have also introduced low-noise versions of the vehicles by equipping them with off-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 lowers 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 — that is, it is not related to the noise radiated while braking. Using K-blocks causes less wheel corrugation compared to

[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, for instance, Diehl and Nilsson (2009).
[5]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.

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	1,500	106.9	75.1
F.tr. (K-blocks)	90	1,500	98.9	67.1

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

traditional brake blocks while braking, which in turn lowers the emissions during normal rolling conditions.

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 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*., 2010). In Andersson *et al*. (2010), house-owners' willingness to pay (WTP) for 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*. (2013) and Andersson *et al*. (2010), we only provide the monetary estimates obtained from these studies and a short description of them in this paper.[6]

[6]Andersson *et al*. (2010) provide the present value of the effect from noise exposure on the property value. For details on the conversion of these values to an annualised value using discount rates and tax effects, and a discussion and description of the health cost component discussed below, see Andersson *et al*. (2013).

Table 2
Welfare Estimates: SEK/Person/Year in 2004 Price Level

	Change		*REBUS* *w/o health*	*REBUS* *w/health*	*ASEK (w/health)*
Road	56	55	363	437	258
	66	65	495	569	568
	75	74	654	729	3,343
Railway	56	55	24	98	NA
	66	65	308	382	NA
	75	74	3,027	3,101	NA

Notes: Average exchange rate 2004: USD 1 = SEK 7.35 and EUR 1 = SEK 9.13 (www.riksbank.se, 27 January 2011).

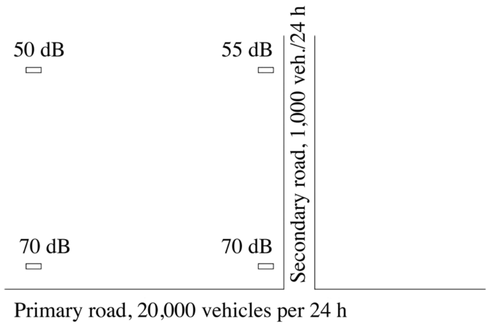
Examples of the monetary welfare estimates are shown in Table 2. The values are per person and year, and in addition to the estimates from Andersson *et al*. (2010, 2013) (REBUS), the table also contains the official Swedish monetary noise values (ASEK). The latter are the current values recommended for use in benefit–cost analysis (BCA) and are based on the effect from road noise on property prices (SIKA, 2008). We prefer the values from Andersson *et al*. (2010, 2013) both because of the quality of the data, and since they provide preference estimates for both railway and road noise. We also employ the ASEK values, though, to test the sensitivity of the SRMC from using different monetary values. Focusing on the REBUS estimates, we provide estimates with and without a health component. It has been argued that WTP estimates from hedonic property studies do not reflect the total social cost of noise exposure, and therefore should be augmented by a value reflecting health costs, since the latter are assumed not to 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 we show the effect on the tariffs from including the health component. The health cost component was estimated to be SEK 74.2 per dB for levels above 53 $L_{\mathrm{AEq,24h}}$ and 0 below that level in Andersson *et al*. (2013).

The results for REBUS in Table 2 reveal a significantly lower marginal WTP for railways than road noise at low levels, and a significantly higher marginal WTP for railways than road noise at high levels. That is, the relationship between the noise level and the marginal cost is more progressive for railways than road noise which has important policy implications, not only for the noise tariffs estimated in this study, but also for BCA using these estimates. Most individuals live at low noise levels (Andersson and Ögren, 2007, 2011), which means that differences in WTP at low levels have a large impact on the welfare estimates. The strong convex relationship suggested by ASEK for road noise is not supported by other findings in the literature and the estimates have been criticised (see Andersson *et al*., 2013).

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 equation (4) and the empirical application along a certain route can be seen as a three-step

Figure 3
The Effect on the Noise Level of Secondary Roads



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 equation (4)). Then, for each inhabitant in the exposed area the marginal acoustical contribution of the vehicle under study must be calculated (ΔL in equation (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 standardised 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 — that is, 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 — that is, one railway line — contributing to the level. Noise maps for road traffic, however, are usually calculated based on the traffic of not just 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 unfortunately makes it more difficult to calculate the marginal effect of following a single route. The marginal change in noise level will not be determined only by the traffic flow along the route under study, but the traffic on other nearby roads as well.

The effect of secondary roads is illustrated in Figure 3, where the noise is illustrated at four buildings in a simple flat landscape with one primary and one secondary road with less traffic. If we use the values from a noise map including all roads as noise sources, the level at the upper right position will be affected by the secondary road; at the other three positions the primary source is dominant. Estimating the WTP directly using 55 dB as opposed to 50 dB, which is the correct value concerning a marginal vehicle on the primary road, gives a slightly overestimated marginal cost. In the Lerum example it was possible to identify and remove approximately 10 per cent of the population who were primarily exposed to noise from secondary roads.[7] Removing

[7]For details, see Andersson and Ögren (2011).

Table 3
Marginal Change in Noise Level as a Function of Distance

	Traffic	*20 m*	*50 m*	*100 m*	*200 m*
Flat ground	20,000	66.9	56.6	49.0	43.2
	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 (2011).

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 most likely audible. However, our approach is a reasonable procedure, since a renewed noise calculation effort would be expensive and the main road dominates the noise level at 90 per cent of the area under study.

Regarding the calculation of the acoustical marginal effect, Andersson and Ögren (2011) showed that ΔL is a close approximation constant over the large area if one noise source is dominant. Table 3 shows the estimates from their calculation, which were 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 $c(L(\cdot))$.

To calculate the acoustical contribution of the marginal vehicle — that is, ΔL – for the different vehicle types, we use the standardised Nordic methods to calculate the SEL of a single vehicle passage at a reference distance from the source of 10 m (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:

$$\Delta L = 10\log\left(10^{0.1L_{\mathrm{AE,10m}}} + 10^{0.1L_{\mathrm{AE,tot,10m}}}\right) - L_{\mathrm{AE,tot,10m}}, \tag{5}$$

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 m from the source) using:

$$L_{\mathrm{AE,tot,10m}} = L_{\mathrm{AEq,24h}} + 10\log(86{,}400), \tag{6}$$

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 0.001–1.0 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 number of inhabitants to the same noise level. In reality it is also important to 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. These assumptions are made to make the comparison between the two modes meaningful — that is, besides traffic-related parameters everything else is equal.

As Figure 2 shows, 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 reduces the number of exposed individuals by about 17 per cent, but not the shape of the distribution shown below — that is, 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:

$$N(L) = 10^{-0.102L + 8.20}, \quad L \geqslant 50, \tag{7}$$

Figure 4

Estimation of Number of Exposed Inhabitants as a Function of $L_{AEq,24h}$ *in 1 dB Intervals*

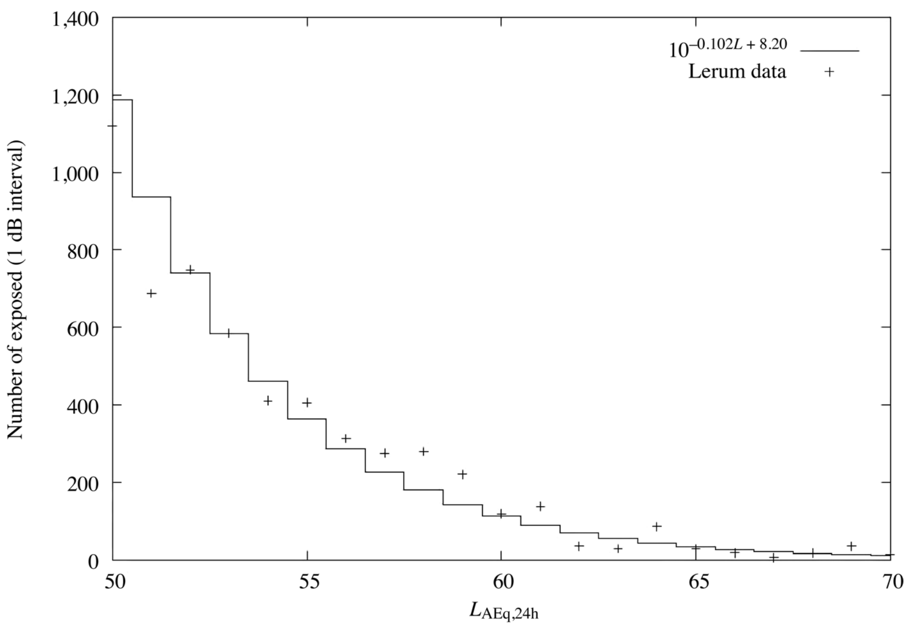


Table 4
Noise Tariffs Calculated Per Vehicle and Unit

Vehicle	*Speed (km/h)*	*Passengers/ freight*[a]	*Tariff, SEK/km*	
			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	1,500	2.82	0.0019
F.tr. (K-blocks)	90	1,500	0.45	0.0003

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

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 individuals, and is plotted together with the original data in Figure 4. Using this simplified population distribution gives an error of less than 5 per cent 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.

Table 4 shows the estimates of the noise tariffs that reflect the SRMC. Tariffs are estimated for the vehicles presented in Section 3, and the upper and lower parts of Table 4 show the estimates for the passenger and freight traffic, respectively. In order to calculate the SRMC per passenger and per metric ton (1,000 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 per cent for long-distance bus travel and 25–50 per cent for railway travel, and the typical load factor for both road and railway freight is 40–60 per cent (EEA, 2010, TERM 029 and 030 indicators). Our assumption facilitates calculations and has negligible effects on the relationships shown in Table 4.

For passenger traffic we find that the SRMC per vehicle for railway traffic, with the exception of *X60 EMU*, is higher than for road traffic. This would suggest higher noise tariffs for railways 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 a 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 — that is, higher on vehicle level and lower per metric ton for railways 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 5 dB quieter than the average truck, which can be achieved mainly by using low-noise tyres 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.

5.0 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 5 and 6. 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 4 in the previous section as our reference cases, and then varied the parameters within ±50 per cent (−50 per cent corresponds to −3.0 dB; +50 per cent to +1.8 dB). The results in Table 5 are for freight traffic, but the results are close to identical for passenger transport (which therefore are omitted here, but are 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 ±10 per cent in the traffic volume is of

Table 5
Noise Tariffs and Parameter Variability

	Changes as % and dB						
Parameter	*−50%* *−3.0 dB*	*−25%* *−1.2 dB*	*−10%* *−0.5 dB*	*±0* *±0*	*+10%* *+0.4 dB*	*+25%* *+1.0 dB*	*+50%* *+1.8 dB*
Total traffic volume							
Railway	0.978	0.992	0.997	1.000	1.003	1.006	1.011
Road	0.986	0.994	0.998	1.000	1.002	1.004	1.008
Noise level of vehicle							
Railway	0.502	0.751	0.901	1.000	1.099	1.248	1.494
Road	0.500	0.750	0.900	1.000	1.100	1.250	1.500
Noise level of fleet							
Railway	0.493	0.746	0.898	1.000	1.102	1.256	1.512
Road	0.493	0.746	0.898	1.000	1.102	1.256	1.512
Number of exposed							
Railway	0.500	0.750	0.900	1.000	1.100	1.250	1.500
Road	0.500	0.750	0.900	1.000	1.100	1.250	1.500

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

Table 6
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	6.94	0.84
ASEK 4[a] (5 dB rail bonus)	1.00	2.04	0.84

Notes: 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).

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 per cent will only increase the SRMC by about 1 per cent.

The second and 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; that is, it is in line with the results in Table 4 of using 'quiet technology'.[8] Manipulating the noise emissions of the whole fleet is slightly more complex, since this also affects the number of exposed individuals 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.

Not reported in Table 5 but also of importance is the vehicle speed, which has a strong influence on the emitted noise and subsequently the SRMC. Lowering the speed from 110 km/h for light vehicles and 90 km/h for heavy vehicles to 70 km/h for both categories lowers the SRMC by about 65 per cent, and the figures are about the same for changing rail vehicle speeds.

In Table 6 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 4. We start by investigating the effect of including the health component discussed in Section 4.3. The effect on the estimates is substantially higher for the railway estimates; these increase by 87 per cent compared with 11 per cent 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 2. Since most individuals are exposed to low noise levels, this will influence the estimates.

[8]A change of +0.4 dB corresponds to increasing the radiated sound power measured in Watts by 10 per cent (+1.0 dB to 25 per cent and +1.8 dB to 50 per cent).

The second analysis is done by interchanging the monetary values of the two noise sources — that is, road is treated as railway and the other way around. Hence this is a test of the use of benefits transfer. 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 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, 2011). 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 railways 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 close to seven times higher 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 4.

For railway freight traffic noise, another interesting issue is if changing the power plant from diesel to electric might reduce the SRMC. Diesel locomotives typically give higher maximum levels and a somewhat higher contribution to the equivalent level. The effect on the noise level from the freight wagons is independent of the power plant, and for most freight trains the effect of the locomotive on the equivalent level will be negligible, since the number of wagons will be large; in our example there are twenty-five wagons. Since our model estimates the SRMC from the equivalent level, the effect of changing the power plant from diesel to electric is negligible.

6.0 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 standardised 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 standardised 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 the number of those 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.*, 2013). 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 relatively densely populated for Sweden and the SRMC would, therefore, be lower for many other areas in Sweden (Andersson and Ögren, 2011). Previous research has also shown the sensitivity of the chosen threshold level (Andersson and Ögren, 2007, 2011). 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 (using monetary values). 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 — that is, 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; this was reduced to one-third and one-sixth for the truck and freight train, respectively, in our example. In both cases the economic incentive to reduce the noise emission using quiet technologies may seem strong, if it is available within a hypothetical noise-charging system. It will, however, depend on the relationship between the charges and the cost of the technology. For instance, in Wiebe *et al.* (2011) the average cost for retrofitting a freight wagon with K-blocks is stated as EUR 7,000, approximately SEK 63,000. The additional operational cost per wagon is EUR 0.004–0.02 (SEK 0.036–0.18) per km. The train set used as an example in Table 4 has twenty-five wagons, which gives a total retrofit cost of approximately SEK 1.6 million, and an estimated increase in operating costs between 0.90 SEK/km and 4.50 SEK/km. The difference in noise tariff (2.37 SEK/km) is larger than the lower estimate of increased operating costs, but not the higher. Note that this simplified analysis does not include other positive effects of K-blocks such as reduced wear on the rail, and it does not include the SRMC over a larger network, where some parts will have higher SRMC and some parts lower, compared to the case study. For the low-noise truck no data on the associated costs are available in the literature; truck tyres are replaced more often but at lower costs than brake systems on freight wagons. The operating costs of low-noise tyres are likely to be similar or slightly higher compared to standard tyres, since the overall construction and technology is similar. Designing and mounting better sound mufflers will increase the purchase cost of the vehicle slightly.

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. Estimates based on a percentage change in traffic volume would not provide the right incentives (Sansom *et al.*, 2001), but 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.[9] 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 night-time 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, a benefit–cost analysis (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 utilisation (that is, 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 illustrates the fact that the model does not favour different vehicle types. For instance, the higher per-vehicle tariff for a bus compared to a car reflects the fact that the bus emits more noise. The per-unit cost for the bus will be lower than the car, since it transports more passengers.

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 these 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 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 thumb for the number of exposed individuals in different areas.

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

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