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Assessment of health impacts and policy options in relation to transport-related noise exposures
Topic paper noise

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Abstract

This paper has been drafted within the framework of a joint international project (Austria, France, Malta, Sweden, Switzerland and the Netherlands). The aim of this project is to describe the state of the art on transport related health impacts, highlight (if possible) costs and benefits, identify methodological aspects and develop directions for policy measures and strategies, with a special focus on children. A synthesis report of the overall results is available at www.herry.at/the-pep.
This topic paper on transport noise is one of the products of this joint project. It provides an overview of the state of the art regarding traffic noise-related health impacts and guidance for the assessment of noise exposures, its health impacts and costs.

In Europe, transport (road, rail and air traffic) is the most important source of community noise. Noise exposure at community levels can produce various health effects including annoyance and sleep disturbance. A small effect on blood pressure is also deemed plausible. The limited number of epidemiological studies in children indicates that noise exposure affects children's learning (cognition), motivation and annoyance.
The benefits of implementing source-measures for noise abatement may well exceed the costs of these measures, as some cost-benefit analyses clearly indicate. A large variety of policies and measures are available which can reduce the noise-related health impacts. A number or priority options have been identified in joint discussions with researchers and policymakers. The results of noise and HIA studies in different countries are difficult to interpret and compare due to methodological differences. Thus, some methodological and research recommendations to improve health impact assessments and cost-benefit analyses, as well as some examples are presented.

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Executive summary

Background and aim

This paper provides an overview of the state of the art regarding traffic noise-related health impacts and guidance for the assessment of noise exposures, its health impacts and costs. It has been drafted within the framework of a joint international project (Austria, France, Malta, Sweden, Switzerland and the Netherlands). The aim of this project is to describe the state of the art on transport related health impacts, highlight (if possible) costs and benefits, identify methodological aspects and develop directions for policy measures and strategies, with a special focus on children. The overall goal of this joint exercise is to provide a contribution to the implementation of the UNECE –WHO Transport Health and Environment Pan European Program (PEP) as well as input for the Ministerial Conference on Environment and Health in Budapest (June 2004).

Trends in traffic noise exposure levels

Community noise is a widespread environmental problem. In Europe, transport (road, rail and air traffic) is the most important source of community noise. Approximately 30% of the European Union's population (EU15) or close to 120 million people are exposed to levels of road traffic noise of more than 55 dB(A). In general, many people are annoyed and disturbed in their sleep at these levels. Exposure to high noise levels has decreased substantially in some countries since 1980 due to technological (e.g. reduction of emissions, change of road surfaces) and spatial measures such as noise barriers and spatial separation of transport and residential functions (see e.g. section 7, box 1). Nevertheless, noise levels are expected to rise again in the next decades due to the growth in traffic volumes, unless additional measures are taken.

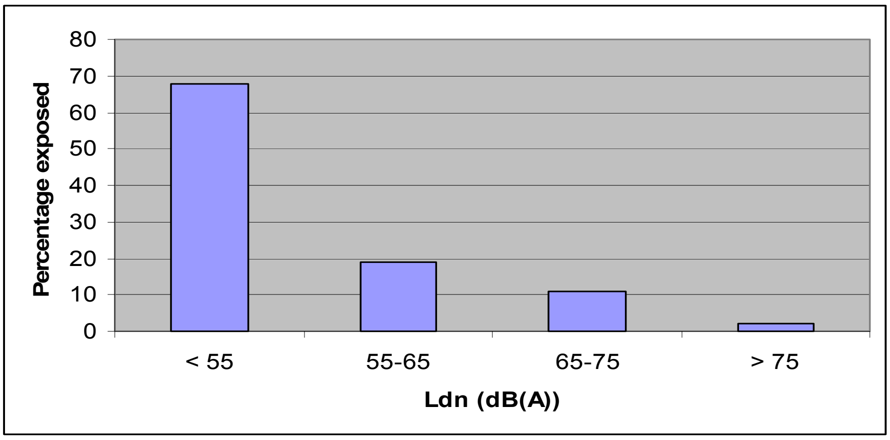


Figure 1 The exposure of European Union's population to traffic noise exposure (façade dwelling) expressed in L_{dn} (Source: Roovers et al., 2000).

Health impacts in adults

A review of epidemiological studies shows that noise exposure at community levels can produce various effects in adults, including annoyance and sleep disturbance. The evidence for a causal association between noise exposure and increased cardiovascular health risk is limited. However, a small effect of noise exposure on blood pressure levels is deemed plausible, since the overall results of these studies on the full range of endpoints from slight elevation of blood pressure to cardiovascular disease are consistent with known stress

reactions and cardiovascular disease progression and supported by laboratory studies on stress reactions and blood dynamics.
Reported annoyance and sleep disturbance levels vary across European countries, but are difficult to interpret due to differences in methodology. Noise exposure explains about 25 - 30% of the observed variance in annoyance. Individual and socio-economic factors are important too (e.g. anxiety, fear, appraisal of a noise source, trust in responsible authorities, perceived control and economical advantages).

Health impacts in children
The limited number of available epidemiological field studies in children shows that noise exposure affects children's learning (cognition), motivation and annoyance. Noise exposure may also have impacts on the cardiovascular and endocrine system. There is no convincing evidence for a direct effect of noise exposure on congenital abnormalities, birth weight or disorders related to the immune system. Current levels of environmental noise exposure in Europe do not have an effect on hearing threshold levels of children. Nevertheless, combined effects should not be excluded: recreational noise (walkmans, discotheques, noisy toys) may make children and adolescents more vulnerable for the effects of traffic or occupational noise.

The overall results from epidemiological field studies on the association between noise and cognition in children indicate that an increase of *aircraft* noise exposure is associated with negative impacts on reading acquisition, up to 6 months impairments in reading age. In general, consistent results are observed for reading ability and memory impairment. Studies looking at the association between noise exposure and attention vary in results. The findings of studies examining the potential impacts of *road* traffic noise are inconsistent.
While some studies indicate that the effects on reading may be reversible if the noise ceases, the long-term developmental consequences of exposure that persists throughout the child's education remain yet to be determined. However, intervention measures (outdoor and indoor) have shown the benefits that might be attained if community noise levels are decreased: reduction of noise levels (indoor and outdoor) has been shown to result in improved long-term memory and reading ability of children.

Children are more vulnerable than adults with regard to effects on learning, while for annoyance the reverse seems more likely. Although children appear to be less disturbed during their sleep than adults, there is some evidence for 'hidden effects' occurring during sleep that, in the long term, might add to the risk of cardiovascular disease. To avoid these effects, protection of children against noise exposure during the night and at school is recommended.

Cost-benefit analysis of transport noise
Noise abatements costs a lot of money but, on the other hand, if abatement fails, noise may have adverse effects on health and well being. These adverse effects can also be expressed in monetary terms. The benefits of implementing several source-measures on cars and trains may exceed the costs of these measures, as some cost-benefit analyses clearly indicate. Low-noise asphalt and reduced speed limits are more cost-effective than erecting noise barriers; instead of just solving local problems they can reduce noise levels throughout a wider urban area.
Equity (or environmental inequity) may also be considered in cost-benefit analyses of policy measures. In the Netherlands for example, low-income groups are more often exposed to high noise levels (> 65 dB(A)) and live less than average in quiet areas (<50 dB(A)).

Health impact assessment and cost-benefit analysis of transport noise: lessons learned
Currently, international comparisons of the noise levels and related impacts in different countries are hampered by (a) possible methodological differences in exposure assessment which may lead to differences in predicted noise levels up to 10 – 15 dB(A) and (b) differences in outcomes of national surveys which can be due to the use of different questions to assess annoyance. The EU Directive 2002/49/EC and ISO technical specification ISO/TS 15666:2003 provide a basis for further harmonisation on noise exposure and annoyance assessment respectively.

For assessment of the potential health risks of transport-related noise exposure, the following options are available:

- Comparison of community noise levels with limit values or policy targets ('distance-to-target'). WHO-guidelines are available which specify noise levels for different settings and activities (see table 3.1).
- Identification of ' hot spots' (areas with high exposure levels) or % population exposed to noise levels above reference or limit values.
- Assessment of the health impacts or number of people affected.

Health Impact Assessment (HIA) can add value to the (transport) policy-making process by helping decision makers identify and assess possible health consequences and optimise overall outcomes of the decision. For more strategic assessments, when different policy options or measures have to be compared, methods are needed to compare the different health impacts of competing decisions (comparative risk analysis) or to compare costs of health impacts with measures, which will reduce these health effects (cost–benefit analysis).

The following approach could be adopted for the health impact assessment of noise:

1. Select health end-points for which there is sufficient evidence (annoyance, sleep disturbance) or limited evidence (hypertension).
2. Assess the exposure distribution of the population using a noise-propagation model or (when not available) a more crude model taking into account traffic and population density. Use the EC-guideline for noise calculations and metrics.
3. Select exposure-response functions based on review of epidemiological studies (see table 7.1 and box). For examining the scientific evidence and selecting valid studies, an advisory report of WHO on evaluating epidemiological evidence provides useful guidance.
4. Calculate the proportion of cases in the study population that can be attributed to noise, based on the basic prevalence in the study population
5. If needed, calculate the total noise-related disease burden or costs.

This approach involves some limitations though. The first is how to deal with uncertainty in causality and exposure-response functions. Do we also include effects for which the evidence is still limited? To overcome this problem one could add a weight factor for the strength of evidence to the calculations. The transferability of estimates to populations other than the study population, from which the estimate has been derived, is another source of uncertainty. Most risk estimates for noise are based on studies in adults. In addition, the prerequisites of certain curves (e.g. some may only be used for strategic, comparative assessments; not for assessment of local and changing situation) should be taken into account. Finally, generalised exposure-response functions for children are lacking, since only a few studies have looked into this. Ongoing projects such as RANCH may provide some useful functions in the near future though.

Exposure-response functions recommended for health impact assessment of traffic noise
Annoyance and (perceived) Sleep disturbance: Use risk estimates from national surveys of good quality. If not available, use functions as described in the EU-guidelines (annoyance, Miedema and Oudshoorn, 2001) or (for sleep disturbance) as described in Miedema et al., 2003. Include correction factors for insulation or window behaviour if needed.
Cardiovascular disease-risk: Use estimates for road traffic and aircraft noise from recent meta-analyis (Van Kempen et al., 2002)
Cognition: To assess the potential impact of aircraft noise on reading and annoyance in children upcoming functions from the RANCH-study may be of use (Stansfeld et al., 2003; Stansfeld et al., submitted).

As an illustration, an assessment of the different health impacts has been made for the Dutch situation (table 1). The best available quantitative estimates for the risks for different cardiovascular endpoints are the results of a meta-analysis published in 2002. These estimates have been used for an indicative assessment, under the assumption that there is a causal association between noise exposure and cardiovascular disease. The results of this assessment of the different health impacts indicate that 1-2 % of the total disease burden might be attributed to health impacts of noise exposure (annoyance, sleep disturbance and cardiovascular disease).

Table 1 Estimates of the number of people affected by road traffic noise exposure in the Netherlands (total population of 16 million) (2000)

Effect	Number of adults affected
Annoyance [a)]	1.5 to 2.2 million
Severe annoyance [a)]	500 to 850 thousand
Sleep disturbance [b)]	550 thousand to 1 million
Severe sleep disturbance [b)]	200 to 450 thousand
Attributable cases of hypertension	Max. 200,000
Deaths attributable to hypertension attributable to noise	Max. 1,100 per year

a) Estimated by means of the (international) exposure-response relationship as derived by Miedema and Oudshoorn, 2001 and only valid for Ldn 45-65 dB(A).
b) Estimates are made on the base of the exposure-response relationship from Miedema et al., 2003 and only valid for Lnight 45-65 dB(A).
c) Estimated by means of a RR of 1.26 per 5 dB(A)) from a meta-analysis on noise and cardiovascular disease (Van Kempen et al., 2002).
d) Deaths attributable to hypertension estimated by means of a Chronic Disease model (Hoogenveen et al., 1998).

Cost-benefit analysis of noise measures has mostly been based on annoyance (amenity loss), assessing the Willingness To Pay for reduction of noise levels. No data are known about the value children put to noise reduction. Just recently, monetary values have been derived within the UNITE project for health impacts (sleep disturbance and cardiovascular impacts) of road noise and aircraft noise, which can be used for calculating external costs of noise. These monetary values derived for health impacts of noise need further validation by health professionals though.

Reference limits for exposure to noise

It is proposed to use national standards (where existing) or WHO guidelines as these specify noise levels for different settings, activities and times. In general, noise levels in residential settings should not exceed 55 dB(A) (Berglund, 1999). A substantial number of people will still be annoyed at this level though. The limit values of WHO for school noise levels (35 dB(A) L_{Aeq}, in school) can be used in setting objectives but may be difficult to reach. The WHO guidelines for night-time noise (45 dB(A) L_{Aeq}) do not allow acting towards reductions of peak levels. Also separate recommendations for aircraft noise should be considered.

Recommendations for further research and exchange of information

It is recommended to have regularly scientific reviews on evidence and consensus building on 'safe' noise threshold levels for different settings, activities and daytimes (update WHO-guidelines), especially with respect to the protection of children. Besides a scientific discussion on the existing WHO guidelines, development of new thresholds based on (new) research should be supported. Results of international studies and good practices should be fed directly in the WHO- and PEP-process and made available to other countries.

To improve the knowledge-base on noise-related health impacts and associated costs in children, the following research options may be considered:

- Study long-term consequences of noise exposure on cognitive development by periodically collecting data on individual performance of children for selected subjects as well as data on the actual noise levels.
- Include other stressors (air pollution) and markers of effect (annoyance, quality of life, behaviour, stress responses) in noise studies. Identify psychological, social and physical protective factors (e.g. restoration). Better information on the context (soundscape) in which adverse effects occur can help architects and land use planners in designing environments which better fit the needs of children (Lercher, 2003).
- Promote intervention studies and identification of best practices of preventing harmful effects of noise in children.
- Assess the health gain of reduction of exposures vs. effectiveness and costs of intervention measures e.g. by using the DALY method. An approach limited to Cost of Illness (COI) is not sufficient since no estimates are available for effects on cognition.
- Support further research on the effects of traffic noise on sleep and cardiovascular risk in children. Evaluate findings from ongoing field studies where the effect of combined exposure of noise and air pollution is studied.
- Support assessments of socio-cultural, economical, and also political factors, which influence annoyance and disturbance responses, in order to feed the decision makers toolbox (e.g. public participation).

Policy options to reduce noise-related health impacts

A large variety of policies and measures are available, which can reduce the impact of traffic noise on health at the local, regional, national and supranational level. A number of priority options have been identified in joint discussions with noise experts and policy-makers (see table 6.1), based on the (magnitude) of expected reductions in noise-related health impacts:

- Development of child-friendly mobility plans, with attention for infrastructure and education measures promoting safe walking and biking by children and their parents.
- Traffic calming measures, such as reduction of speed limits and traffic volume in residential areas.
- Reduction of speed in non-urban road infrastructures, e.g. by promotion of eco-driving and education of driving instructors and drivers.
- Employee travel: charging for parking, incentives for public transport and cycling.
- Integration of land use, transport policy and urban planning. Define objectives for urban and transport planning with regard to e.g. the design of quiet areas, location of schools and dwellings in relation to busy roads, railways and airports.
- Restrictions for heavy lorries, noisy trains and aircraft in/over residential areas (significantly decreasing the number of trucks during the night and weekends and inhibiting/limiting aircraft and train noise at night).

- Regulations for emissions of rail and road vehicles, aircraft; for tyres, road surfaces. Further development and enforcement of (innovative) technological measures reducing emissions at the source and exposures. Enforcement and control of implementation of EU-guidelines.
- Development and monitoring of noise abatement plans.

Since the type and size of transport problems differ per country, region and urban area, different (packages of) measures are needed containing one or more of the above noise-reducing measures. In developing such a package, it is recommended to give priority to those interventions that also address other transport related health effects, since this allows for economic efficiencies and synergies. For example, measures that reduce the volume and speed of traffic around schools and within or around residential areas will reduce noise, air pollution, energy use and improve safety as well.
In the framework of policy development and target setting it is recommended to calculate and compare health impacts and costs of different plans and scenarios. Goals and thresholds for action, monitoring and evaluation of policy/development plans need to be defined.

Contents

1. Introduction

1.1 Background: the PEP

This paper is a contribution to the integration of environmental health aspects into transport policy. It has been drafted within the framework of a joint exercise (Austria, France, Malta, Sweden, Switzerland and the Netherlands) to expand and further develop the methodology for cost-benefit analyses of transport-related health impacts. The aim of the project is to provide an overview of the state of the art on transport related health impacts, highlight (if possible) costs and benefits as well to develop directions for political measures and implementation strategies, with a special focus on children. The overall goal of this joint exercise is to provide a contribution to the implementation of the UNECE –WHO Transport Health and Environment Pan European Program (PEP) as well as input for the Ministerial Conference on Environment and Health in Budapest (June 2004).

As a contribution three background papers on noise have been prepared:
1. Health impact assessment of transport-related noise exposures (adults); state of the art on noise exposure, health impacts and guidelines for health impact assessment
2. Traffic-related noise exposure and health impacts in children
3. Policy options to diminish noise exposures and health impacts related to transport

This final topic paper is an integration of these three papers, combined with the output of discussions during four international expert-review workshops in respectively Vienna, Stockholm, The Hague and Malta. Reports of these meetings and related papers are available at www.herry.at/the-pep. Within the framework of this project it was not possible to collect (new) data for the countries involved. When data were not available or comparable, illustrating data from the Netherlands have been used.

Aim and objectives
This paper provides guidance for assessment of the impacts of traffic noise exposure on public health and related costs. It starts with a brief description of the general approach to assess environment-related health impacts (1.2). This is followed by an overview of the available methodology to assess noise exposure, illustrated with some data from the EU and the Netherlands (section 2). In section 3 an outline of the current state of the art with regard to the epidemiological evidence on noise-related health risks is given, with a special focus on the evidence available for children. In addition, the potential health benefit from reducing noise exposure based on intervention studies is discussed. This is followed (section 4) by an illustration of a health impact assessment (HIA) of transport noise based on data from the Netherlands. In section 5 the methodology and results of cost-benefit-analyses of noise impacts are discussed, followed by an evaluation of policy measures (section 6) and key messages and recommendations (section 7) for future assessments and noise policies.
This paper is based on recent scientific reviews and discussions with (inter) national experts in the field before and during the workshop series.

1.2 Methods for health impact assessment

Health impact assessment (HIA) is a combination of procedures, methods and tools by which a policy, programme or project may be judged as to its potential effects on the health of a population and the distribution of these effects within the population. HIA can add value to the policy development process by (Kemm, 1999; Kemm and Parry, 2004):

- Raising awareness among decision makers of the relationship between health and the physical, social and economic environments (e.g. by quantifying the magnitude of harmful and beneficial impacts);
- By helping decision makers identify and assess possible health consequences and optimise overall outcomes of the decision (e.g. by clarifying the nature of trade-offs in policy making);
- Allowing better mitigation of harmful factors.

The following approach is proposed to assess the health impacts of traffic noise, which is based on the usual procedures for environmental health risk assessment (Hertz-Piccioto, 1998):

(1) Select a set of health endpoints for which there is sufficient evidence for an association with the risk factor under study. For each of these indicators do the following steps:
(2) Exposure assessment: Combine data on (sub) population density with noise level distributions, e.g. by means of Geographical Information Systems (GIS). Noise levels may be based on monitoring data, legally required noise propagation models or simple models taking into account degree of urbanisation, traffic or vehicle density and emission levels;
(3) Identify coefficients and confidence intervals of exposure-response relationships and thresholds of effects: Epidemiological data can serve as the base for an exposure-response assessment if they demonstrate an exposure-response relationship and are of sufficient quality. In examining the scientific evidence the validity of the studies (influence of chance and bias) needs to be evaluated (WHO, 2000).
(4) Estimate the proportion of cases observed in the study population that is attributable to the risk factor under study. This is a function of the population exposure distribution, exposure-response relationships and the observed incidence and prevalence rates of the health end-point in the study population.
(5) Calculate the total noise-related health loss or costs, if desirable.

The impact of traffic-related exposures on human health can take numerous shapes of various severity and clinical significance. Effects of air pollution for example range from increases in respiratory symptoms and hospital admissions to mortality, while noise pollution is associated with annoyance, sleep disturbance and (perhaps) increased incidence or aggravation of cardiovascular diseases. For decision-making and strategic assessments these different health effects have to be compared with other health effects caused by competing decision alternatives (comparative risk assessment) or with costs of measures (costbenefit-analysis). Thus, a common metric that allows aggregating a wide range of health outcomes would enable decisions that are more informed (Hofstetter, 2002; De Hollander et al., 1999). Experience with health metrics in environmental decision-support tools is limited to the use of years of lives lost, monetary cost estimates of health impacts and Quality Adjusted Life Years (QALYs) or Disability Adjusted Life Years (DALYs). All these studies used human health metrics in order to aggregate different health outcomes in one dimension so as to make them more comparable and interpretable (Hofstetter, 2002). In section 4 and 5 some examples of disease burden calculations and cost estimates of noise-related impacts are presented.

2. Noise exposure assessment: methodology and available exposure data EU

This section describes the available methodology to assess and compare traffic-related noise exposure in the EU. Some data from the EU and the Netherlands will be presented.

2.1 Characterisation of noise exposure

The available metrics to characterize noise exposure are based on physical quantities to which 'corrections' are applied that take into account the sensitivity of the human ear. These corrections depend on the frequency, noise characteristics (impulse, intermittent or continuous noise levels), and the source of noise. Within the framework of this paper the following metrics are important:

- ***Sound pressure level***. The sound pressure level (L) is a measure of the air vibrations that make up sound. Because the human ear can detect a wide range of sound pressure levels (from 20 micro-Pascal up to 200 Pascal), they are measured on a logarithmic scale with units of decibels (dB) to indicate the loudness of a sound.
- ***Sound level***. The human ear is not equally sensitive to sounds at different frequencies. To take account of the loudness of a sound a spectral sensitivity factor is used to weigh the sound pressure level at different frequencies (A-filter). These, so called A-weighted sound pressure levels are expressed in dB(A).
- ***Equivalent sound levels***. When sound levels fluctuate in time, the equivalent sound level is determined over a specific period of time. For this purpose the A-weighted sound level is averaged over a period of time (T), using a prescribed procedure (symbol $L_{Aeq,T}$). A common exposure period T in community studies/regulation is from 7 to 23 hours ($L_{Aeq,7\text{-}23hr}$).
- ***Day-night level (L_{dn})***. This metric is used in environmental impact assessment as it correlates much better with community annoyance than the equivalent sound level. L_{dn} is the equivalent sound level over 24 hours, increasing the sound levels during the night (23-07 hours) by 10dB(A) since noise during the evening and the night is perceived as more annoying than during daytime.
- **Day-evening-night level (L_{den})** is constructed in a similar way as the L_{dn}, increasing the sound levels in the evening (19-23 hours) with 5 dB(A) and those during the night (23-07) with 10 dB(A).
- ***L_{night}*.** The equivalent sound level over nighttime (23.00 pm – 07.00 am).
- ***Sound exposure level (SEL)*** of a noise event, such as the noisy passage of an aircraft, is the equivalent sound level during the event normalised to a period of one second.

Usually, the values of these metrics are assessed outdoors.

In most European countries, A-equivalent indices (L_{Aeq}-type) are more common than statistical indices (L_{10} -, L_{50}-type). Unfortunately, noise indices differ per country and within a country even per transport mode. Gottlob (1995) and more recently Flindell et al. (2000) gave good overviews of the indices used for the different modes in European countries. They concluded that especially the indices used to describe noise exposure by aircraft vary considerably between the different countries. Table 2.1 illustrates the findings of Gottlob and Flindell for the Netherlands. Per noise source the indices differ for the periods (day/night or day/evening/night) considered, the maximum or the average exposure and for the long-term time-period (1–3 years) covered. Similar findings are observed in other countries as well.

Table 2.1 Noise indices in use in the Netherlands

Source		**Periods considered and penalties used**	**Night-time**	**Long-term period considered**
Road traffic	L_{Aeq}	Maximum of day and (night + 10 dB(A))	23.00-7.00	1 year, working days only
Railroad traffic	L_{Aeq}	Maximum of day, (evening + 5 dB(A)) and (night + 10 dB(A))	23.00-7.00	3 years
Air traffic	L_{Aeq}	Average of day, (evening + 5 dB(A)) and (night + 10 dB(A))	23.00-7.00	1 year

2.2 Noise calculation methods

Noise exposure is usually assessed according to national noise calculation methods. In the Netherlands, for road traffic RMV2002 is the legally prescribed method. Like all national European noise calculation methods, it first calculates the noise emission by the source, taking into account the characteristics of the source (type of car, speed, type of pavement, height of the source etc.). The next step is the calculation of noise loads at the receiver. To do so, characteristics of the noise propagation path have to be taken into account (e.g. distance, type of ground, presence and type of buildings or other objects etc.). Corrections can be made for the meteorological conditions (temperature, wind). To estimate the number of people exposed, noise propagation models can be combined with demographic data e.g. using GIS (for example figure 2.1)

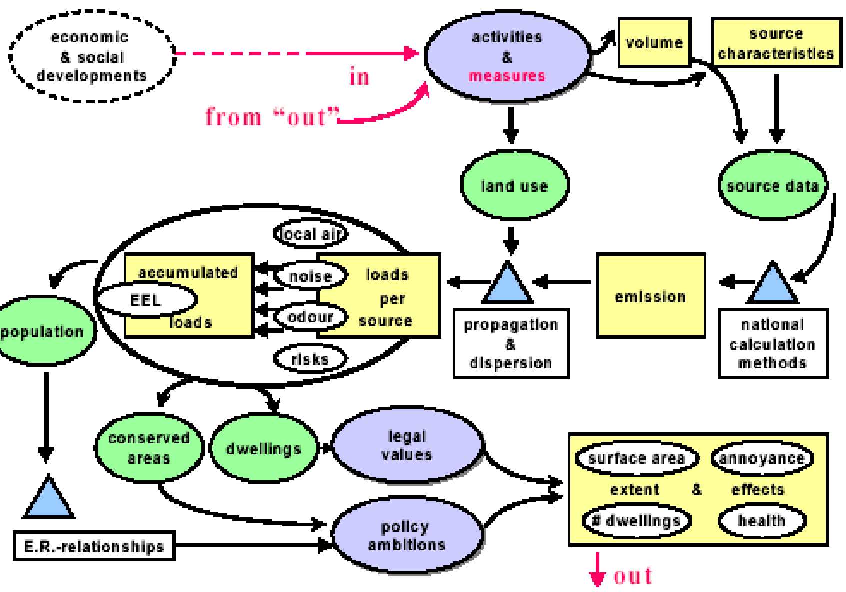


Figure 1: Conceptual design scheme of EMPARA where EEL is Environmental Exposure Level, an index based on a source-dependent weighting of exposure to annoyance

Figure 2.1 Example of Dutch noise calculation model (EMPARA) where EEL is Environmental Exposure Level, an index based on a source-dependent weighing of annoyance caused by the different sources

For regional scale calculations (i.e. whole-city to transnational) it is suggested to use relatively 'simple', spatially and temporally aggregated dispersion/propagation models and link those to routine statistical information on population distribution (from census data), stratified as appropriate by age, gender and socio-economic status. For analysis of urban transport, local scale modelling (i.e. street-level to within-city/district) using high resolution and detailed dispersion/propagation models is recommended.

Different countries have different calculations methods, which, given the same standard situation, usually do not lead to the same outcome. This may partly be explained by different characteristics of driving style, composition of fleet and composition of road/rail in the different countries. Dittrich (2000) measured 2-3 dB(A) difference in noise emission of (the same) train wagons in different countries (Austria, Netherlands, Italy and France), due to the use of different types of railroads.
Apart from these real existing differences, other (and bigger) differences in outcome between national assessments are due to (undesired) methodological artefacts. Pompoli et al. (1995), Van den Berg et al. (1996) and Van Leeuwen et al. (1997) calculated standard road-traffic situations using different national methods and concluded that differences in calculation methods may lead to differences in outcome of 6 – 10 dB(A) (table 2.2). The way the national calculation methods are implemented in mathematical models again may lead to differences of 6 dB(A) (Mank, 2000). Pompoli et al. found that interpretation of the data by different acoustical experts may vary by 1–3 dB(A).

Table 2.2 Nature of differences resulting in different outcome of noise calculations

Difference in	**Results in difference in dB(A)**	**Source**
National noise calculation methods	6 – 10	Pompoli et al., 1995 Van den Berg et al., 1996 Van Leeuwen et al., 1997
Implementation of method	6	Mank, 2000
Interpretation by different experts	1 – 3	Pompoli et al., 1995

The different components in table 2.2 may not be completely independent of each other. Nevertheless, in extreme situations the overall difference between two calculations done by two different experts, using two different national methods, may easily amount up to 10 – 15 dB(A).

In Germany, Austria and Switzerland the so-called ‘Schienenbonus’, a reduction of 5 dB(A) of the outcome of the railroad noise calculation, is applied to account for the fact that, at same noise levels, the noise of trains causes less annoyance than the noise of cars or planes. In the Netherlands it is not accounted for in the noise levels, but in the noise standards, the maximum allowed noise levels. Noise standards are (slightly) higher for noise from trains than for noise from cars. In the Netherlands, however, a ‘temporary’ reduction of the noise immissions by cars of 5 dB(A) on roads with a maximum speed below 70 km/h and 2 dB(A) on other roads is applied (art. 103, Wgh). This reduction is motivated by the expectation that cars will become less noisy in future.

2.3 EU-Noise policy on indices and calculation methods

In 2002, an important step to improve comparability of data and monitoring of noise levels throughout the EU was taken by the European Parliament (EC, 2002). In that year Directive

2002/49/EC was issued relating to the assessment and management of environmental noise. It aims amongst others to harmonize noise indices and noise calculation methods. The A-equivalent indices L_{den} and L_{night} are the harmonized noise indices, to be used throughout the EU-countries for all transport modes.
Member States are obliged to make noise maps for all agglomerations with more than 250 000 inhabitants and for all major roads (>6 million vehicle passages a year), railways (>60 000 train passages per year) and major airports. Furthermore, the directive proclaims the development of a common noise calculation model and designates for the time-being so-called interim methods (the French method for road-traffic noise, the Dutch method for railroad noise and the ECAC.CEAC method for noise of aircrafts). However, as long as national methods ‘do not differ too much from the interim-methods’, the national methods may still be used for noise assessments and for reporting to Brussels. Therefore, in near future, noise exposure assessments will most likely make use of the different national methods. EU-noise policy not only deals with harmonization of indices and calculation methods, but also with the setting (and periodically tightening) of emission limits on tyres, cars and (international) trains. The new EU directives do not provide specific standards, this is up to each member country. One action proposed by the EU is the production of noise maps including the number of people exposed to L_{den} - levels greater or equal to 55 dB(A). At this level, a substantial number of people will still be annoyed.

2.4 Noise exposure

2.4.1 Noise exposure distribution in Europe

In few countries population exposure models have been developed for policy analysis, in which detailed source information at street, grid and city level is combined with data on demographics and dwellings in Geographic Information Systems. However, if such models are not available a more crude approach might be used, based on strong relations between the size of the city and the population exposure distribution. Roovers et al. proposed a simple modelling approach for the European scale that might be used as an example. Residential areas were classified into five categories of noisiness, based on both city size and regional characteristics, ranging from rural to extremely noisy. Regional characteristics involved factors such as latitude (southern cities tend to be noisier), traffic technology (engines, tyres) and densities, urban traffic infrastructure, and meteorological factors (ventilation behaviour) etc. Through analysis of existing data for a series of European cities a crude population exposure distribution for these categories was determined. Successively, these distributions were used to assess the traffic noise exposure for the European population (figure 2.2). Around 13% is exposed to levels above 65 dB(A) ($L_{dn,\ \text{façade dwelling}}$) (EEA, 1999; Roovers et al., 2000). Noise levels distributions in the EU vary widely as shown in figure 2.3. These figures are nevertheless merely indicative, due to the present differences in methodologies in the member states.

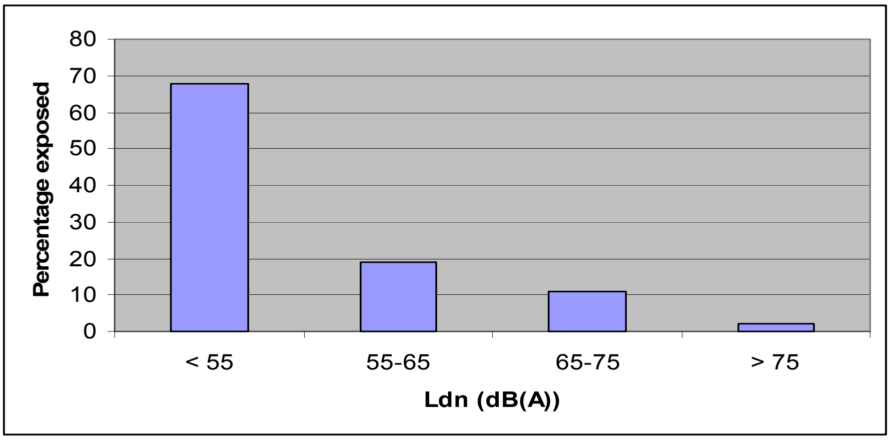


Figure 2.2 Traffic noise exposure distribution in EU (Roovers et al., 2000).

Data on population exposure to aircraft noise in the EU data are scant and unreliable. For a more reliable assessment of population noise exposure distribution on a (supra)national scale, more data have to be collected on crude determinants of exposure, such as city size, traffic technology and density, meteorology, airport size and passenger volumes, urban planning in relation to measured noise levels (EEA, 2000).

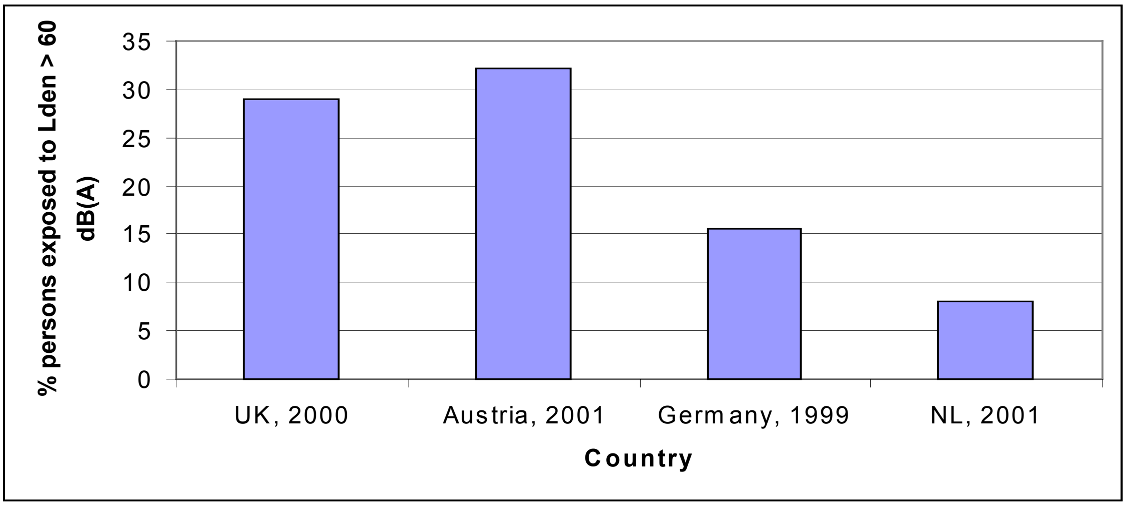


Figure 2.3 The percentage of persons exposed to road traffic noise levels > 60 dB(A) (Lden) in different countries for different years (Sources: United Kingdom, the 1999/2000 UK National Noise Incidence Study; Austria: Federal Environment Agency, 2001; Germany: Umweltbundesambt, 1999; the Netherlands (NL): RIVM, 2003).

Exposure to high noise levels has decreased substantially in some countries due to technological measures (e.g. reduction of emissions, change of road surfaces) and spatial measures such as noise barriers and spatial separation of transport and residential functions. Nevertheless, noise will remain a major problem due to the enormous growth in traffic (especially road and air, e.g. figure 3.4) and the 24-hour economy. The OECD predicts an increase in motor vehicle kilometres of 40% in the next 20 years (OECD, 2001).

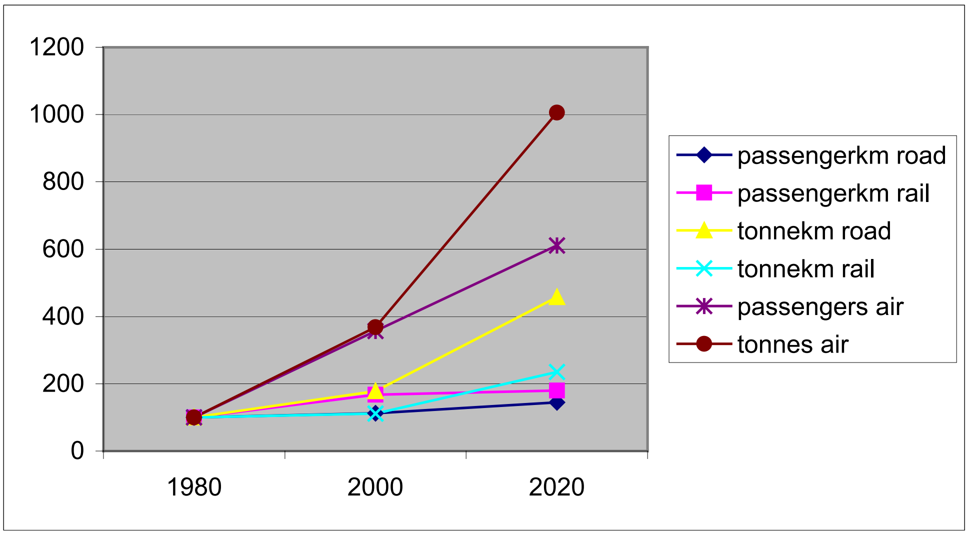


Figure 2.4 Relative growth (1980 =100) of road, rail and air transport in the Netherlands (source: Feiman et al., 2000)

Freight transport is growing faster than passenger transport. Similar developments can be observed in the EU15, the only exception being freight transport by train, which declines in Europe by 0.6 percent per year (EEA, 2000). For Europe as a whole, EEA expects an ongoing shift towards road and air transport. Although the number of vehicles has risen drastically for all modes of transportation, the noise emission per vehicle has decreased during the last decades (this is especially the case for airplanes, less so for passenger cars or freight trains). As a result, noise levels due to air, road and rail traffic in e.g. the Netherlands have (in general) declined slightly in the recent past (RIVM, 2002). Nevertheless, noise levels are expected to rise again in the next decades, mainly due to the ongoing growth of traffic. The same trend is expected for the whole European region.

2.4.2 Noise exposure in the Netherlands

In the Netherlands, as in all European countries, noise from road traffic is the main source of noise exposure. Noise exposure is not equally distributed throughout the country, neither geographically nor socio-economically. The western part of the Netherlands (‘Randstad’) is the most densely populated area, where road, airport and railroad infrastructure is concentrated. As a result, noise levels in the Randstad are higher than elsewhere in the country.

Low-income groups have to deal more than average with high noise exposure (above 65 dB(A)) and live less than average in quiet areas (below 50 dB(A)) (figure 2.5).

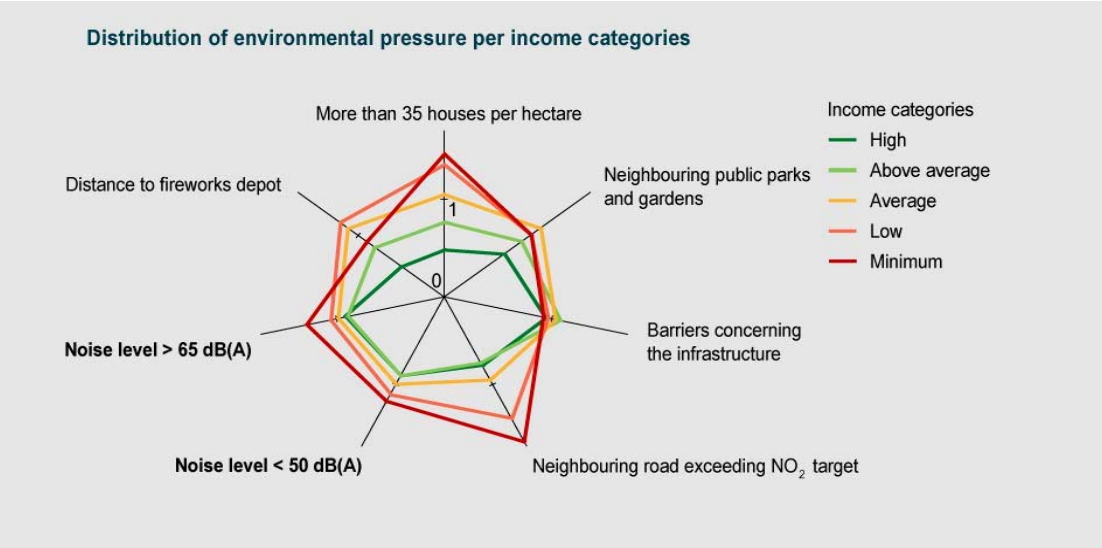


Figure 2.5 Distribution of environmental pressures ('bads') and 'good'(green areas) per income category. Index 1 is the average of the total population, a value smaller than 1 is favourable (source: Bouwman et al., 2001).

2.4.3 Noise exposure of children

Traffic-noise exposure data on children are scarce and limited to a few field studies. In general, the world of the child is becoming noisier. Since the mid-fifties traffic volume has increased, causing higher day and night-time noise levels at home, school and during outdoor activities. Children are exposed to multiple sources of noise: in and outside school, during their recreational activities, when watching television, listening to the radio or loud music, either in a discotheque or when listening to a personal stereo (Francois and Vallet, 2001). In many households, nowadays, noisy toys are abundant and television or audio-equipment is turned on for the whole day (Passchier-Vermeer, 2000). Noise and acoustic problems are the second most common environmental problem at school (Holmberg and Lundquist, 2001). It has been estimated that school-age children may be routinely exposed to more noise than 24hrs equivalent sound level of 70 dB(A). In most European countries, the noise limits for schools are lower than for dwellings though (Gottlob, 1995; table 2.3).

Table 2.3 Recommended indoor noise level (Source: Francois and Vallet, 2001)

Country	Unit	Year of publication	Type of activity: Classroom	Library	Music room	Hall, corridor	Canteen, gymnasium	Room for childr. with hearing difficulties
Germany	L_{Aeq} / L_{Amax}	1987	30-40 / 40-50	30-40/ 40-50				
Belgium	L_{Aeq}	1977/87	30-45[a)]		30-40		35-50	
France	L_{Aeq}	1995	38	33			43	
Italy	L_{Amax}	1975	36			40	40	
Portugal	L_{Aeq}	2000	35				40-45	
Former Czechoslovakia	L_{Aeq}	1977	45				55-60	
UK	$L_{Aeq,\,1h}$	1997	40	40	30	50		30[b)]
Sweden	L_{eq}	1995	30	35			40	
Turkey	L_{eq}	1986	45				60	

a) Maximum sound levels depend on the outdoor noise level for the zone. There are four categories of zone.
b) Maximum levels of background noise in all classrooms for children with hearing difficulties should be at least 10 dB lower than these values.

3. Health effects of noise

In this section, the current state of the art with regard to the evidence on noise-related health risks is given, with a special focus on children. The strengths and weaknesses of previous studies are described as well as ongoing studies, which may provide new data and knowledge to be used in future assessments. In 3.1 an overview is given of the evidence on noise-related health risks in adults and children. There is sufficient evidence that noise causes annoyance and sleep disturbance in adults and has impacts on children's learning. In addition, there is limited evidence for an association between noise exposure and cardiovascular impacts. These effects are described in more detail in the subsequent paragraphs. The strength of the available evidence, the influence of modifying or confounding factors, and (if available) exposure-response functions which can be used in future assessments are described.

3.1 Overview of reported noise effects on health and well-being

Long-term exposure to noise has been associated with a wide range of effects on human health and well being. We can distinguish between social psychological responses, such as annoyance and sleep disturbance and disturbance of daily activities on the one hand, and physiological effects on the other, such as hearing impairment, hypertension and aggravation of cardiovascular symptoms. In addition, in children noise-related impacts on cognitive functioning have been observed.
Stress may play an important role in the aggravation of health impacts. One of the many models available and depicting the possible mechanisms of noise-induced health effects and their interactions is presented in figure 3.1.

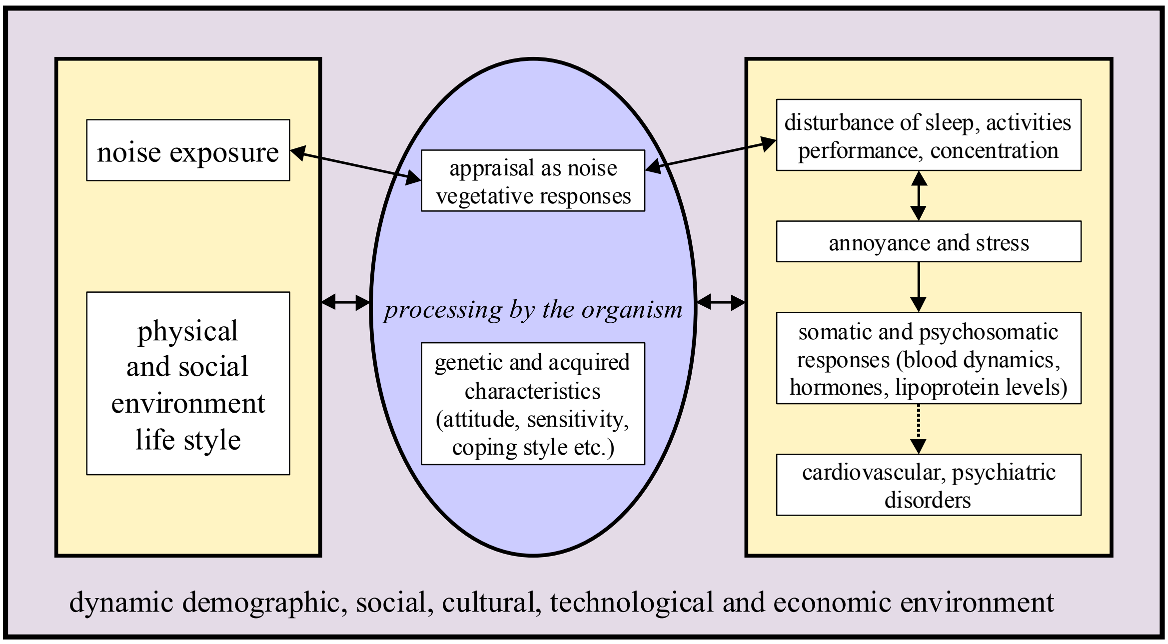


Figure 3.1 Conceptual model representing relation between noise exposure, health and quality of life (Source: HCN, 1999)

It shows that noise may directly or indirectly (via disturbed activities) influence vegetative, hormone, or emotional regulation mechanisms. Various factors can modify the way in which the individual processes the noise signal. Examples of such factors are the attitude towards noise and noise sensitivity. The model in figure 3.1 is based on what is found in adults. About the underlying mechanisms of noise affecting children less is known. It is suggested that children might be more sensitive to noise than adults because of noise exposure during critical developmental periods (organ development of foetuses, babies, learning of children). In addition, they may have less possibilities of controlling the noise or a less developed coping repertoire as compared with adults (Stansfeld, 2003; Bistrup, 2003).

3.1.1 Evidence for noise-related health risks

Table 3.1 gives an overview of the effects in both adults and children that have been reported in relation with noise exposure, rating the strength of evidence in terms of the categories proposed by the International Agency on the Research on Cancer IARC (sufficient, limited, inadequate and lacking) and the available guideline values. This overview is based on a large number of previous major scientific reviews (e.g. Health Council of the Netherlands (HCN), 1999; Passchier-Vermeer, 2000; Porter et al., 1998; Stansfeld et al., 2000) and WHO-guidelines (Berglund et al., 1999; WHO, 2002).

Adults
From these substantial reviews, it can be concluded that there is sufficient evidence for an association between noise-exposure and health risks such as hearing damage (although unlikely to occur at typical levels of community noise exposure), annoyance and sleep disturbance. The evidence for cardiovascular risk (e.g. increased rates of hypertension or ischaemic heart diseases) is more limited. Due to inconsistent results and inherent methodological shortcomings, no definite conclusion with respect to the causal role of environmental noise in mental health impacts can be drawn. A significant association between self-reported noise exposure and depression as well as cognitive failures was observed (Smith, 2001). Nevertheless, a number of studies applying psychometric questionnaires to assess psychological morbidity yielded inconsistent results (Stansfeld, 2000).

Children
There is sufficient evidence that noise exposure affects children's learning (cognition), motivation and annoyance. It may also have impacts on the cardiovascular and endocrine system (table 3.1). Studies investigating impacts on mental health in children yielded inconsistent results. The possible effects of noise on cognitive functioning were studied the most frequently. There is no convincing evidence for a direct effect of noise exposure on congenital abnormalities, birth weight or disorders related to the immune system. The information available indicates that exposure to high *occupational* noise levels of the pregnant mother is associated with hearing impairment and growth retardation of the child (Passchier-Vermeer, 2000). A few studies show an increased risk of low birth-weight in children of aircraft-noise exposed mothers, but the influence of important confounders such as SES and smoking was not taking into account (Ando, 1973; Knipschild, 1981; Matsui, 2003).

The use of noisy toys as well as very loud noise during concerts, in discotheques and through headphones can be at the origin of development of tinnitus or hearing impairment among young people. It is estimated that 20% of young people across Europe are overexposed to loud music and at immediate risk of hearing loss (WHO, 2003). Current levels of environmental noise exposure in Europe do not have an effect on hearing threshold levels of children. The effects of recreational noise (walkmans, discotheques), although an important contributor to current prevalence of hearing loss in adolescents, are not further discussed here since the scope of this paper is transport-related noise exposure. Nevertheless, combined effects should not be excluded: recreational noise may make children and adolescents more vulnerable for the effects of traffic or occupational noise.

An overview of the noise-related health risks for which there is limited to sufficient evidence (annoyance, sleep disturbance, cardiovascular disease, cognition) will be presented in the following paragraphs. This overview is based on a number of recent reviews by national and international advisory committees and/or established scientists. The effects investigated in children are discussed in more detail, based on the results of recent field studies (see table 3.2), results will be explained in 3.2-5.

Table 3.1 Overview of reported responses to environmental noise exposure in children and adults and the available WHO guideline values (Source: Berglund, 1999; Babisch 2001; HCN, 1999; Hygge 2003; Van Kempen et al., 2002; Miedema, 2001; Passchier-Vermeer, 2000, 2003; Stansfeld, 2003).

Effect/response	**Population**	**Strength of evidence**[a)]	**Specific environment(s)** [b)]	**Noise source** [c)]	**Guideline value**		
					L_{Aeq} [dB]	**Time base (hrs)**	**$L_{Amax,}$ [dB]**
Annoyance	Adults	Sufficient	Outdoor living area, daytime and evening	Environ	55 [d)]	16	-
	Children	Sufficient	School, playground, outdoor		55	during play	
Psychosocial well being	Adults	Limited		Environ			
Well being/perceived stress	Children	Sufficient/Limited	School		35 [e)]		
Psychiatric disorders	Adults	Limited		Environ			
	Children	Inconclusive		Environ			
Performance	Adults	Limited		Environ			
Cog. Performance:							
Reading	Children	Sufficient	School class rooms and pre-schools indoors	Air	35 [e)]		
Memory	Children	Sufficient	School class rooms and pre-schools indoors	Air	35 [e)]		
Auditory discrimination	Children	Sufficient	School class rooms and pre-schools indoors		35 [e)]		
Speech perception/intelligibility	Children	Sufficient	School class rooms and pre-schools indoors		35 [e)]	During class	
Academic performance	Children	Sufficient	School class rooms and pre-schools indoors		35 [e)]		
Attention	Children	Inconclusive	School class rooms and pre-schools indoors		35 [e)]		
Motivation	Children	Sufficient/limited	School	Air	35 [e)]		
Catecholamine secretion	Children	Limited/inconclusive		Road, Air	35		
Biochemical effects	Adults	Limited		Environ			
Cardiovascular system							
Blood pressure changes	Adults	Limited		Environ, Occup			
Blood pressure changes	Children	Limited		Road, Air			
Hypertension							
Use of anti-hypertensives	Adults	Limited [f)]		Road, Air			
Medical consult	Adults	Limited [f)]		Road, Air			
Angina Pectoris	Adults	Limited [f)]		Road, Air			
Myocardial infarction	Adults	Limited [f)]		Road, Air			
Ischeamic Heart disease (total)	Adults	Limited/sufficient [f)]		Road, Air			

Effect/response	**Population**	**Strength of evidence**[a)]	**Specific environment(s)** [b)]	**Noise source** [c)]	**Guideline value**		
					L_{Aeq} [dB]	**Time base (hrs)**	**$L_{Amax,}$ [dB]**
Effects on sleep [g)]							
Changes in EEG parameters	Adults	Sufficient	Sleep				
Awakenings	Adults	Sufficient	Sleep				
(onset of) motility	Adults	Sufficient	Sleep	Air			
Subjective sleep quality	Adults	Sufficient	Sleep				
Heart rate	Adults	Sufficient	Sleep				
Mood next day	Adults	Sufficient	Sleep				
Hormones	Adults	Limited	Sleep				
Immune System	Adults	Inadequate	Sleep				
Performance next day	Adults	Limited	Sleep				
Sleep disturbance	Children	Inconclusive	Inside bedroom		30	8	45
Sleep disturbance, window open	Adults/childr		Outside bedrooms		45	8	60
Sleep disturbance, inside	en				30	8	60
Hearing impairment [g)]		Sufficient	Industrial, commercial, shopping and traffic areas, indoors and outdoors		70	24	110
			Public addresses, indoors and addresses		85	1	110
			Ceremonies, festivals and entertainment events (< 5 times /year)		100	4	110
			Music through headphones/earphones		85	1	110
			Impulse sounds from toys, fireworks and fire arms				120-140
Birth weight	Children	Inadequate		Environ, air			
Immune effects	Children	Inadequate		Environ			

a) Classification of evidence of causal relationship between noise exposure and health endpoint;
b) Sleep = during sleeping time, school = exposure of children at school;
c) Environ = environmental exposure, road = road traffic noise, air=aircraft noise;
d) This is the guideline value for serious annoyance;
e) To prevent disturbance of information extraction;
f) Based on the results of Van Kempen et al., 2002;
g) For sleep disturbance and hearing impairment, several WHO guideline values are available for specific environments.

Table 3.2 Summary of health outcomes observed in recent field studies in children (from: Matheson, 2003, Van Kempen, 2003), references in table (+ = positive association observed, 0 = no association observed, - = negative association, NI = not investigated)

Study[a)]	Design	# schools	# children (age)	Exposure		
				Source	Metric	Range
LA-study	Cross-sectional /1-yr follow-up	7	262	Air	Peak sound level	High: 95 dB
Munich	Nat. experiment	-	326 (9-10yr)	Air	$L_{Aeq,\ 24\ hrs}$	Gr 1 68/54 Gr 2 59/55 Gr 3 53/62 Gr 4 53/55[b)]
SEHS	Cross-sectional /1-yr follow-up	8	340 (8-11 yr)	Air	$L_{Aeq,16hr}$	High: > 66 dB Low: < 57 dB
WLSS	Cross-sectional	20	451 (8-9 yr)	Air	$L_{Aeq,16hr}$	High: > 63 dB Low: < 57 dB
Tyrol	Cross-sectional	26	1230 (8-11yr)	Rail, Road	L_{dn}	High > 60 dB Low < 50 dB
RANCH	Cross-sectional	89	2844 (9-10 yr)	Air, Road	$L_{Aeq,7\text{-}23hr}$	Air: 30-77 dB Road: 32-71 dB

Study → Outcome ↓	LA-study	Munich	SEHS	WLSS	RANCH	Tyrol
Summary of health/quality of life outcomes						
Annoyance	NI	+	+	+	+	+
Quality of life	NI	-	NI	NI	+	-
Motivation and helplessness	+	+	0	NI	NI	NI
Stress Hormones	NI	+	0	0	NI	+/0
Blood pressure	+	+	NI	NI	+/0	+
Summary of cognitive outcomes						
Reading	0	+	+/0	+/0	+	NI
LT-memory	NI	+	+/0	0	+	NI
Working memory	NI	+	NI	0	+	NI
Attention	+/-	+	+	0	-	0
Mental Health/behaviour	NI	+	0	+	-	NI

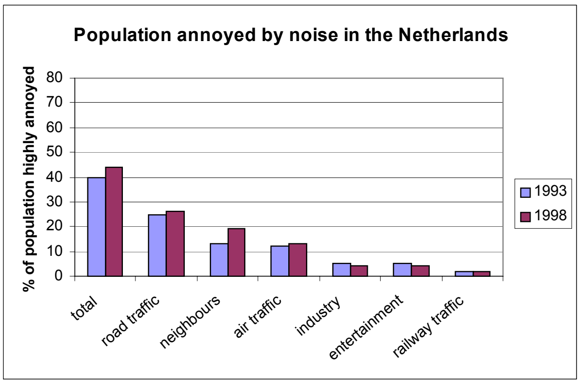
a) LA-study: Los Angeles Airport study (Cohen et al., 1980; Cohen et al., 1981). Munich: The Munich Airport Study (Evans et al., 1995; Evans et al., 1998; Hygge et al., 2002). SEHS: Schools Environment and Health Study: Haines et al., 2001ab); WLSS: The West London Schools Study: Haines et al.) Tyrol: The Tyrol Study: Lercher et al., 2002; RANCH: Stansfeld 2003, www.RANCHproject.org
b) Gr 1: noise levels of group old airport-aircraft noise before/after airport switch; Gr 2 noise levels of group old airport-no aircraft noise; Gr 3 noise levels of group new airport-aircraft noise; Gr 4 noise levels of new airport-no aircraft noise.

3.2 Annoyance

Annoyance can be defined as 'A feeling of displeasure associated with any agent or condition, known or believed by an individual or group to adversely affect them' or 'a feeling of resentment, displeasure, discomfort, dissatisfaction or offence which occurs when noise interferes with someone's thoughts, feelings or daily activities'. The degree of annoyance caused by noise exposure depends on several characteristics, such as sound level, spectral characteristics and varies with time of the day or season. During the night and late evening noise is more annoying because quietness is expected. Based on the results of surveys it has been observed that noise exposure explains about 25-30% of the observed variance in annoyance. Non-acoustical factors also play a major role (Job, 1999; Stallen, 1999; Guski, 1999). Examples of non-acoustical factors are individual noise sensitivity, fear with respect to the source, attitude towards the source, perceived control over the situation, perceived economic or societal advantages of noise generating activity. Several reviews show that anxiety (fear of the noise source) and noise sensitivity are the most important non-acoustical factors of influence on exposure-response relationships (Fields, 1993; Guski, 1999; Job, 1999; Stallen, 1999; Miedema and Vos, 1999). According to Guski factors with a social character (appraisal of a noise source, trust (in those responsible for noise and noise abatement), history of noise exposure) are important because they apply for whole groups of the population and can be used to reduce noise annoyance.

3.2.1 Adults

Noise annoyance is always assessed on the level of populations using questionnaires. Recently, efforts have been made by the International Commission on Biological Effects of Noise (ICBEN) and the International Organization of Standardization (ISO) towards the use of standardized questions asking for the degree of annoyance in a 0-10 or 100 scale. To determine the percentage of people annoyed and highly annoyed, a cut-off value of 50 and 72 is being used. Road traffic noise, neighbour noise and aircraft noise usually are the most common sources of annoyance in Europe (figure 3.2).



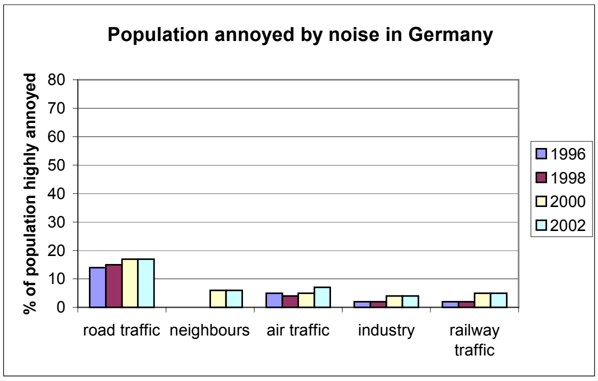


Figure 3.2 Percentage of the population highly annoyed by noise in the Netherlands and Germany (Source: RIVM, 2002; UBA, 2003). Comparable questions were used.

3.2.2 Children

Only a few field studies are known in which residential noise annoyance of children is measured in a systematic and quantitative manner (table 3.2). Most studies focus on aircraft noise. No source-specific exposure-response curves are available as yet. According to Lercher (2003) this is due to a lack of standard methodology to measure annoyance in children, the use of different noise indices as well as insufficient representative data to draw a generalised curve from.

In the Munich studies (table 3.2) an increase in annoyance in children living near the new airport was observed during the measurement period (three waves) while the mean annoyance in the children living near to the closed airport dropped from a high to a low score at wave three. A child-adapted questionnaire with 21 Likert-scaled items was used, covering different degrees of noise perception, air quality and residential qualities (green space, playgrounds etc) (Bullinger, 1998/99).
In an experimental study within the Tyrol studies (table 3.2) children were asked to assess the annoyance of road and railway noise sounds presented via headphones by using a Visual

Analogous Scale. Children from the quiet area (n = 63, L_{Eq8hr} < 40 dB(A)) had consistently higher annoyance scores for both highway and railway noise than children from the noise-exposed group ($L_{Eq,8\text{-}night}$ > 50 dB(A)). For both groups an increase in annoyance ratings with (laboratory) noise levels was observed. Rail noise was rated more annoying at 60 dB(A) and 70 dB(A), but equally annoying as motorway noise at 50 and 80 dB(A) (Sukowski, 2000). A survey among 530 13-15 year old children in Germany (using the same questionnaire as in the Munich study) also showed that children report lower mean annoyance levels than their mothers. The highest mean annoyance ratings were observed in the aircraft exposed rural areas while road traffic noise annoyance ratings equalled those of air pollution or odour annoyance (Bullinger et al., 1997).
Focus group discussions in a small international (n=36) sample indicate that the interviewed children were most affected by neighbours noise and road traffic noise (Millenium Conference Study, Haines 2003). This is comparable with the results of community surveys in adults. The children rated water pollution as the most damaging source of pollution, followed by air and lastly noise: *'It depends where you are though. Long term it's water pollution and air pollution, but walking down the street it's noise pollution that affects you more.'*
Analysis of a small sample (n=18) of the West London School Study showed that the impact of noise exposure on everyday activities (eg schoolwork, homework, playing) was larger for the children exposed to high levels of aircraft noise (Leq 16 hr > 63 dB(A)) compared with the low noise exposed children (<57 dB(A)) and focus group samples. The sample of children exposed to aircraft noise expressed high annoyance levels, with responses consistent with those in adults (irritation, fear, anger). In both studies when asked, children selected their bedrooms and green areas in their neighbourhoods as places to find some respite from noise pollution (Haines, 2003). The sample sizes of both studies are too small though to derive a more definite conclusion on coping strategies in children.
Preliminary findings of the three field surveys in the RANCH study show positive associations between aircraft and road traffic noise exposure and annoyance reported by the children (Van Kamp, 2003). Annoyance was measured using a for adults standardised general purpose noise reaction question with a 5-point verbal scale ranging from not at all annoyed to extremely much. The use of these questions enables a comparison with previous studies as well as with parents and teachers ratings. For both parents and teachers steeper exposure-response curves were observed than for children (Van Kamp et al, 2003). Analysis of a small subsample of the RANCH-study shows that children with high psychological restoration scores have low annoyance scores, suggesting that psychological restoration may protect against annoyance. 'Psychological restoration' is children's capability to appreciate restorative environments by creating feelings of pleasantness and tranquillity (Gunnarsson et al., 2003). Also, children seem to have different ways than adults to avoid the noise e.g. children more frequently raised the volume of radio/walkman (Öhrström et al., 2003).

All four field studies showed that children are annoyed by long-term noise exposure. The emotional response of children to noise exposure seems to be consistent with adult reactions.

3.2.3 Available exposure-response relationships for annoyance

Based on a pooled analysis of original datasets from noise-annoyance surveys carried out in Europe, Australia, Japan and North-America, exposure response relationships have been derived for road, rail and air traffic noise (Miedema and Oudshoorn, 2001). These can be used to predict the number of (highly) annoyed people in an exposed population (eg. figure 3.3, table 7.1). Based on this pooled analysis, aircraft noise appears to be the most annoying noise source, followed by highway traffic noise, traffic noise from other roads and railway noise. The curves derived by Miedema and Oudshoorn are recommended for use in the EU

Directive on Noise (EU position paper, 2002). Although these curves have been derived from probably the most elaborate database currently available, methodological differences in the original studies (e.g. poor exposure assessment, differences in adequacy noise insulation) may have influenced the observed relationships (Finegold, 2002; Fidell, 2003). Some of the surveys included in the analysis are rather outdated. According to TNO, recent analyses (not published yet) do not reveal any systematic changes of the exposure-response relationships over the time span covered by the data sets used.

Figure 2: The percentage highly annoyed persons (%HA) as a function of the noise exposure of the dwelling (Lden). The solid lines are the estimated curves, and the dashed lines are the polynomial approximations. The figure also shows the 95% confidence intervals (dotted lines).

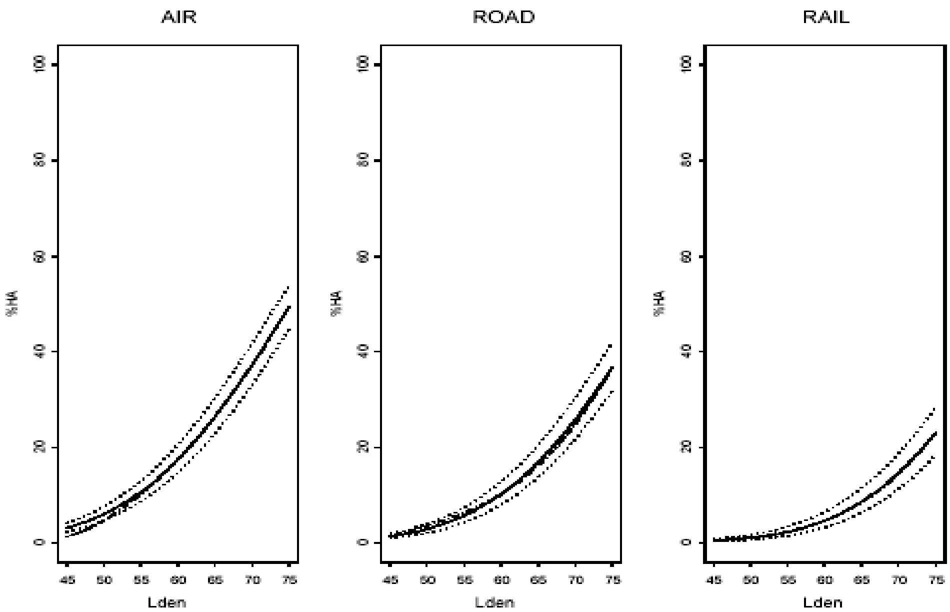


Figure 3.3 The percentage highly annoyed as a function of the noise exposure of the dwelling (L_{den})(Miedema an Oudshoorn, 2001).

Applications and limitations
According to the position paper the exposure-response functions are only to be used for aircraft, road traffic, and railway noise and for assessment of long-term stable situations (EU, 2002). They can be utilised for strategic assessments, in order to estimate the effects of noise on populations in terms of annoyance. They are not applicable to local, complaint-type situations, or to the assessment of the short-term effects of a change of noise climate. The curves have been derived for *adults*. The curves are not recommended for specific sources such as helicopters, military low-flying aircraft, train shunting noise, shipping noise or aircraft noise on the ground [taxi-ing] (EU, 2002).

Re-analysis of the existing database may be considered. An important elaboration of the relationships would be the inclusion of more (exposure) variables as predictors of annoyance (in addition to L_{den}), e.g. sound insulation or the presence of a relatively quiet side of the dwelling (Miedema and Oudshoorn, 2001). There is also a need for quantifying the influence of noise sensitivity on annoyance. Since actual annoyance levels differ between Northern and Southern EU member states (due to e.g. window behaviour, cultural differences), the transferability of these curves for general use remains a question.

Cumulative effects
Miedema has developed a method to calculate the cumulative noise levels by taking into account the differences in annoyance-response levels to various noise sources, expressed as *Environmental Exposure Level (EEL;* Miedema, 1998*)*. This method is being applied in Dutch noise calculations, but internationally still under debate. The EEL is based on the functions described in figure 3.3. Given the estimated exposure level, the annoyance is estimated for each source. Next, on the basis of the function for road traffic noise and annoyance, it is derived what level of road traffic noise causes the same level of annoyance. Thus, the Environmental Exposure level or index is defined as the 24-hour value for road traffic noise causing the same degree of annoyance as the noise levels of the source under study. The index may not be suitable for situations under change or with L_{Aeq} levels of more than 75 dB(A).
There are some studies available which show *combined impacts of traffic-related noise and air pollution on annoyance.* The higher the road traffic noise levels people are exposed to, the more likely they are to be annoyed by exhaust smell (and vice versa) (Klaeboe, 2000). Based on these results Klaeboe recommends to integrate air-pollution modelling into a broader framework, in order to correctly assess the effect or traffic measures. This means the development of modelling tools such as structure equation models, simultaneously modelling the effect of each environmental exposure on their respective annoyance responses (and their interactions).

3.3 Sleep disturbance

Night-time noise affects the sleep quality and the mood and performance the next day. Sleep disturbance may manifest itself in various ways (see table 3.1). Noise may affect sleeping behaviour (e.g. increasing the time awake during the night), sleep-pattern (as measured by Electroencephalogram, EEG), physiological responses or it may cause chronic changes. The following effects can be distinguished:

- Primary effects like difficulties falling asleep, awakenings, sleep stage changes and instantaneous arousal effects during the sleep (temporary increase in blood pressure, heart rate, vasoconstriction, release of stress hormones in the blood, increased motility);
- Secundary or 'after effects' measured the next day: decrease of perceived sleep quality, increased fatigue and decrease in mood and performance;
- Long-term effects on well being: increased medication use or chronic annoyance.

Thus, sleep disturbance by night-time noise is measured by various indicators such as sleep-pattern, (self-perceived) sleep quality, attention tests (performance) or mood-questionnaires the next day. Motility as measured by wrist-actimetry is an indication for the number of awakenings during the night.

3.3.1 Adults

Noise increases the changes between sleep stages (W, 1, 2, 3, 4, REM) and the number of awakenings during the night, starting from SEL levels of about 35 and 60 dB(A), respectively. Reported sleep quality is likely to be affected at night-time noise levels above 40 dB(A). In most studies an effect of night-time noise on performance and mood the next day is only seen at levels above 60 dB(A). Night-time noise exposure may increase heart rate during the night; habituation to this effect does not seem to occur. The observation threshold is a SEL value of 40 dB(A). Age, sex, season, medical condition and medication are important factors of influence with regard to the level of sleep disturbance. There are some indications that noise-induced sleep stage changes are associated with elevated (stress) hormone levels (Passchier-Vermeer and Passchier, 2000; Maschke, 2003). It is not known whether these more or less instantaneous effects contribute to chronic changes or long-term health effects. Recovery mechanisms can prevent the occurrence of further effects.

3.3.2 Children

Only a few studies investigating the effect of noise on sleep EEG, awakenings and perceived sleep quality in children are available. Most studies in children are limited to (pre-term) children exposed to high noise levels in incubators or hospital wards (Kahn, 2003). The few studies in healthy children involved a very small number of children. Changes in sleep quality and quantity are seen when a child is exposed to noise during sleep. Young children are less prone to awakenings due to aircraft noise than adults (Lukas, 1972). An increase in body movements and awakenings (but no changes in EEG) and time falling asleep was observed in children from a quiet area (n = 8) when exposed to increasing sound levels during several nights (Eberhardt, 1990). After a noise-reduction measure (reducing the noise level in the bedroom by 11 dB(A)) Eberhardt observed a reduction in time falling asleep and a very small increase in REM sleep in children (n = 5) who lived along streets with night traffic. It is assumed that brain restoration occurs mainly during REM sleep. Eberhardt estimates that the same sleep EEG reactions occur in adults and children if the night-time exposure of children is 10 dB(A) higher than the exposure of the adults. During the last third of the night, in which REM sleep is predominant, children under experimental conditions show more noise-induced EEG awakenings than during the beginning of the night (Passchier-Vermeer, 2000).
A study by Muzet et al comparing traffic noise-induced sleep disturbance and cardiovascular responses in three age groups showed the highest cardiovascular response in children (6-12) as compared with young adults and elderly people (Muzet et al., 1980).
Although children appear to be less disturbed during their sleep than adults (with respect to awakenings and sleep quality) there is evidence for ‘hidden effects’ occurring during sleep (e.g. cardiovascular and hormonal responses). These effects do not seem to diminish (adaptation) and in the long term might cumulate, adding to the risk for e.g. cardiovascular diseases or hypertension.
The preliminary results from the RANCH study in Sweden show that children seem to have better perceived sleep quality than adults. Children scored better than adults on some sleep indicators (sleep quality, tiredness) but not on others (sleep latency, wake episodes). Sleep impairment in children seems to start at higher noise levels than in adults (Öhrström et al., 2003).

3.3.3 Available exposure-response relationships for sleep disturbance

On the basis of meta-analyses several researchers have proposed exposure-response relationships. The results of these reviews have been criticised (Berglund et al., 1999; Miedema et al., 2003; Muzet, 2003): Only relationships for effects like awakenings and sleep stage changes have been proposed, while other effects like (perceived) sleep quality or performance have not been included. In most analyses and original studies, indoor noise exposure has not been measured but been estimated and factors such as noise insulation and ventilation behaviour have not been taken into account. In addition, exposure-effect relationships derived from laboratory and field studies are very different (Franssen and Kwekkeboom, 2003). For this reason, only synthesis curves based on field studies are recommended for use in health impacts assessments.

Based on an analysis of original data from 15 datasets (12 field studies, 12000 observations) in the TNO archive, relationships have been proposed (table 7.1) that give the percentage highly sleep disturbed (*%HSD*), sleep disturbed (*%SD*), and (at least) a little sleep disturbed (*%LSD*) by road traffic and railway noise as a function of the outdoor L_{night} at the most exposed façade (Miedema et al., 2003). Sleep disturbance questions vary a lot between surveys, in wording and in the number or response categories. In order to obtain comparable disturbance measures the sets in the selected studies were translated into a scale from 0 to 100. Cut-off points to assess the percentage of highly sleep disturbed persons were used analogue to the definitions of percentage (highly) annoyed persons. No relationships for aircraft noise were proposed because of the large variance in results.

Relationships for awakenings and instantaneous and mean motility have also been tentatively proposed (Miedema et al., 2003). Instantaneous motility measured by actimetry correlates well with EEG- and behavioural awakenings. In a recent extensive study around Schiphol Airport mean motility during sleep has been associated with number of sleep and health complaints and self-reported sleep quality (Passchier-Vermeer et al., 2002). Since this study has sufficient power and several short-comings of earlier studies have been accounted for (e.g. control for outcome dependency due to repeated measurements, indoor noise measurements, data on important mediating or confounding factors) Passchier-Vermeer proposes to use the exposure-effect relationships for instantaneous and mean motility derived from this aircraft noise study. An important factor influencing this relationship is the individual long-term aircraft noise exposure during sleep. With higher aircraft noise exposure (L_{night} 40 dB(A)) the probability of instantaneous aircraft noise-related increase in motility is much lower. Mean motility during sleep is strongly related to age and is also a function of noise exposure during the sleeping period.

Applications and limitations
The curves described in table 7.1 have been derived for adults. They describe the level of annoyance due to night-time noise, which is not the same as perceived sleep quality. The curve for aircraft noise is based on only one (but extensive) field study. Further verification of the relationships proposed is needed with attention to construction of the dwellings (insulation, position of the bedroom) and other use of windows. In conclusion, these curves may not be generally applicable and should be used with great care.

3.4 Cognition

3.4.1 Overview of studies in children

The following results have been found in children exposed to high levels of noise (aircraft, train and road), as compared to children in quieter schools: (a) deficits in sustained attention and visual attention; (b) difficulties in concentration; (c) poorer auditory discrimination and speech perception; (d) memory impairment for tasks that require high processing demands; and (e) poorer reading ability and school performance on national standardised tests (Stansfeld et al., 2000) (Stansfeld and Haines, 2002). Table 3.2 gives an overview of the most recent studies investigating the effects of noise exposure on children's cognition. For reading ability consistent results are observed, indicating a negative association between chronic (long-term) noise exposure and reading acquisition. Studies looking at the association between noise exposure and attention deficits vary in results. Nearly all studies have involved a cross-sectional design, small samples sizes, and lack of adjustments for potential confounders such as socio-economic status. Only a few studies have examined exposure-response relationships. Studies with an intervention design are discussed in section 3.4.2.

A most interesting longitudinal study (table 3.2) examined the effects of changes in aircraft noise exposure on the health and performance of both children attending schools near the old Munich airport which was closed, and children living near the new airport. The noise levels near the old airport declined from 68 to 54 dB(A) after it closed. Deficits in long-term memory and reading comprehension were observed in children living near the old airport as compared with children living in a quiet area. These impairments diminished within 2 years after the airport was closed. The same cognitive skills were adversely affected in children living close to the new airport within 2 years after the opening (Evans 1995, 1998; Hygge et al., 2002). Reading comprehension and sustained attention of children in UK attending high noise schools was poorer compared with children of low aircraft noise exposed schools (Haines et al., 2001; table 3.2).

Ongoing studies

The EU funded RANCH study - Road Traffic and Aircraft Noise Exposure and Children's Cognition and Health- examines the relationship between chronic exposure to aircraft or road traffic noise and impaired cognitive function, health and noise annoyance in children. It is comprised of 3 field studies of 9-10 year-old children living around major airports in the UK, Spain and the Netherlands. In addition, the project includes a field study in Sweden focusing on the impact of road traffic noise on well-being and sleep in young children and adults by measuring perceived sleep quality and awakenings (motility, using wrist-actimeters). Studies of soundscapes are carried out embedded within the airport study in the UK and road traffic noise studies in Sweden. The field studies in the UK, the Netherlands and Spain have the same design, allowing comparisons of the data collected at these three sites.

In the three field studies, the cognitive performance and health of 2844 children aged 9-10 years visiting 89 primary schools exposed to different levels of road traffic and air traffic noise (expressed as LAeq7-23) were compared cross-sectionally. For this purpose, cognitive tests, questionnaires (for both children, parents and teachers) and a blood pressure protocol were developed. The cognitive outcomes included reading comprehension, episodic memory, working memory, prospective memory and sustained attention. In the Netherlands, additional computer-tests were administered measuring switching attention, memory, motor and perceptual skills. Health outcomes included annoyance, blood pressure, mental health and self-reported health. The schools were selected according to the noise exposure of the school area – based on model calculations or on-site measurements- and such that children were matched on socio-economic state and ethnicity. The data was pooled across the three centres and analysed using multilevel modelling, adjusting for confounding factors at the school and the individual level. Until now, the findings show that aircraft noise exposure at school is associated with reading, episodic memory and working memory. Aircraft noise exposure at school is not associated with impairment of either prospective memory or sustained attention. Road traffic noise exposure is not associated with either reading comprehension, episodic memory, working memory, prospective memory or sustained attention. In the Dutch sample, aircraft and road traffic noise effects were observed in the more complex attention tasks. The observed differences in responses to road- and aircraft noise may be due to difficulties in estimating road traffic noise levels at schools. Exposure-response functions will be developed which can be used for health impact assessments. A problem is that the long-term consequences of the cognitive effects found are difficult to interpret. Final results are expected in 2004. For more information on the RANCH-study see Stansfeld et al., 2003 and www.ranchproject.org.

Motivation
The Los Angeles and Munich Study investigated the effects on children's motivation (see table 3.2). Motivation is seen as an indicator for the child's vulnerability to learned helplessness. Motivational effects were often investigated by examining whether success or failure on a puzzle-solving task would affect the child's performance on a second puzzle-solving task (Matheson et al, 2003). Children highly exposed to chronic environmental noise seem to be less motivated when placed in situations were task performance is dependent on persistence. Associations were found between noise exposure and reduced persistence on challenging puzzles. Teachers in high noise areas report greater difficulties in motivating children as compared to their colleagues in low noise areas (Stansfeld et al., 2000).

3.4.2 Results of noise-intervention-studies in children
The National Institute of Public Health in Denmark and several consultants from other countries collected and evaluated examples of noise prevention or reduction in children's daily settings by carrying out a literature review and interviews of key-experts (Bistrup 2002, 2003). In this review a few successful interventions in the area of transport noise are being described.
In New York City railway noise intervention measures (rubber pads on tracks, sound-absorbing ceilings in school) reduced the noise levels in classrooms by 6-8 dB(A), resulting in improved reading ability of the children in classrooms facing the railway tracks (Bronzaft, 1981). The Los Angeles Study (table 3.2) showed a reduction of noise levels by 7 dB(A) in noise-abated classrooms with some small improvements on cognitive performance and motivation but not on reading scores (Cohen et al., 1980). Closure of the old Munich Airport (table 3.2) resulted in reduction of noise levels from 68-54 dB(A) and an improvement in long-term memory recall and reading while the reverse effect occurred in children living near the new airport. Acoustic treatment of classrooms reducing the background noise by 5-7 dB(A) resulted in improved speech and word intelligibility in school children (Mackenzie and Airey, 1999) and better cognitive performance in children of preschool age (Maxwell and Evans 1998, 2000). Experiments in the Munich and Tyrol studies showed that children chronically exposed to high noise levels were less affected by acute noise at testing than control children (Meis et al., 2000).

3.4.3 Available exposure-response relationships for cognition
At the moment for children no generalised exposure-response relationships are available which can be used for further health impact assessments, except for reading for which the coefficients from the West London studies and/or the RANCH-study can be used.
Until now, for reading three attempts have been made for deriving a exposure-effect relation. Green et al, investigated the effect of air traffic noise exposure on reading, expressed as the percentage of students reading below grade level. To this end data for the years 1972 - 1976 were used. The results suggested that a one unit increase in noise score would be accompanied by an increase of 0.62% in the number of students reading one or more years below grade level in an average school. Later, Haines et al carried out a similar study investigating the effects of air craft noise exposure on national standardised scores (SATS) in English, spelling, handwriting, creative writing, reading, mathematics and science from 11000 children from 123 schools. Chronic exposure to aircraft noise exposure was significantly related to poorer reading and mathematics performance. However, after adjustment for SES, these associations were no longer statistically significant. Recently, the RANCH-study found that air traffic noise exposure was associated in a linear exposure-effect association with reading comprehension. It was estimated that a 5 dB(A) increase in noise

was associated with a 2-month impairment in reading age in the UK and a 1-month impairment in the Netherlands.
In the framework of a WHO-project to derive exposure-response relationships and indicators for noise impact assessment, Hygge tentatively has developed some hypothetical exposure-response curves for recall and reading in children conversing individual performance scores from cross-sectional studies to cumulative curves. He warns for several drawbacks though. The effect of changing noise levels may not be accurately predicted and conversing exposure periods from the different studies into one metric may be debatable. The analysis provides some insight though in the slope of the curves for different noise levels and outcomes, and could be validated by new empirical studies (Hygge, 2003).

3.5 Cardiovascular diseases

3.5.1 Overview of epidemiological studies in adults

Noise exposure is associated with blood pressure changes and ischemic heart disease risk but epidemiological evidence is still limited according to various recent reviews (HCN, 1993; Morrell et al., 1997; Porter et al., 1998; Babisch, 1998; Van Kempen, 2002). The literature suggests that noise-induced cardiovascular effects can be seen as the consequence of stress. Small, transient stress-related hemodynamic responses that are harmless on an individual level may result in slight shifts in blood pressure at population level. In a smaller, susceptible proportion of the population this may lead to increase in hypertension and, eventually, in the prevalence of IHD, including angina pectoris and myocardial infarction. Recently a meta-analysis has been performed of 43 occupational and environmental epidemiological studies on noise exposure and cardiovascular endpoints, including blood pressure, hypertension, medical consultations, use of cardiovascular medicines, angina pectoris, myocardial infarction and prevalence of ischemic heart disease (Van Kempen et al., 2002). The analysis revealed inconsistent, sometimes even contradictory results of individual studies, and summary relative risks were only significant in a few cases. A significant association for occupational noise and air traffic noise exposure and hypertension was observed, but not for road traffic noise. These results are in agreement with an earlier review by Babisch, who concluded that there was little epidemiological evidence of an increased risk of hypertension in subjects exposed to road traffic noise (Babisch, 1998). In cross-sectional studies, road traffic noise exposure is associated with an increased risk of myocardial infarction and ischemic heart diseases. In follow-up studies, the findings for ischemic diseases (IHD-total) were not confirmed (Babisch, 1993, 1998). The meta-analysis also showed that air traffic noise exposure was positively associated with medical consultation, use of cardiovascular medicines and angina pectoris (table 3.3).
In the period 2000-2004 some new community noise studies investigating the effects of road traffic, air traffic and rail traffic noise on cardiovascular disease have been published. The conclusions from these studies do not really differ from what is already found in the published reviews on this topic. New is that the effect of night-time noise exposure was investigated and that the effects of air pollution were also taken into account.
The results of the Spandauer Gesundheits Survey showed that night-time noise exposure was stronger associated with medical treatment for hypertension than day-time noise exposure (Maschke, 2003 -adults). Time-series analysis of hospital admission data in Madrid in the period 1995-1997 showed a clear association between emergency admissions for all and specific (circulatory, respiratory) causes and environmental noise levels (61-72 dB(A)). Other explanatory factors such as air pollution levels were controlled for in the models. About 5% of all emergency admissions could be attributed to high noise levels (Tobias et al., 2001).

Table 3.3 Summary estimates, expressed as $RR_{5\ dB(A)}$, for the association between noise exposure, hypertension, and ischemic heart diseases, adjusted for sex and age (Source: Van Kempen et al., 2002).

Noise exposure[a]	Outcome	$RR_{5\ dB(A)}$	95% CI[d]	# estimates	Measurement range (dB(A))
Occupation	Hypertension[b]	1.14	1.01 – 1.29 *	9	55 – 116
Road traffic	Hypertension	0.95	0.84 – 1.08	2	<55 – 80
	Use of antihypertensives	0.96	0.76 – 1.22	2	> 50 – 73
	Consultation of GP/specialist	0.91	0.73 – 1.12	1	55 – 70
	Angina Pectoris	0.99	0.84 – 1.16	2	51 – 70
	Myocardial Infarction[c]	1.03	0.99 – 1.09	3	51 – 80
	IHD-total[c]	1.09	1.05 – 1.13 *	2	51 – 70
Air traffic	Hypertension	1.26	1.14 – 1.39 *	1	55 – 72
	Use of antihypertensives	0.99	0.87 – 1.14	1	55 – 72
	Consultation of GP/specialist	1.10	0.95 – 1.27	2	55 – 77
	Use of cardiovascular drugs	1.05	0.99 – 1.11	2	38 – 77
	Angina Pectoris	1.03	0.90 – 1.18	1	55 – 72

[a] The noise exposure measures differed between the noise exposure sources: occupational noise exposure expressed in $L_{Aeq, 8h}$,in dB(A), road traffic noise exposure expressed in $L_{Aeq, 6\text{-}22h}$,in dB(A) and air traffic noise exposure expressed in $L_{Aeq, 7\text{-}19h}$,in dB(A). [b] Adjusted for age, sex and type of work. [c] Only prevalence estimates. [d] CI = Confidence Interval * Significant, $p<0.05$. RR= Relative Risk exposed vs non-exposed

3.5.2 Children

Only nine studies investigated the effects of air traffic, road traffic and rail traffic noise on blood pressure in children aged 3-16 years (see for more details annex 1). Seven of these studies were cross-sectional in design; one was a follow-up-study (Cohen, 1981), the Munich study was an intervention study (Evans et al, 1998). Six studies found blood pressure elevations associated with noise exposure. In three studies a statistically significant noise-related increase in blood pressure has been observed, but differences in ethnicity or social class could have confounded the results. In the Los Angeles study an association between aircraft noise exposure and an increase in systolic and diastolic blood pressure was observed (Cohen, 1981). In the Munich airport study, a marginally significant increase in systolic blood pressure in high aircraft-noise exposed children was observed as compared with children from the control group. Preliminary results of the RANCH-study are inconclusive (Van Kempen et al., 2003).

Effects of road (Ising, 2002), rail and road (Evans et al., 2001) and air traffic noise exposure (Evans et al., 1998) on stress responses, e.g. cortisol (measured in overnight urine and salivary cortisol), adrenaline and noradrenaline have been studied. The results of these studies were inconclusive: if any associations were observed, the effects were only small. Stress hormone levels (epinephrine) were higher in children exposed to aircraft noise at the old Munich airport. After the move of the airport the levels of epinephrine rose among children living under the flight paths of the new airport (Hygge, 1996; Evans et al., 1998). Children exposed to road and rail noise levels of more than 60 dB(A) had raised urinary cortisol levels but no difference in urinary (nor)adrenaline (Evans et al., 2001) as compared to lower-exposed children (L_{dn} 50 dB(A)). Results were adjusted for socio-economic status, hearing, family size etc. Ising observed raised cortisol levels in children exposed to road traffic noise (indoor levels Lmax 33-52 dB(A)) during the first but not the second half of the night (Ising, 2002).

3.5.3 Available exposure-response functions for cardiovascular risk

Limitations and applications
It should be noted that the estimates in table 3.3 have been based on cross-sectional studies, which are hampered by poor (retrospective) exposure assessment. In most studies there were limited possibilities of controlling confounding variables (diet, smoking, BMI) and selection bias (Self-selection: healthy neighbour effect, sensitive subjects may have moved out of the polluted areas thus diluting the effect of interest). In most reviews, the evidence for a causal relation between noise exposure and cardiovascular health risk is considered to be limited (a.o. Babisch, 2001). However, a small effect on cardiovascular risk is deemed plausible, since the overall results on the full range of endpoints from slight elevation of blood pressure to Angina pectoris are consistent with known cardiovascular disease progression and supported by laboratory studies on stress reactions and blood dynamics. Nevertheless, well-designed cohort studies with good exposure characterization will be needed to confirm these suggestions.
In conclusion, some risk estimates for road traffic and aircraft noise are available for adults. The summary estimates in table 3.3 for hypertension (aircraft noise) and IHD (road traffic noise) could be indicatively used for further health impact estimations. The thresholds of no-effect (or reference level) are still debatable though. No estimates are available for railway noise.

Ongoing studies
The overall goal of the EU-sponsored project Hypertension and Exposure to Noise near Airports (HYENA, 2003-2006) is to examine the impact of long-term noise generated by aircraft and road traffic near airports on cardiovascular outcomes reflected by high blood pressure. The study is carried out in the UK, Sweden, Germany, Greece, Italy and the Netherlands. An additional goal is to evaluate the modifying effects of traffic-related air pollution on noise-associated cardiovascular risk factors and disease. At some sites the effect of traffic noise exposure on stress hormone levels will be studied too.

4. Health Impact Assessment of traffic noise

In this section an example is presented of a health impact assessment of (road traffic) noise. The numbers presented are based on calculations for the Dutch situation and serve only as an illustration. Guidelines for the calculation of noise-related health impacts such as annoyance, sleep disturbance and cardiovascular impacts are being developed in the framework of a WHO-environmental health indicator project (WHO, 2004).
Assessments can be made both for the separate health outcomes and for the total health loss or disease burden, expressed in *Disability-Adjusted life years* (DALYs). Calculation of the total noise-related disease burden enables a more comparative analysis of the environmental health impacts associated with transport (see also section 2). DALYs are a member of the family of aggregated health indicators as developed in the framework of the World Bank/WHO Global Burden of Disease project. The method tries to aggregate three important dimensions of public health: (i) the loss of life expectancy, (ii) the loss of quality of life, and (iii) the number of people affected. The unit of measurement is time (disability adjusted life years).
To assess the health impact of noise, information on population exposure distribution, exposure-response relationships, and incidence and prevalence rates is combined to estimate the annual numbers of people affected. To calculate the disease burden, this number is than multiplied by the severity (ranging from 0 to 1) and the duration (in years) of the condition. In formula:

DALY = Number affected x Severity x Duration

4.1 Health Impact Assessment

Selection of health endpoints
Given the current quantitative insights in health impacts of noise exposure, quantitative assessments of noise related health impacts or loss could be based on severe annoyance prevalence and sleep disturbances (both as proxies for decreased quality of life). In addition, we can indicatively assess the occurrence of noise-induced hypertension (population blood pressure distribution) as risk factor for cardiovascular disease, although the evidence for this health outcome is still limited. The coefficients and confidence intervals of the corresponding exposure-response functions to be used are described in section 3 and table 7.1 respectively.

Exposure assessment
To assess the noise exposure distribution of the Dutch population, a GIS-based noise-propagation model combining source information with population and built environment data was used (EMPARA, see section 2). According to these model calculations about 40% of the Dutch population is exposed to road traffic noise exposure levels of more than 55 dB(A) (figure 4.1).

Estimation of number of people affected by annoyance and sleep disturbance
Quantitative exposure response functions are available to predict traffic noise-related annoyance levels in steady state situations (Miedema and Oudshoorn, 2001). In addition, generalised exposure response functions for perceived sleep disturbance are available (Miedema et al., 2003). There is no consensus yet on exposure-response functions for other

indicators of noise-disturbed sleep. The number of people severely annoyed by road traffic noise was estimated by combining the noise exposure distribution as shown in figure 4.1 with the exposure-response function (ERF) derived by Miedema (2001). Noise levels were recalculated into L_{den} , the metric the selected ERF is based on. It is estimated that in total 1.8 million people are annoyed by road traffic noise and about 600,000 people severely annoyed (table 4.1).

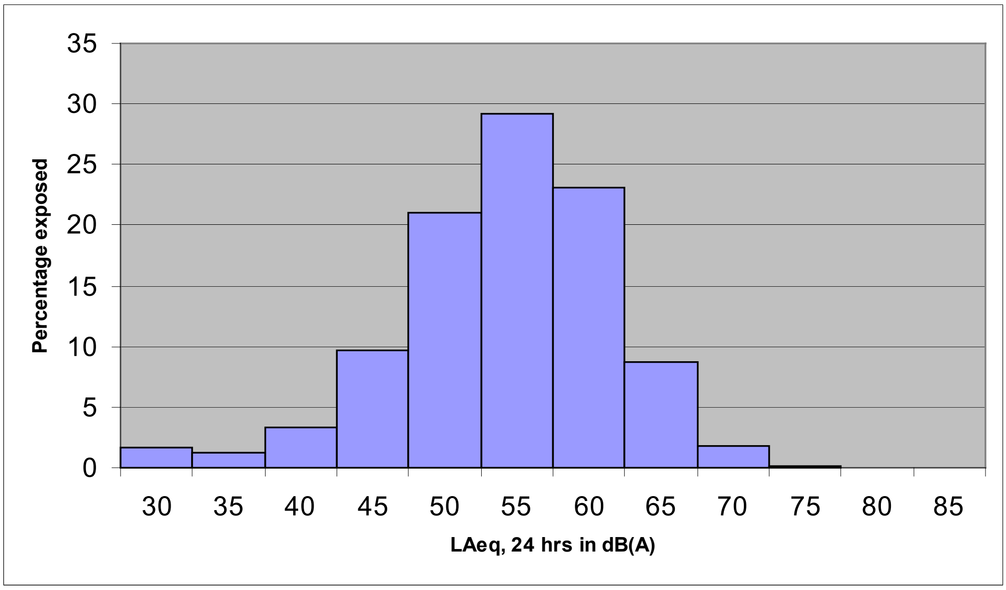


Figure 4.1 Exposure of the Dutch population to noise of road traffic in 2000, expressed in equivalent noise levels for 24 hours (L_{Aeq})(Source: RIVM, 2004). For clarification of noise metric, see table 2.1.

Table 4.1 The percentage of people exposed to and severely annoyed by road traffic noise in the Netherlands (adults only, total population 16 million).

Exposure category, Lden (dB(A))	**Average Lden (dB(A))**	**% of population exposed**	**% severely annoyed**	**Number per 1,000,000**
<40	40	7.5	0	0
41-45	43	11.8	0.5	588
46-50	48	23.1	2.7	6,224
51-55	53	29.4	5.4	15,880
56-60	58	20.2	8.8	17,777
61-65	63	6.7	13.8	9,195
66-70	68	1.2	21.3	2,654
>71	73	0.1	31.8	433
Total		**100**		**52,751**

The fraction of people experiencing severe sleep disturbance can be assessed in a similar way. It was estimated that 200,000 – 450,000 people may experience severe sleep disturbance due to road traffic noise (figure 4.2).

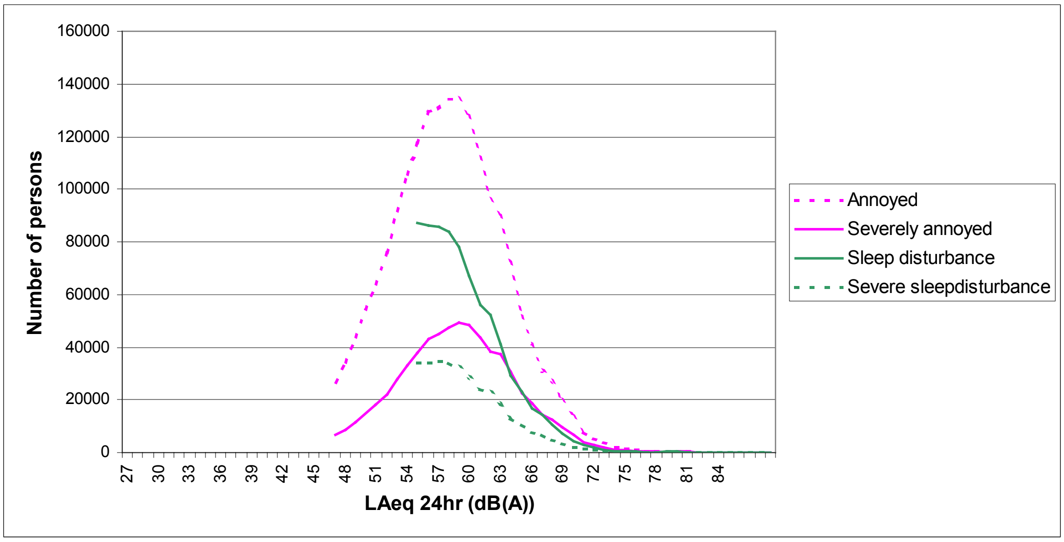


Figure 4.2 The estimated number of people that are (severely) annoyed or experience (severe) sleep disturbance due to road traffic noise exposure in the Netherlands in 2000, per dB(A).

Mortality due to noise attributive hypertension

Noise exposure may have a small effect on cardiovascular disease progression. Looking at studies investigating the effects on the cardiovascular system a range of endpoints that is consistent with known cardiovascular disease progression can be observed (Van Kempen et al., 2002). Following the reasoning of Dutch Health Council model (see section 3), we assume that the risk elevations associated with noise exposure for the several endpoints are an indication of a small contribution to total disease prevalence. From this worst-case perspective we calculated the annual hypertension mortality that may be attributed to noise exposure (population attributable risk or PAR). We made no distinction between aircraft, road or rail traffic noise, although these different noise types may be processed in different ways. The calculation consisted of three steps. First, we calculated PARs by combining the exposure population distribution (figure 4.1) with quantitative exposure-response information, applying equations 1 and 2:

1) $$RR_i = e^{\frac{(L_i - L_{cut-off})}{5} * \beta}$$

2) $$PAR_{hypertension/noise} = \frac{\sum_{i>0}(RR_i - 1) * p_i}{\sum_{i\geq 0} RR_i * p_i}$$

3) $$PAR_{mortality/noise} = PAR_{mortality/hypertension} x PAR_{hypertension/noise}$$

in which:

PAR = Population Attributable Risk
RR_i = relative risk in exposure class i,
L_i = exposure level in class i, expressed in dB(A),
$L_{cut\text{-}off}$ = cut-off or reference level,
β = the risk function estimate (per 5 dB(A))
p_i = exposure probability in class i.

The exposure response function used was derived from a meta-analysis on noise and cardiovascular disease (see section 3). A β of 0.23 was chosen (95% CI 0.13 - 0.33) or a relative risk per 5 dB(A) of 1.26 (95% CI 1.14 - 1.39). This coefficient is derived from an aircraft noise study, assuming a similar magnitude of risk for road traffic noise. In addition, it is assumed that the relation between noise exposure and the prevalence of hypertension is exponential, with a reference value of 50 dB(A). The population attributable risk for noise-induced hypertension was 0.06. This means that a maximum of 200,000 cases of hypertension could be attributable to road traffic noise exposure. Since most people suffering from hypertension do not experience problems in their daily functioning, this health state is normally not incorporated in the calculation of the burden of disease. Therefore we have also estimated the fraction of noise-related mortality attributable to hypertension (0.0043) by multiplying the PAR for noise-induced hypertension (0.06) with the population attributable risk for hypertension-induced mortality (0.073) (equation 3). By multiplying this PAR with annual mortality data, obtained from Dutch health statistics we estimated that maximum 1,100 people may die annually due to noise attributive hypertension.

4.2 Noise-related disease burden

4.2.1 Assessment of noise-related disease burden in the Netherlands

To estimate the number of disability adjusted life years that is lost per year of exposure, we combined the predicted number of people affected (see 4.1) with weight factors for the severity and duration of the condition, using the following equation:

$$DALY_{e.e.} = \sum_{i=1}^{n} \sum_{k} I_k * f_k(RR_i, C_i) * S_k * D_k, \text{ in which}$$

$DALY_{e.e.}$ = health loss related to n environmental exposures, measured as disability or quality adjusted life-years per year of exposure.
$f_k(RR_i. C_i)$ = a set of functions (including exposure C_i and associated relative risk measures RR_i) representing the population attributable fraction (PAF) of condition k.
I_k = annual incidence of response k, S_k = severity factor discounting time spent with the condition, D_k = duration of the condition; in case of premature mortality: loss of life expectancy.

In 4.1 the calculation of the individual health outcomes has been described

Severity weights

In the determination of the severity of health statuses all sorts of values and judgements play a role. To attribute weight to environmental health impacts we used the experiences from a Dutch Burden of Disease Project. In the Dutch project 52 diagnoses of public health significance were given weights by a panel of physicians, using two different valuation instruments. A standardised classification of the health states according to EuroQuol was provided to assist the panel members. This instrument to measure quality of life involves a three-point scale for six health dimensions: mobility, self-care, daily activities, pain/discomfort, anxiety/depression, and cognitive function. In order to determine a severity factor for severe annoyance and severe sleep disturbance, we made use of the results of another Dutch study in which a group of environmental physicians, epidemiologists and public health professionals was asked to evaluate and weigh a number of environment-related health effects on a Visual Analogue Scale ranging from 0 (healthy) to 1 (death), using the scale of calibration states which was drawn up earlier (Van Kempen, 1998).

Duration

The average duration of a health response, or the loss of life expectancy as a consequence of premature mortality, was estimated using mortality and morbidity statistics and relevant literature. For mortality due to noise attributive hypertension we used the average years of life lost for ischemic heart diseases. For noise annoyance and sleep disturbance we used annual prevalence rates (based on periodic surveys), assuming that people will be annoyed or sleep disturbed throughout the year. Therefore, the duration of these conditions is defined as 1 year in the DALY calculations.

Table 4.2 provides an overview of the noise related disease burden expressed in DALYs. The noise attributable loss of 'disability' adjusted life years is potentially largest for the social psychological endpoints 'severe annoyance' and 'sleep disturbance'.

Table 4.2 Estimation of the noise attributable disease burden, expressed in Disability Adjusted Life Years (DALYs), per million inhabitants.

Effect	**Number affected**	**Severity**	**Duration**	**DALYs per million inhabitants**
Severe annoyance	500 to 850 thousand	0,02 *	1 year (prevalence rates)	400 to 2700
Severe sleep disturbance	200 to 450 thousand	0,02 *	1 year (prevalence rates)	150 to 1300
Mortality due to noise attributive hypertension	Max. 1100 per year**	1	10,5 years*	Max. 700
Total				Max 4700

* modus from distribution
** indicative value, worst case calculation: RR from aircraft noise study was used, under the assumption of a similar association for road traffic noise

4.2.2 Advantages and limitations

Uncertainty analysis using Monte Carlo techniques shows that altering severity factors does not substantially affect the overall picture as compared with the uncertainties in some dose-response estimates. The lower the disability weights attributed to health states get though, the more sensitive they are to variation. As the less severe responses tend to affect the highest number of people, variation in these severity weights has a large impact on the estimates of disease burden. Estimation of the duration of the pollution-related condition is another source of uncertainty.

Looking at the response definitions the question is whether we include both clinically measurable and more social responses such as annoyance and sleep disturbance in the calculations. The application of severity weights, although formally derived in a relatively sophisticated way, introduces a subjective aspect into the model, which is sometimes disputed. These severity weights only seem to be critical with respect to mild responses with a substantial prevalence (such as annoyance). Nevertheless, health preference measurements (to derive severity weights) seem to be rather stable, even across countries.

Although uncertainty of the various variables in the calculations can make results difficult to interpret, the quantification of the disease burden can be very useful for policy making. It enables comparative risk evaluation and evaluation of the efficiency of environmental policies (in terms of health gain).
As an example of comparative risk evaluation, figure 4.3 shows transport related DALYs for the Netherlands for the period 1980 – 2020. These are preliminary estimates, but show the potential. The estimation of transport-related disease burden allows the evaluation of possible trade-offs among specific policies, e.g. air pollution versus accidents. In order to this, better estimations of the contribution of road traffic emissions to air pollution levels are needed

though. An additional positive aspect of integrated health impact assessments like this is the possibility of not overlooking the ‘side effects’ resulting from a policy addressed to a single issue, allowing to maintain insight on what would happen not only in terms of e.g. emission reductions but also in terms of road safety or noise. For planning and policy-evaluations it may be preferable to carry several dimensions for health outcomes though and to make the reduction to a single (aggregated) dimension at the latest stage.

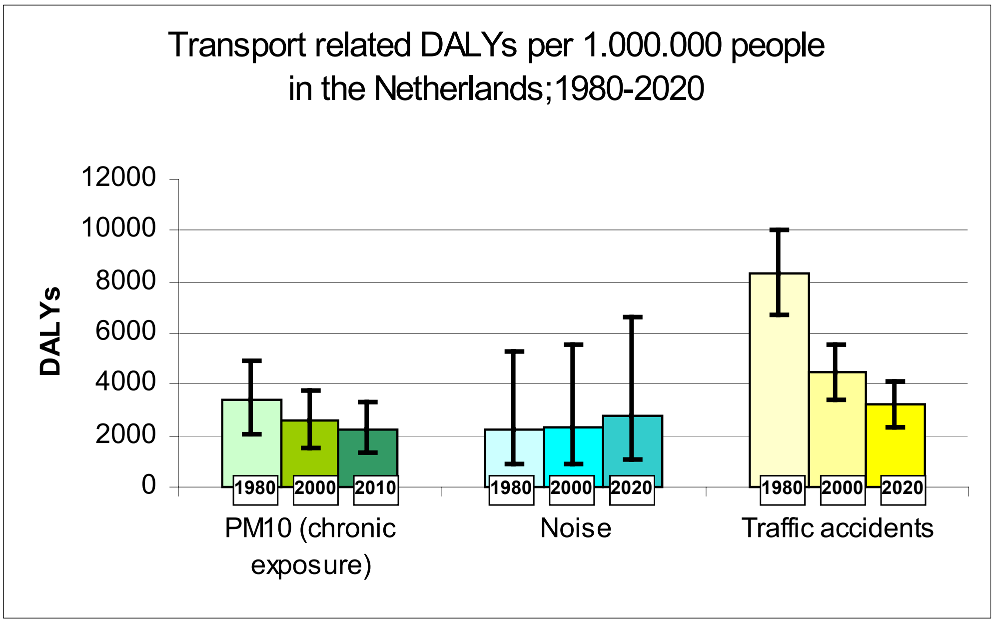


Figure 4.3 Transport related disease burden in the Netherlands

5. Economic valuation of noise impacts

5.1 Available methodology for cost-benefit analysis

Environmental policy is focused on preventing unwanted environmental effects in order to protect the environment and human health. Measures to reduce transportation noise and to maintain environmental noise quality are costly to implement. An obvious question is whether the social benefits of reduced noise levels can justify these high costs. In order to find the social optimal level of investments in noise reducing measures we need to know the social costs of noise (or external costs), preferably expressed in Euros (Navrud, 2003).
The problem is that costs and benefits are measured in different units. Monetary values seek to overcome this problem of comparability.

Valuation of health impacts of traffic noise exposure is the last part of a three-stage process, consisting of:

1. calculation of exposure
2. calculation of population impacts using exposure-response relationships
3. valuation of health impacts in monetary terms:
    a. estimation of unit costs of impacts identified in monetary units
    b. estimation of the mean aggregate monetary value of the noise-related health impacts

It is important to bear in mind that every stage has its own uncertainties. Exposure data are often not complete, due to lack of data on traffic, characteristics of surroundings and time-activity patterns (see section 2). For adults, exposure response relationships are available for annoyance (road, rail and air traffic), (perceived) sleep disturbance (road, rail), and (indicatively) for cardiovascular risk. For children no generalised exposure response relationships are available as yet. This means that, even before starting to make cost estimates of effects, there is a substantial uncertainty on whether the starting point of the valuation is the right one.

The purpose of the valuation is usually to express the severity of the noise problem in terms of changes in welfare. Welfare consists of three components: (i) resource costs i.e. medical costs paid by the individual, health service or insurance, (ii) opportunity costs i.e. the costs in terms of lost productivity and the opportunity cost of leisure (leisure time loss), (iii) disutility i.e. other social and economic costs including any restrictions on or reduced enjoyment of desired leisure activities, discomfort or inconvenience (pain or suffering), anxiety about the future, and concern and inconvenience to family members and others. In health valuation literature the first two components are summed to produce what is known as the 'Cost-Of-Illness' (COI) measure of welfare. All three components are thought to be non-overlapping. Yet, there is a clear danger of overlap, since any individual tends to include in its assessment of loss of welfare both financial and non-financial concerns. In the case of noise, disutility clearly dominates over eventual medical costs. Therefore, valuation techniques for noise concentrate on putting a price on the utility loss. Welfare loss is expressed in monetary terms.

In order to assign a monetary value to goods and services, economists make use of the Willingness To Pay: the maximum amount a person is willing to pay to obtain a good or service. In the field of noise the revealed preferences and stated preferences-methods are used

most often. Methods based on revealed preferences consider noise (or better: silence) as a free-market good that can be bought. It is included in the price of dwellings, for example. By comparing real estate prices in neighbourhoods with different noise loads, the price of silence (or the price on avoiding negative health effects) comes out. With the stated preference methods people say how much they value a certain good, for example a silent environment. There seems to be a growing support for this kind of methodology. Nevertheless, both methods have their advantages and disadvantages (SIKA 2003, Dusseldorp et al., 2001; Nijland et al., 2003). One thing is clear, both methods are aimed at adult behaviour and adult responses (children do not buy houses and usually children are not considered as respondents in surveys). Thus, for the valuation of noise impacts for the time being it is recommended to use the Willingness To Pay (WTP) data from studies in adults. For the assessment of children's health costs the use of COI is limited since effects on cognition and well being (annoyance) are not included in this type of valuation.
For a more detailed overview of the noise valuation studies, which have been carried out, see the report of the Stockholm workshop (SIKA, 2003).

5.2 Cost-benefit analysis studies in EU

It has been estimated that for Europe as a whole, the overall external costs (abatement costs) of road- and rail-traffic noise amount to 0.4 % of the total GDP (ECMT, 1998). In the Netherlands, as in most European countries, making the cars and trains more silent is regarded as the preferred way of noise abatement (as opposed to the construction of noise barriers, for example). It surely is the most cost-effective way, as many studies have shown. The implementation of several source-measures on cars and trains in the Netherlands will cost about 2 billion Euros (net present value). On the other hand, the benefits in terms of reduced annoyance are estimated to amount to about 4-6 billion Euros (Van Kempen et al., 2001). Thus, the benefits of source measures for the Dutch society as a whole would exceed the costs. The EU-working group on health and socio-economic aspects (based a.o. on a review of studies in different EU-countries; Navrud, 2003) estimates that every household values the benefits of noise reductions with 25 Euros per decibel per year. No data are known about the value children put to noise reduction.

A limitation of the estimates presented above is that they only consider the annoyance impact of noise. Just recently, several studies proposing monetary values for other noise related effects such as sleep disturbance and ischaemic heart disease have been published though (table 5.1). In the framework of the UNITE-project monetary values for each impact have been derived as the sum of (i) WTP to avoid each type of episode of ill health, (ii) health care costs of treatment when relevant and (iii) productivity loss. Based on generalized exposure-response functions for each impact and the costs of the separate impacts, the total costs were calculated (table 6.2). Amenity losses were estimated using a NSDI-value derived from revealed preference (Hedonic Pricing)-studies. Subsequently, these values can be applied in order to calculate the total welfare loss from noise or the total increase in welfare due to noise reducing measures. This method of calculating the external costs of transport noise is called the Impact Pathway Approach (IPA). It has to be kept in mind that these are worst-case calculations. Considerable uncertainty is attached to the economic estimates of myocardial infarction, hypertension and sleep disturbance, especially since there is still debate about the epidemiological evidence for cardiovascular risk. It is advised to add some weight factor to the calculations based on the existing amount of evidence for the causal relationship between noise exposure and some of these health impacts.

Table 5.1 Economic values for road noise derived from reviews, which can be used for calculating external costs of noise.

Health effects from noise exposure	Expectancy value (per 1,000 adults exposed)	Costs (Euro) per unit	Source
Myocard infarction (MI): Fatal, years of life lost (YOLL) Non-fatal, days in hospital Non-fatal, days absent from work Expected cases of morbidity	 YOLL=0.084*(Lden-5.25)[a)] #days=0.504*(Lden-31.5)[a)] #days = 0.896*(Lden-56)[a)] # = 0.028*(Lden - 1.75)[a)]	 96,500 Euro/YOLL 680 Euro/cardio-related inpatient days 100 Euro/day 14,400 Euro/case to avoid morbidity	Bickel et al., 2004
Angina Pectoris: Days in hospital Days absent from work Expected no of morbidity days	 #days=0.168*(Lden- 10.5)[a)] #days=0.684*(Lden - 42.8)[a)] #days = 0.240*(Lden - 15)[a)]	 680 Euro/cardio-related inpatient day 100 Euro/day 230 Euro/day to avoid morbidity	Bickel et al., 2004
Hypertension Days in hospital	 0.063 (Lden - 4.5)[a)]	 350 Euro/inpatient day	Bickel et al., 2004
Sleep disturbance road traffic	0.62 (Lnight - 43.2)[b)]	220 Euro/person/year (COI)	Bickel et al., 2004
Annoyance SP studies % depreciation in house prices per 1 dB(A) increase in noise		 23.5 Euro/dB/Household/year 0.55 (range 0.08 - 2.22) %	 Navrud, 2003 Bateman et al., 2002

a) Threshold is 70 dB(A) Lden; b) Threshold is 43.2 dB(A). Other assumptions: Myocardial Infarction, 7 years of life lost per fatal heart attack in average; base risk is 0.005 and survival probability: 0.7; Angina Pectoris, base risk: 0.0015. The Lnight as assessed outside at the most exposed facade.

Table 5.2 Estimated costs (€) for impacts due to noise in Europe, the Netherlands and Switzerland (average costs per case), which can be used in future cost-benefit analysis of noise (source: UNITE-project, calculated from Suter et al., 2002; Certan et al., 2003)

Impact	Avg Europe	Netherlands	Switzerland
Myocardial infarction (fatal, 7 YOLL) Total per case	 522900	 592000	 664314
Myocardial infarction (non-fatal, 8 days in hospital, 24 days at home) Medical costs Absentee costs WTP Total per case	 4720 2820 15070 22610	 5680 3140 17630 26450	 6400 2808 16905 26113
Angina Pectoris (severe, non-fatal, 5 days in hospital, 15 days at home) Medical costs Absentee costs WTP Total per case	 2960 1760 9440 14160	 3560 1960 11040 16560	 4000 1755 3520
Hypertension (hospital treatment, 6 days in hospital, 12 days at home) Medical costs Absentee costs WTP Total per case	 1830 1580 550 3960	 2210 1760 620 4590	 2552
Medical costs due to sleep disturbance (per year)	197	223	264
Average (net) rent per person per year (basis of calculation of WTP for avoiding amenity losses)		1285	

Using the values from the UNITE project the total external costs from noise exposure for Zurich airport were estimated at 17.7 million Euro per year (1998 prices). Disutility costs (annoyance) of noise dominate over eventual medical costs.

6. Policy analysis and options

6.1 Policy analysis and risk management

Risk management is the process of deciding what should be done about a hazard such as noise, the population exposed or (potentially) affected, implementing the decision, and evaluating the results. The use of causal chains or causal webs is helpful in the evaluation of the potential of risk reducing measures (figure 6.1).

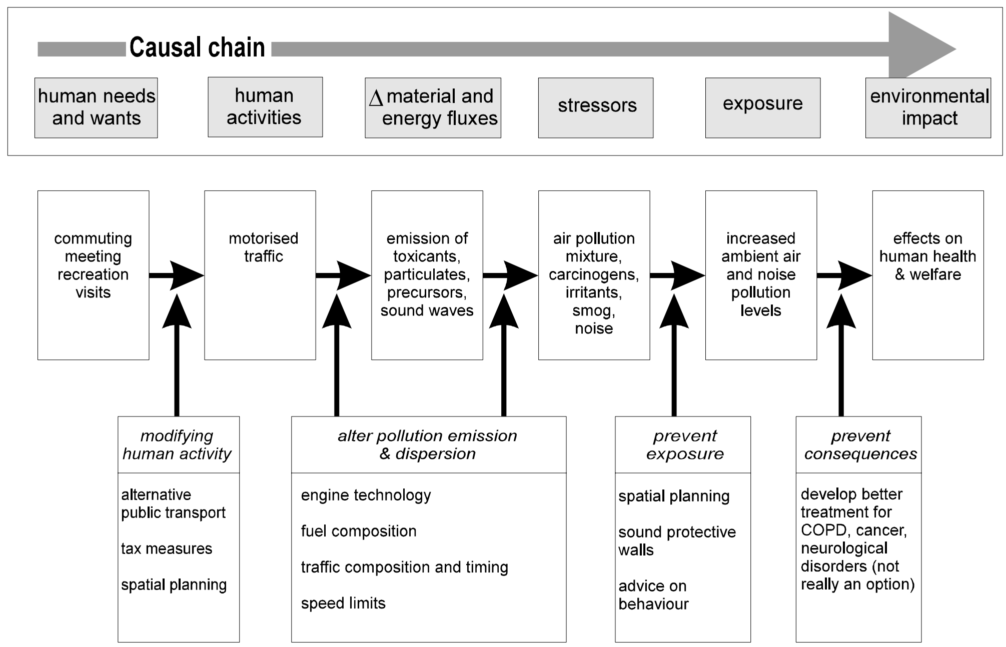


Figure 1. Relevance of causal chain structure for risk management: the example of traffic emissions

Figure 6.1 Causal chain depicting the relationship between traffic emissions and potential health impacts. This provides a framework for the evaluation of the potential of traffic measures to reduce exposures and health risks.

Noise exposure can be reduced by measures at the source (e.g. reduction of traffic, reduction of emissions, traffic management, driver behaviour), prevention of dispersion (e.g. spatial planning), or measures that reduce exposure of the dwelling (noise barriers, insulation measures) (figure 6.1). The overall relationship between traffic volume, noise emissions and health effects such as annoyance and sleep disturbance is well established. A difficulty in determining the health effects of noise abatement measures is that annoyance is not only dependent on noise exposure levels but also on non-acoustical factors like individual noise sensitivity, fear with respect to the source and perceived control over the situation. Interventions on these non-acoustical factors can also be successful in reducing annoyance by noise.

In practice noise management consists of different policy instruments and measures such as:
1. Elimination of unacceptable levels by a legal limit;
2. Preservation and extension of quiet (residential and natural) areas by policy targets;
3. Improvement of the acoustical quality in residential areas by noise barriers, traffic measures and zoning.

6.2 Effects of noise abatement measures

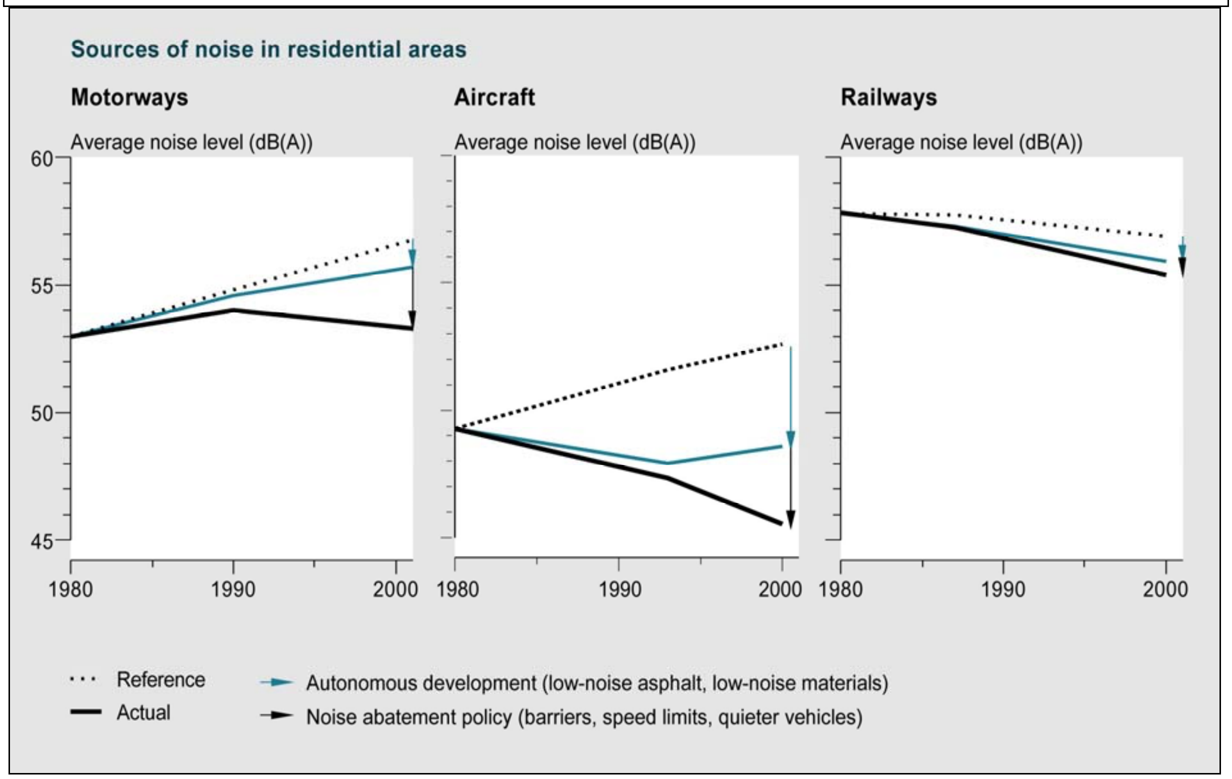
Exposure to high noise levels has decreased substantially in some countries due to (a) spatial planning, (b) traffic volume management or (c) technical improvements. Spatial planning, especially at the local level, may influence noise loads by separating source from receiver. Volume reducing measures typically interact with spatial planning (car-free zones, park and ride facilities etc.). Technical measures may reduce emission levels (e.g. silent pavement, grinding of the railroad tracks or new break systems on freight wagons), may hamper the transmission of noise (noise barriers) or may eventually, if other measures are impossible or too expensive, reduce indoor noise levels (isolation).
A reduction of noise exposure was established in the Netherlands by noise barriers along motorways, introduction of speed limits and the use of quieter vehicles (case study; box 1).

Spatial planning
Lately more and more studies are focusing not merely on simple dose-response relationships but also on the whole setting in which the noise is perceived (soundscape). A negative response to noise such as annoyance can be modified by both acoustical and non-acoustical parameters. People that feel unsafe due to road traffic are more annoyed than others. Improvements that include traffic safety measures can therefore be an effective way of reducing the noise annoyance even though the actual noise level remains unchanged. Negative reactions to noise can be influenced by other exposures (eg air pollution). New studies that focus on the exposure pattern indicate that people (including children) who have access to a quite side of their house, a quiet back yard or even a quiet park nearby are less annoyed by noise than people without access to such areas. This fact gives the city planner an additional tool for reducing noise annoyance (Gjestland and Job, 2003). However, the promotion of these type of measures should only be done in parallel to active noise-reducing measures.

Noise limits and targets
WHO gives a guideline for outdoor areas of L_{Aeq} 55 dB(A) for serious annoyance (table 3.1). This does not mean that exposure to levels below this limit does not cause annoyance. The new EU directives do not provide specific standards, this is up to each member country. One action proposed by the EU is the production of noise maps including people exposed to noise levels equal to 55 dB(A) (L_{den}). At this level a substantial number of people will still be annoyed. On the other hand the EU is also making plans for identifying and preserving quiet areas. Norway is perhaps the only country that has a goal to reduce annoyance. The Netherlands abandoned this goal in their most recent Environmental Policy Plan.

BOX 1 Dutch noise abatement policy has effects

In the Netherlands, exposure to noise from motorway, rail and air traffic has declined slightly since 1980, despite the doubling of traffic volumes. Erection of noise barriers, the use of highly pervious asphalt and quieter lorries has all helped to reduce average motorway noise levels in residential areas (see figure). To further enhance road safety, open asphalt is applied on about half of the highways. The open asphalt has an acoustical side-effect, i.e. a reduction of the noise emission with 2 –3 dB(A). It is expected that by 2010 all highways have open pavement, or silent variants of it, like two-layered open asphalt, which may reduce noise emissions with 4 – 6 dB(A). Noise barriers reduce the exposure levels with 5-15 dB(A), depending on height, material and local circumstances.
To maintain this trend an effective noise abatement policy will still be needed in the future because the volume of the traffic is expected to continue to rise. Renewal of the aircraft fleet and optimisation of runway use and flight paths have reduced average levels of aircraft noise in residential areas, despite a quadrupling of the number of flights. The noisiest aircrafts are no longer permitted to use the airport. Although noise abatement policies are having an effect, a considerable number of homes in the Netherlands still experience high levels of noise and annoyance levels are not decreasing. In particular, city centre traffic noise has not been reduced at all. At 40,000 to 60,000 dwellings in the Netherlands facade noise levels exceed 70 dB(A), the limit value for 2010 laid down in the Fourth National Environmental Plan. Extra measures are needed to meet this limit value. Low-noise asphalt and reduced speed limits are more cost-effective than erecting noise barriers; instead of just solving local problems they can reduce noise levels throughout a wider urban area. At some hot spots (e.g. Overschie), where all (technical) measures have already been taken, a maximum speed of 80 km/h is implemented. This reduces noise levels by 2-3 dB(A) and NO_2-levels by 10% [RIVM, 2003]. About 10 km/h speed reduction leads to approximately 1 dB(A) noise level reduction.



Innovative research to reduce traffic noise
In the Netherlands the Ministry of Transport, Public Works and Water Management and the Ministry of Housing, Spatial Planning and the Environment have initiated a sizeable research and development program to reduce road traffic noise: the Innovation Program Noise (IPG). This program has to result in noise reducing measures, which make it possible to reach strong strategic goals regarding the reduction of noise impacts on the population. The focus is on source- oriented measures, which are generally more cost-efficient than effect related measures. A similar program is initiated for the national railway system.

The Innovation Program, with a budget of more than 50 million euros, addresses the following topics:
- investigation of the potential noise reductions by improvement of road surfaces, tyres and vehicles and noise barriers;
- scientific research into the knowledge needed to realize the reduction effects;
- development of the technologies and products to a level of general application in the national main road and vehicle population.

The program must result in a significant reduction of the noise production (including shielding effects) of the main road network system. In case of combinations of measures after 4 years of IPG for every location, the technology for 8 dB(A) noise reduction will be feasible.

6.3 Policy options

In general, in the case of noise, spatial planning, traffic volume management and technological measures are the most effective measures in reducing noise exposure levels. Low-noise asphalt and reduced speed limits are more cost-effective than erecting noise barriers; instead of just solving local problems they can reduce noise levels throughout a wider urban area. Specific measures are needed for the most annoying noise sources such as mopeds.

A large variety of measures are available, which can reduce the impact of traffic noise on health at the local, national or EU level. Based on several discussions with policy-makers and noise experts, the following priority options have been identified, based on the magnitude of expected reductions in traffic noise-related health impacts (++, see table 6.1 for more details). The focus has been *mainly* on measures dealing with road and rail transport noise.

Priority options with estimated highest impact (per policy area):

Transport policy
- Introduce traffic calming measures in urban areas.
- Extend and improve safe bicycle and pedestrian infrastructures.
- Integrate noise limits in transport policies. Distinguish between existing and new situations, since they might be different with respect to the measures that are feasible. In new situations exposure should be avoided, whereas in existing situations effort should be put in reducing the traffic noise level. Give priority to measures, which reduce outdoor noise levels compared to levels, which only reduce inside levels (like insulation).

Spatial/urban planning

Integrate transport policy with spatial and urban planning. Prevent or reduce children's (and adult's) noise exposure by taking care of their interests in the spatial planning process. This means reform of the design standards for infrastructures:

- Promote the establishment of quiet areas in residential areas and around schools to allow adults and children to recover from long-term noise exposure.
- Avoid location of (new) schools, day-care centres and hospitals near busy roads, railway tracks and airports, but locate them still within easy reach of residents.
- Take noise into account as a part of the designing process of the school or kindergarten itself (orientation towards quiet side, isolation etc.).

Education

- Educate driving instructors and drivers. 'Eco-driving' or 'driving on velvet paws' will safe energy, and reduce noise and air pollution emissions. Include field tests showing actual travelling times at different travelling speeds.
- Raise awareness among parents, schoolteachers, but also teenagers and children about the effects of noise, including the hidden effects. Make citizens aware of their own contribution to the problem. Make people aware of the risks of loud music (walkmans, discotheques) and noisy toys.
- Promote walking and cycling.

Regulatory and technological measures

- Traffic calming measures: Reduce and enforce speed limits e.g. during certain hours.
- Restrictions for nighttime traffic especially for heavy duty vehicles: Measures should be taken to reduce noise exposure in children during their sleep in consideration of the potential long-term effects of the physiological stress response. The '24 hours' economy will lead to an increase in (freight) transportation during night-time. Quiet times should be protected by, for instance, significantly decreasing the number of trucks during the night and weekends and inhibiting night-time flights. Allow only low-noise trains or limit the number of trains at night. One should, however, be aware of the side effects these noise measures could cause on other pollutants. When a decrease in trains for instance leads to a loss of market share for railway transport in favour of road transport, this will increase air pollution.
- Develop, enforce and control implementation of EU-regulations for more silent rail, aircraft and road vehicles, tyres and surfaces. Outside the scope of transport, EU-regulations for other noise sources, e.g. to limit noisy toys, are recommended too.
- Impose tighter noise requirements for sensitive areas. WHO gives a guideline for outdoor areas of L_{Aeq} 55 dB(A). At this level, a considerable number of people will still be annoyed.
- Optimisation of runway use and flight paths.

Fiscal measures

In the case of noise, fiscal measures discouraging car use have a low impact on actual noise levels. A large decrease in traffic volume is needed, for a relatively small decrease in noise level. Incentives for silent tyres and low-emission vehicles are probably more effective.

Table 6.1 Options for measures to regulate transport noise. Impact-scores and priorities are based on the results of joint discussions with policy-makers and noise experts.

Instrument	**Level of Impact**						**Implementation Strategy**							**10 prior items**
							Level				**Timeframe**			
	Estimated level of impact (noise reduction): **high = "++" or "- -"** **low = "+" or "-"** **no impact =** ***empty***						**Level:** **LO = local** **RE = regional** **NA = national** **EU = Europe**				**Start for implementation:** **S = short/immediately** **M = medium (1-5 y)** **L =long (> 5 years)**			
		NO					**LO**	**RE**	**NA**	**EU**	**S**	**M**	**L**	
FISCAL, ECONOMIC														
Use of roads and public space														
Road tolls on major intercity network		+						X	X			X		
Road pricing for cars, LDV's and buses		+							X	X		X		
Parking fees extension and increase		+					X							
Vehicles and technology														
Incentives for electric and clean, silent (tyres) and ultra-low emissions vehicles		+							X	X			X	
Provide incentives for hybrid and fuel cells for vehicles		+							X	X			X	
Incentives for public transport fleet renewal		+							X					
Incentives earlier replacement of older diesel trucks by new (less polluting) trucks.		+							X	X	X			
Environmentally friendly modes and habits														
Subsidies to promote multi-modal transport		+							X					
Employee travel: Charging for parking, incentives for public transport and cycling		++							X		X			X
REGULATORY														
Fuels and emissions														
Technical check of exhausts		+					X		X					
Road traffic														
Restrictions for HDV(night bans, weekend bans)		++					X		X			X		X
Traffic calming in towns: Improving city-logistic, 30 km/h limit, reducing parking space, car free zones		++					X				X			X
Reduce speeds and control on non urban road infrastructures		++						X	X					X
Access restrictions for conventional vehicles in urban areas plus adequate signalisation in cities		++					X				X			X
Regulations in urban areas in order to give preference to environmentally friendlier cars		+					X							
Improvement of the traffic flow (driving in the same speed, traffic light adjustment)		+						X	X		X			
Environmentally friendly modes														
Reform of the design standards for infrastructures, transport codes, zoning regulations to promote walking, cycling, public transport		+					X	X					X	
Noise regulations														
Noise: Regulations for rail and road vehicles, tyres, surfaces and for aircraft emissions		++							X	X				X
Impose tighter noise requirements for sensitive areas		++					X	X	X					X
Optimisation of runway use and flight paths		+					X				X			
Noise abatement plans and measures		++					X	X	X					X

Table 6.1 continued	***Options for measures/policies to regulate transport noise***													
Instrument	**Level of Impact**						**Implementation Strategy** **Level**				**Timeframe**			**10 prior items**
	Estimated level of impact: **high = "++" or "- -"** **low = "+" or "-"** **no = *leave empty***						**Level:** **LO = local** **RE = regional** **NA = national** **EU = Europe**				**Start for implementation:** **S = short/immediately** **M = medium (1-5 y)** **L =long (> 5 years)**			
Silent roads (surfaces)		++					X	X	X	X		X		X
INVESTMENTS														
Fuels and emissions														
Extension of rail infrastructure for freight and passengers		-												
Extend and improve bicycle and pedestrian infrastructures		+					X	X				X		
Extend and improve infrastructures and services for regional rapid public transport		+						X	X			X		
Governance arrangements														
Policy integration														
Integration of land use and transport policy, Urban planning		++					X	X	X					X
Monitoring														
More severe control of speeds and driving times of HDVs		+							x	x		x		
Monitoring system for noise emissions/		+/-							x			x		
EDUCATION AND HORTATORY														
Implement nationwide awareness programmes		+							X	X				
INNOVATION														
Car free day initiative		+					X	X			X			
Implementation of telecommunication technologies in order to achieve a more efficient and environmentally friendly logistic and mobility (navigation system)		+					X					X		

6.4 Development of a strategy towards reduction of transport-related noise exposures

Since the type and size of transport problems differ per country, region and urban area, different (packages of) measures are needed containing one or more of the above noise-reducing measures. In developing such a package, it is recommended to give priority to those interventions that also address other transport related health effects, since this allows for economic efficiencies and synergies. For example, measures that reduce the volume and speed of traffic around schools and within or around residential areas will reduce noise, air pollution, energy use and improve safety as well.
Thus, a strategy towards noise reduction cannot be developed on its own but should be related to an overall strategy to prevent and reduce transport-related health impacts. Below, some of the steps are described that could be part of such a strategy. Some of them relate specifically to noise, some are more generic.

1 Identify the problems and priorities

In Europe, transport (road, rail and air traffic) is the most important source of community noise. Noise exposure causes annoyance, sleep disturbance and has effect on children's learning. It is also suspected to contribute to the development of cardiovascular disease. In general, the increasing demand for mobility and the increase in car use and air travel is the main driving force to tackle. Especially in densely populated areas, the increase in transport and related infrastructure causes problems, including community noise.

2 Identify objectives

In the framework of policy development and target setting it is recommended to calculate and compare health impacts and costs of different plans and scenarios. Goals and thresholds for action, monitoring and evaluation of policy/development plans need to be defined. Objectives need to be defined for urban and transport planning with regard to e.g. the design of quiet areas, location of schools and location of infrastructure promoting walking and cycling.

3 Promote discussion between parties involved

Cooperation between traffic sector, land use and urban planners, and health specialists should be enhanced. If health specialists can be involved more in urban/spatial planning processes, the health consequences of transport-regulating or urban planning can be identified and dealt with earlier.

4 Develop noise abatement programmes and action plans

Many countries in the WHO-Europe Region are developing or already implementing a National or sometimes even regional Environment and Health Action Plan. Targets for mobility, conditions for urban planning and targets for raising public awareness about transport-related health issues can be incorporated in such action plans, which can serve as a base for sector policies and local plans.

5 Monitoring and evaluation

Map and monitor the noise exposure of the population, using the EC-Directive for noise calculations. In the Netherlands, for example, the (development in) noise exposure of the population is mapped and evaluated yearly, results are published in the Environmental Balance (www.environmentaldata.nl). In several EU countries (e.g. Austria, the Netherlands, Germany, Sweden) annoyance surveys are being carried out on a regular basis.
Results of good policy or measures (good practices) should be fed directly in the PEP-process and made available to other countries.

6 Promote research for update/development of guidelines

Scientific reviews on evidence and consensus building on 'safe' noise threshold levels for different settings, activities and daytimes should be supported. WHO-guidelines are based on such reviews and specify noise levels for different settings, activities and times (Berglund, 1999, see table 3.1). In general, noise levels in residential settings should not exceed 55 dB(A), but a substantial number of people will still be annoyed at these levels. Threshold levels for schools are also defined (35 dB(A) L_{Aeq}, in schools) but the questions is how to reach these relatively low levels. The WHO guidelines for night-time noise do not allow acting towards reduction of peak levels. Also separate recommendations for aircraft noise should be considered, since aircraft noise has another peak-to-mean ratio than road traffic noise. Besides a scientific discussion on the existing WHO guidelines, development of new thresholds based on (new) research should be supported. Results of ongoing international studies should be fed directly in the WHO-process and made available to other countries.

7. Discussion and conclusions

7.1 Noise exposure assessment

Road traffic is the main source of community noise exposure in Europe. Differences in methodologies, however, preclude comparisons of the noise situation between countries. Noise indices and calculation methods differ per country and per transport mode. Differences in noise exposures between countries partly reflect actual differences, but for the greater part these differences are methodological artefacts and may lead to undesired differences in predicted noise levels up to 10 – 15 dB(A). Apart from these differences, some countries apply a 'bonus' in exposure levels for the noise of trains and/or cars. When comparing noise exposure data from different countries, one should be aware of (a) possible artefacts and of (b) possible country-specific, source dependent noise reductions.

The recent EU-directive on environmental noise aims at harmonising noise indices and noise calculation methods and therefore will improve the comparison of noise data in the future. For the moment, however, one has to deal with individual national noise indices and noise calculation methods.
Systematic assessment is lacking at the national as well as European level on how many children are exposed to what kind of noise levels. Knowledge on current exposure levels is needed as background for establishing new policy.

Recommendations

- For an European-wide comparison of noise exposure levels, it is recommended to use L_{den} and L_{night} as noise indices.
- If these data are produced using national calculation methods, it should be clear whether specific (national) reductions (e.g. Schienenbonus. or art. 103) have been applied. To enhance comparison, it is recommended to refrain from these reductions in reporting.
- For international comparisons, noise exposures should not only be estimated with the national calculation method, but also with a 'common' method, preferably the designated interim-method (EC, 2002 and section 2.3), to avoid methodological artefacts in the comparison.
- When population exposure models combining detailed source information at street or city level with population and built environment data are not available, a more crude approach like the one used by Roovers et al. (2000) might be used (section 2.4).

7.2 Health impacts of transport noise

There is sufficient evidence that noise exposure at community levels can produce various effects in adults, including annoyance and sleep disturbance. Hearing damage is unlikely to occur at typical levels of community noise exposure. The evidence for a causal association between noise exposure and increased cardiovascular health risk is limited. However, a small effect of noise exposure on blood pressure levels is deemed plausible, since the overall results of these studies on the full range of endpoints from slight elevation of blood pressure to cardiovascular disease are consistent with known stress reactions and cardiovascular disease progression and supported by laboratory studies on stress reactions and blood dynamics. Nevertheless, well-designed cohort studies with good exposure characterization will be needed to confirm these suggestions. In the meantime, some estimates for the cardiovascular risk associated with exposure to road traffic and aircraft noise are available which can be used to get an indication on the potential health impact.

There is sufficient evidence that children chronically exposed to high levels of *aircraft* noise, have impaired reading and memory as well as raised levels of annoyance and reduced motivation. The findings of studies examining the potential impacts of *road* traffic noise on cognition are inconsistent.
The evidence for an association between chronic exposure to aircraft noise and increased blood pressure in children is inconclusive. Children seem to be more vulnerable than adults with regard to cognition, but not with regard to annoyance. An important research question to answer is whether the observed cognition effects in children persist over time. Results from intervention studies suggest that the effects observed reverse after a decrease in noise exposure.
Although children appear to be less disturbed during their sleep than adults (with respect to awakenings and sleep quality) there is evidence for 'hidden effects' occurring during sleep (e.g. cardiovascular and hormonal responses). These effects do not seem to diminish (adaptation) and in the long term might accumulate, adding to the risk for e.g. cardiovascular diseases of hypertension.

7.3 Assessment of transport-related health impacts and costs: lessons learned

For assessment of the potential health risks of transport-related noise exposure, the following options are available:

- Comparison of community noise levels with limit values or policy targets ('distance-to-target'). WHO-guidelines are available which specify noise levels for different settings and activities (see table 3.1). In general, noise levels in residential settings should not exceed 55 dB(A) (Berglund, 1999). A substantial number of people will still be annoyed at this level though.
- Identification of ' hot spots' (areas with high exposure levels) or % population exposed to noise levels above reference or limit values (health risk indicator).
- Assessment of the health impacts or number of people affected.

The general approach for health impact assessment as described in section 1.2 could be adopted for the health impact assessment of noise. This type of approach involves some limitations though.
The first limitation is the uncertainty in exposure-response functions. The question is whether we only model those effects for which there is sufficient evidence for causality (annoyance, sleep disturbance) or whether we also include effects for which the evidence is more limited (cardiovascular diseases). To overcome this problem it is recommended to add a weight factor for the strength of evidence to the calculations.

The transferability of risk-ratios/exposure response relationships from one population to another (differences in susceptibility, base-line risk) is another source of uncertainty. Most risk estimates for noise are based on studies in adults. The estimates for cardiovascular diseases are based on males. Effects on cognition are mainly observed in children.

At the moment for children no generalised exposure-response functions are available which can be used for further health impact assessments. To assess annoyance, the relationships recommended for adults in the EU-guidelines may be used. This can result in a (slight) overestimation since children seem to be less annoyed than adults. For comparison, the annoyance curves from the TYROL studies (for rail and road) and RANCH (for aircraft noise) may be used.

When comparing the outcome of national surveys one should be aware of the use of different questions to assess annoyance. ISO technical specification ISO/TS 15666:2003 provides a basis for further harmonisation on annoyance assessment.

Recommendations for health impact assessment of noise exposure:
- Where risk estimates for annoyance or sleep disturbance based on national surveys of good quality are available, this is preferred (WHO, 2004). If not, the relations as described in table 7.1 can be used to predict annoyance or sleep disturbance levels, taking into account the prerequisite of these curves (e.g. only for comparative assessments, not for assessment of local and changing situations). The inclusion of correction factors for insulation or window behaviour is recommended.
- With regard to cardiovascular diseases, some risk estimates for road traffic and aircraft noise are available for adults. The thresholds of no-effect (or reference level) to be used are still debatable though. No estimates are available for railway noise.
- To assess the potential impact of aircraft noise on cognition the upcoming exposure-response functions from the RANCH-study (Stansfeld et al, 2003) may be of use.

Cost-benefit analysis
Noise abatements are expensive but if abatement fails, noise may have adverse effects on health and well-being. These adverse effects can be expressed in monetary terms.
Valuation techniques for noise impacts concentrate on calculating a price on the utility loss (social and economical costs), e.g. the Willingness To Pay-methods. Households in the EU are willing to pay 25 Euros for a noise reduction of one decibel per year (Navrud, 2003). No data are known about the value children put to noise reduction.
Noise related health effects such as sleep disturbance and ischaemic heart disease have rarely been given a monetary value. As one of the first, the EU-funded project UNITE derived monetary values for these health impacts. Amenity losses were also estimated. However, it has to be kept in mind that considerable uncertainty is attached to the economic estimates of myocardial infarction, hypertension and sleep disturbance, due to the uncertainties in the exposure-response functions used and in the estimations of duration and severity of impacts.

Recommendations for cost benefit analysis of noise measures:
- Economic values are available for health and social impacts of road noise and aircraft noise which can be used for calculating external costs of noise (section 5).
- The use of Cost of Illness is limited since effects on cognition and well being (annoyance) are not included in this type of valuation.
- The monetary values derived for the health impacts of noise within the UNITE framework need further validation by health professionals.

Table 7.1 Exposure response relationships which can be used to assess health effects of traffic noise in the European Region (sources: Miedema and Oudshoorn, 2001, Miedema et al., 2003; Van Kempen et al.,2002; Passchier-Vermeer et al., 2003))

Effect	**Noise metric**[a]	**Source**	**Popula-tion**	**Exposure-response relationship**
Annoyance				
Percentage annoyed	Lden	Aircraft	Adults	%A = $8.588 \cdot 10^{-6}$ (Lden-37)^3 + $1.777 \cdot 10^{-2}$ (Lden-37)^2+ 1.221 (Lden-37);
	Lden	Road		%A = $1.795 \cdot 10^{-4}$ (Lden-37)^3 + $2.110 \cdot 10^{-2}$ (Lden-37)^2+ 0.5353 (Lden-37);
	Lden	Rail		%A = $4.538 \cdot 10^{-4}$ (Lden-37)^3 + $9.482 \cdot 10^{-3}$ (Lden-37)^2+ 0.2129 (Lden-37);
Percentage highly annoyed	Lden	Aircraft	Adult	%HA=$-9.199 \cdot 10^{-5}$ (Lden-42)^3 + $3.932 \cdot 10^{-2}$ (Lden-42)^2+ 0.2939 (Lden-42);
	Lden	Road		%HA = $9.868 \cdot 10^{-4}$ (Lden-42)^3 - $1.436 \cdot 10^{-2}$ (Lden-42)^2+ 0.5118 (Lden-42);
	Lden	Rail		%HA = $7.239 \cdot 10^{-4}$ (Lden-42)^3 - $7.851 \cdot 10^{-3}$ (Lden-42)^2+ 0.1695 (Lden-42)
Sleep disturbance				
Motility (mean)	L_{night}	Aircraft	Adults	M_{night} = 0.000192 x (L_{night} - Ldiff1-Ldiff2)[b]
Percentage highly sleep disturbed sleep disturbed a little sleep disturbed	L_{night}	Road	Adults	%HSD = $20.8 - 1.05L_{night} + 0.01486L_{night}^2$ %SD = $13.8 - 0.85L_{night} + 0.01670L_{night}^2$ %LSD = $-8.4 + 0.16L_{night} + 0.01081L_{night}^2$.
highly sleep disturbed sleep disturbed a little sleep disturbed	L_{night}	Rail	Adults	%HSD = $11.3 - 0.55L_{night} + 0.00759L_{night}^2$ %SD = $12.5 - 0.66L_{night} + 0.01121L_{night}^2$ %LSD = $4.7 - 0.31L_{night} + 0.01125L_{night}^2$.
Cardiovascular diseases				
Hypertension	$L_{Aeq.7\text{-}19}$	Air	Adults	RR = 1.26 (CI = 1.0-1.13), see further table 3.3

a outdoor at the most exposed façade
b Ldiff1 : difference between L_{night} en L_{Aeq} most exposed façade. default = 0 dB(A)
Ldiff2 : difference between L_{Aeq} outdoor and in the bedroom. default = 21 dB(A)

Research recommendations to improve health impact assessments and cost-benefit analysis with a special focus on children

- Develop exposure response functions for traffic noise exposure and children's cognition/ annoyance/blood pressure, based on ongoing studies e.g. RANCH. Use these coefficients for future health impact assessments and adapt them when new expert-reviewed results from major studies become available.
- Map the current and monitor the future noise exposure of children (at home and at school), taking into account housing conditions, the location of schools, kindergartens, playgrounds etc. Use the EC-guideline for noise calculations and metrics. Collect data on individual performance of children for selected subjects. By following the same individuals and by comparing children in high- and low-noise exposed living areas and school areas investigate whether noise has a long-term impact on children's cognitive development and whether different noise sources have different impacts. On the basis of this study, further decisions can be made whether additional noise guidelines for children's settings are needed.
- Include other stressors (air pollution!) and markers of effect (annoyance, quality of life, behaviour, stress responses) in noise studies. Identify psychological, social and physical protective factors (eg restoration). This may provide data on relative importance of noise

exposure. In addition, better information on the context (soundscape) in which adverse effects occur can help architects and land use planners in designing environments which better fit the needs of children (Lercher, 2003).

- Advocate use of standardised annoyance questions for children, e.g. the scales used in the RANCH –study.
- Support further research on the effects of traffic noise on sleep and cardiovascular risk in children. Evaluate findings from ongoing field studies (HYENA) where effect of combined exposure of noise and air pollution is studied.
- Support assessments of socio-cultural, economical, and also political factors which influence annoyance and disturbance responses in order to feed the decision makers toolbox (e.g. public participation).
- Assess the health gain of reduction of exposures versus effectiveness and costs of intervention measures, taking into account the influence of non-traffic noise sources e.g. by using the DALY method. An approach limited to Cost of Illness (COI) is not sufficient since no estimates are available for effects on cognition.
- Promote intervention studies and identification of best practices of preventing harmful effects of noise in children.

7.4 Policy options to reduce noise-related health impacts

A combination of technological measures (e.g. reduction of emissions, road surface) and spatial planning has proven to be successful in different EU-countries. Despite the enormous growth in traffic, noise levels have not or hardly increased. Since noise levels are expected to rise in the next decades, however, other measures in addition to ongoing technological developments and standards are needed. Reduction of speed, traffic-calming measures and promotion of other travel modes (eg. cycling) seem promising. The benefits of noise measures on health and welfare of the population exceed the costs of abatement measures, as some analyses show.

A large variety of policies and measures are available, which can reduce the impact of traffic noise on health at the local, regional, national and supranational level. A number of priority options have been identified in joint discussions with noise experts and policy-makers (see table 6.1), based on the (magnitude) of expected reductions in noise-related health impacts:

- Development of child-friendly mobility plans, with attention for infrastructure and education measures promoting safe walking and biking by children and their parents.
- Traffic calming measures, such as reduction of speed limits and traffic volume in residential areas.
- Reduction of speed in non-urban roads, e.g. by promotion of eco-driving and education of driving instructors and drivers, including field tests showing actual travelling time at different travelling speeds.
- Night-time regulations for heavy lorries, noisy trains and aircraft in/over residential areas (significantly decreasing the number of trucks during the night and weekends and inhibiting/limiting aircraft and train noise at night).
- Incentives for employee travel.
- Integration of land use, transport policy and urban planning. Define objectives for urban and transport planning with regard to e.g. the design of quiet areas, location of schools and dwellings in relation to busy roads, railways and airports.
- Regulations for emissions of rail and road vehicles, aircraft; tyres and road surfaces. Enforcement and control of implementation of EU-guidelines.

- Further development and enforcement of (innovative) technological measures reducing emissions at the source and exposures.

In international frameworks (e.g. WHO) guidelines for noise exposures have been developed, which are the basis for noise limit setting. Besides a scientific discussion on the existing WHO guidelines, development of new thresholds based on (new) research should be supported. Results of ongoing international studies should be fed directly into the WHO-process and made available to other countries. In addition, results of good policy or measures (good practices) and innovative research should be fed directly into the PEP-process and made available to other countries.

Policy recommendations related to children's health

Transport noise poses a health problem that affects many people, including children. Since there are indications that chronic noise exposure may influence children's performance and well being it is advised to keep this in mind while developing and applying noise regulations, control measures and planning.

- Reduce children's noise exposure at home and school, by reducing the noise emissions or duration of exposure, especially from road traffic (tightening limits for tire noise is a very promising one, as well as reducing speed limits eg during certain hours).
- Reduce children's noise exposure by taking care of their interests in the spatial planning process. Take noise into account as a part of the designing process of the school or kindergarten itself (orientation towards noisy side, isolation etc.). Avoid locations of (new) schools and day-care centres near busy roads, railroad tracks and airports, but locate them still within easy reach of residences.
- Increase number of green/quiet areas where children with chronic noise exposure can recuperate.

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ANNEX 1 Overview of studies on noise and blood pressure in children

Table A1 Characteristics of studies on the association between noise exposure and blood pressure in children

Author	Country	Design	Population	N	Noise exposure			Adjustments [e)]
					Source(s)	Levels	Measurement	
Karsdorf, 1968	Germany	Cross	12-16 yr.	269	Road traffic	70 phon, 63 phon and very quiet area[b)]	SLM, interior	3, 22
Karagodina, 1969	Russia	Cross	9-13 yr.	NR[a)]	Air traffic	NR[a)]	??	-
Roche, 1982		Cross	NR[a)]	NR[a)]	Divers	-	Questionnaire[d)]	33
Cohen et al, 1980	USA	Cross	3 – 4th grade	262	Air traffic	$L_{Amax,\ mean}$ 74 dB(A) $L_{Amax,\ mean}$ 56 dB(A)	SLM, interior	22; 24, 26 – 32
Cohen et al, 1981	USA	Cross/ Follow-up	3 – 4th grade	262/ 163	Air traffic	$L_{Amax,\ mean}$ 74 dB(A) $L_{Amax,\ mean}$ 56 dB(A)	SLM, interior	22; 24, 26 – 32
Regecova, 1994	Slowakia	Cross	3 – 7 yr.	1542	Road traffic	$L_{24h,mean} \leq 60$ dB(A) $L_{24h,mean}$ 61-69 dB(A) $L_{24h,mean} \geq 70$ dB(A)	SLM	23
Evans et al, 1998	Germany	Before/after	9.9 yr.	217	Air traffic	$L_{eq,\ 24\ h}$ 62 dB(A) and L_{01} 73 dB(A); $L_{eq,\ 24\ h}$ 55 dB(A) and L_{01} 64 dB(A);	SLM	24; 25
Morell et al, 1998	Australia	Cross	Year 3	1230	Air traffic	15 – 45 ANEI[c)]		1-22
Evans et al, 2001	Austria	Cross	9-10 yr.	115	Road & rail traffic	$L_{dn,average}$= 46 dB(A); $L_{dn,average}$= 62 dB(A)	Calculated	3; 23; 28; 33-35

a) NR=Not Reported; b) 70 phon is about 70 dB(A); c) ANEI = Australian Noise Energy Index; d) Recall exposure to noise events; e) 1= Resident aircraft noise level; 2 = Road/rail noise sources; 3 = sexe; 4 = weight; 5 = subscapular skinfold; 6 = pulse rate; 7 = eating before school; 8 = salt on food; 9 = family history of high blood pressure; 10 = Parental history of high blood pressure; 11 = child history of high blood pressure; 12 = Speaking background; 13 = organised sport; 14 = child activity; 15 = play activity during recess; 16 = glass doors; 17 = insulation; 18 = top floor occupancy; 19 = large windows; 20 = timber/fibro house; 21 = ambient temperature; 22 = grade/schoolyear; 23 = age; 24 = socio-economic state; 25 = type of occupation in household; 26 = race; 27 = parent's occupational level; 28 = parent's educational level; 29 = number of children in family; 30 = numbers of months enrolled in school; 31 = height; 32 = ponderosity; 33 = body mass; 34 = family size; 35 = density (people/room).

Table A1(continued) Characteristics of the studies on association between noise exposure and blood pressure in children

Author	Device used	Position	Visits	Measurements per visit	Result
Karsdorf, 1968	NR	NR	NR	NR	The pupils in the school exposed to considerable traffic noise proved to have much higher blood pressure values than those of the other school[a)]
Karagodina, 1969	NR	NR	NR	NR	Blood pressure abnormalities were reported in children residing near airports in comparison to relatively quiet comparison groups [b)]
Roche, 1982	Mercury sphygmomanometer	Sitting	1	1	No relation found between noise exposure and resting blood pressure [b)]
Cohen et al, 1980	Automatic BP recorder (SR-2 Physiometrics)	Taken in a quiet room	2	1	Noise was significantly associated with elevations (3 mmHg) in both systolic and diastolic blood pressure
Cohen et al, 1981	Automatic BP recorder (SR-2 Physiometrics)	Taken in a quiet room	2	1	No differences in blood pressure as a function of noise were found
Regecova, 1994	Doppler phenomenon- based ultrasound device, 7.5x19.5 or 11x27 cm cuffs, K1 and K5	Supine, after 5min bed rest	1	2-3	Comparison of the mean blood pressure values showed significantly elevated levels of both SBP and DBP in noisy or very noisy environments in comparison with those in quiet environments
Evans et al, 1998	Automated monitor A&D Digital, UA 751	Sitting, with right arm supported at heart hight at table	2	4 + 6 baseline	Children living close to the new airport experienced elevation in resting blood pressure after the airport opened. The matched children in nearby communities experienced stable levels of resting blood pressure
Morell et al, 1998	Dynamap Vital Signs Monitor 8100 automated BP machine		1	3	Aircraft noise or other noise sources were not statistically linked either to systolic or diastolic blood pressure
Evans et al, 2001	Calibrated sphygmomanometer (bosch, Sysditon model)	Sitting	1 practice reading, 2 readings over a 6 min period		Children in the noisier areas had elevated resting systolic blood pressure. Diastolic blood pressure was lower in the noisier group.

a) maximal difference of 16 mmHg was found for both systolic and diastolic blood pressure in girls attending class 10, when comparing the quiet school with the most noisiest school; b) only qualitative results were available