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Selection and evaluation of exposure-effect-relationships for health impact assessment in the field of noise and health

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Abstract

Selection and evaluation of exposure-effect-relationships for health impact assessment in the field of noise and health

This report is a background document that can be used to assess the health impact attributable to noise in the Netherlands. To this end the available exposure-effect-relationships in the field of noise and health are reviewed and evaluated, using data published in the epidemiological literature as well as previous reviews. Their applicability for assessment in the Netherlands was demonstrated in case-studies.

Only the relationships describing the association between noise and annoyance, sleep disturbance and cardiovascular disease are considered to be suitable for health impact assessment purposes. Only the relationships for which the evidence for an association between exposure and effect was considered sufficient and which were derived either by means of a quantitative summary of published data or a re-analysis of individual data based on primary studies, were selected. Finally, recommendations were made for the applicability of these exposure-effect-relationships regarding the health impact of noise exposure.

Key words: noise, health, annoyance, sleep disturbance, cardiovascular disease, exposure-effect-relationships

Rapport in het kort

Selectie en evaluatie van blootstelling-effect relaties voor gezondheidseffectschattingen op het gebied van geluid en gezondheid.

In dit achtergrondrapport wordt de laatste stand van zaken weergegeven met betrekking tot blootstelling-effect relaties op het gebied van geluid en gezondheid en hun toepasbaarheid voor de inschatting van de effecten van geluid in Nederland.
Voor een aantal relevante gezondheidseffecten worden de beschikbare blootstelling-effect relaties besproken. Aan de hand van een aantal case-studies wordt de bruikbaarheid van de verschillende relaties voor gezondheidseffectschatting (GES) geanalyseerd. Alleen de relaties die de invloed van geluid op effecten beschrijven waarvoor bewijs was en die zijn afgeleid door middel van een meta-analyse of gepoolde analyse worden uiteindelijk bruikbaar bevonden. Het resultaat is een set van relaties en aanbevelingen die ingezet kunnen worden voor de inschatting van de effecten van geluid in Nederland. Niet alleen in termen van risico's, maar ook in termen van aantallen getroffenen.

Trefwoorden: geluid, gezondheid, hinder, slaapverstoring, hartvaatziekten

Contents

1 Introduction

1.1 Aims and objectives

At the moment, environmental health risk assessment is increasingly being used in the development of environmental policies, public health decision making, the establishment of environmental regulations and the planning of research. This not only involves the identification of environmental hazards, but also the quantification of the expected health burden: health impact assessment (WHO-Working Group, 2000). After selecting a set of endpoints for which there is sufficient evidence for an association with the risk factor under study (sometimes called hazard identification), the expected health burden due to an environmental exposure in a specific population can be quantified by combining data on population density with exposure distributions on the exposure (exposure assessment) and information on exposure-effect-relationships[1]. Exposure-effect-relationships are not only useful for estimating the number of people that is affected, but they can also be used to inform the public and to increase the public and political awareness. Furthermore, they are important when determining threshold and/or guideline values and they can play a role in monitoring.

This report is a background document that can be used when assessing the health impact attributable to noise in the Netherlands. To this end the relevant exposure-effect-relationships in the field of noise and health are evaluated. Finally, some recommendations are given for the applicability of these exposure-effect-relationships regarding the assessment of health impact of noise exposure. The contents of this report is a more extensive treatment of work based on material presented already in Staatsen et al (2004), a book chapter dealing with noise and health (Van Kamp et al., 2004) and the minutes of the WHO-Working group on noise and health indicators (WHO, 2002) (WHO, 2003).

1.2 Noise and health

In most of the industrial world, noise is a pollutant that is persistent and inescapable. One of the most important sources of community noise is caused by transport, comprising road and rail traffic, aviation and shipping (Schafer, 1971). It has been estimated that approximately 30% of the European Union's population are exposed to levels of road traffic noise of more

[1] Alternative phrasings are: dose-effect-relationship, dose-response-relationship, exposure-response-relationship. However, dose refers to an accumulated dose that stays in the body after exposure; response refers to the number of people affected (Briggs, 2003).

than 55 dB(A), and that 20% of the population of the European Union experiences noise levels that are considered unacceptable (Berglund et al., 1999). During the last decades, 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. However, due to the enormous growth in traffic and the 24-hour economy, noise will remain a major problem: Because the noise emission per vehicle has decreased during the last decades, the transport-related noise levels have declined slightly in the recent years (RIVM, 2002). But due to the ongoing growth of traffic the noise levels are expected to rise again in the next decades (Staatsen et al., 2004).

Long term noise exposure is associated with a number of effects on health and well-being. These include community responses such as annoyance, sleep disturbance, disturbance of daily activities and performance, and physiological effects such as hearing loss, hypertension and ischemic heart disease (Berglund et al., 1999). Although there is much discussion about how noise can affect human health, it is hypothesised that stress plays an important role. A model of the Dutch Health Council points out the complexity between noise and health (HCN, 1999) (Figure 1). The model assumes that health status is determined by a combination of endogenous and exogenous factors such as the physical and social environment and life style. Noise exposure is only one of these exogenous factors. This process may be modified by personal characteristics such as attitude and coping style. Noise exposure induces disturbance of sleep and daily activities, annoyance and stress, which may lead to all sorts of intermediate responses, such as hypertension. In turn, these may affect the risk on cardiovascular disease or psychiatric disorders.

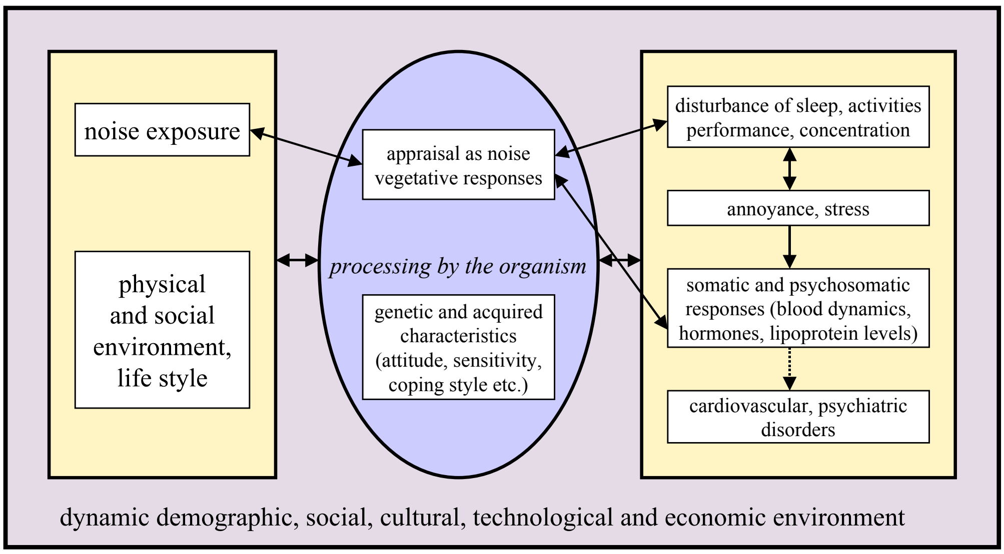

Figure 1. Conceptual model on noise and health (Source: HCN, 1999).

In the end this report provides guidance for the assessment of the impacts of noise exposure on public health. Section 2 addresses which exposure-effect-relationships are relevant to evaluate: exposure-effect-relationships for which the evidence for an association with noise was at least 'limited' and that were derived by means of meta-analytic techniques. These relationships will be presented and evaluated in section 3. Their applicability for health impact assessment is demonstrated in case-studies. After a discussion, some key messages and recommendations will be presented in section 4.

2 Methods

2.1 Identification of effects

Based on recent overviews regarding the effects of environmental noise, a set of health endpoints that are reported in relation to noise exposure, were identified (HCN, 1994), (Porter et al., 1998), (HCN, 1999), (Van Kempen et al., 2002), (Staatsen et al., 2004) (Van Kamp et al., 2004):

- direct masking effects (e.g. speech interference);
- behavioural responses such as coping strategies (e.g. closing of windows) and complaints;
- ‘social’ responses such as annoyance or perceived sleep disturbance;
- acute physiological responses (endocrine and neurophysiological reactions, such as transient blood pressure increases and sleep stage changes);
- cognitive responses such as task interference, effects on children’s learning;
- chronic physiological responses e.g. hypertension;
- clinical morbidity e.g. mental health, cardiovascular diseases, immune system deficiencies, teratogenic effects and hearing loss.

2.2 Selection of effects

Some reviews have focused on the evidence provided by the results of epidemiological environmental noise studies. In order to assess the degree of certainty concerning the relationship between exposure to noise and a particular effect, the available evidence in these overviews was rated in terms of the categories proposed by the International Agency of the Research on Cancer (IARC) as ‘sufficient’, ‘limited’, ‘inadequate’ or ‘lacking’[2]. For this report we were only interested in the effects for which the evidence for an association with community noise exposure is ‘sufficient’ according to recent reviews (HCN, 1994) (Porter et al., 1998) (Staatsen et al., 2004) (Van Kamp et al., 2004) and which are likely to occur at typical levels of community noise. According to these overviews there is sufficient evidence that noise causes annoyance and sleep disturbance in adults and has impacts on children’s learning. The evidence for an association between noise exposure and cardiovascular impacts is inconclusive. Some reviewers claimed that there is ‘sufficient’ evidence for: (i) the relationship between noise and hypertension (HCN, 1994); (ii) a causal association between

[2] ‘Sufficient’: a relationship has been observed between noise exposure and a specific health effect, chance, bias, and confounding factors can be ruled out with reasonable confidence; ‘Limited’: an association has been observed between noise exposure and a specific health effect, chance, bias, and confounding factors cannot be ruled out with reasonable confidence; ‘Inadequate’: the available studies are of insufficient quality, lack the consistency or statistical power to permit a conclusion regarding the presence of absence of a causal relationship; ‘Lacking’: several adequate studies are mutually consistent in not showing a positive association between exposure and health effect.

noise exposure and ischemic heart disease (Porter et al., 1998); (iii) an association between ambient noise and ischemic heart disease (Porter et al., 1998). Others state that there is ‘limited’ evidence for associations between noise and blood pressure changes, hypertension, angina pectoris and myocardial infarction (Staatsen et al., 2004). Because several reviewers indicated the evidence as ‘sufficient’ we decided to include cardiovascular disease effects into our evaluation.

Although there is ‘sufficient’ evidence for an effect on hearing, it is unlikely that hearing damage occurs at typical levels of community noise exposure. Therefore the effects on hearing were not dealt with in this report.

2.3 Selection and evaluation of exposure-effect relationships

For each of the selected effects we tried to identify the exposure-effect-relationships that are known up to today, using data published in the epidemiological literature. Exposure-effect-relationships can be derived either from single studies, a quantitative summary of published data (meta-analysis), a re-analysis of individual data based on primary studies (a pooled analysis) or a prospectively planned, pooled analysis of several studies, where pooling is already part of the protocol (Blettner et al., 1999). For this report we were only interested in the exposure-effect-relationships that were derived either by means of a quantitative summary of published data or an analysis of individual data based on primary studies (afterwards or prospectively planned). In case of good study quality, the exposure-effect-relationships of single studies were also evaluated.

We evaluated the exposure-effect relationships in a more or less systematic way. The following factors were evaluated: how were the relationships derived, what were the characteristics of the underlying data (design, when and where were the data collected, population characteristics, exposure characterisation, outcome), which statistics were applied and can be said something about the shape. The applicability of the curves was demonstrated by means of case-studies. Where relevant, we presented the curves derived from single studies.

Evaluation of exposure-effect-relationships is only one of the aspects that can be used to answer questions regarding causality (‘is there any other way of explaining the set of facts before us; is there any other answer equally or more likely than cause and effect’). Next to exposure-effect-relationships, the evidence on the strength of the association, its temporality, biological plausibility, coherence, consistency are also important for causality and thus for the assessment of the validity of epidemiological studies for purposes of health impact assessment. In order to get a better feeling of the other causality criteria, we evaluated the causality of the relationships by looking at the underlying studies that assess the impact of noise exposure on the different effects. We looked at the design used (e.g. cross-sectional studies, ecological studies), the characterisation and metric of exposure, the operationalisation of the outcome, the populations under study, alternative explanations for the observed associations in the studies (chance, bias and confounding). As part of this we also looked at the biologic plausibility. How does the exposure-effect relationship relate to what is known about the biological mechanism ?

3 Results

Before evaluating the relationships between noise and health, we evaluated the causality of the relationships by looking at the underlying studies that assess the impact of noise exposure on the different effects. Subsequently, the exposure-effect relationships were evaluated. The applicability of these curves in health impact assessment were demonstrated by means of case-studies.

More information regarding the operationalisation of the exposure and effect metrics can be found in appendix I and II. Characteristics and formulas of the exposure-effect relationships of the selected effects are presented in appendix III to V.

3.1 Annoyance

It is generally accepted that annoyance is the major effect of environmental noise. Annoyance is a negative evaluation of environmental conditions and 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’ ((Lindvall and Radford, 1973) (Koelega, 1987) (in: WHO, 2000)) (ISO, 2001).

3.1.1 Studies investigating annoyance

The association between noise exposure and annoyance is usually investigated by means of surveys or cross-sectional studies: estimated, yearly averaged noise levels from several sources (air traffic, road traffic, rail traffic, industry) are linked with the annoyance people perceive during a certain period. Sometimes researchers are able to investigate the effect of a new runway or highway or noise abatement measures on peoples’ annoyance in before-after studies (‘natural experiments’). Most studies investigate the effects in adults (older than 18 yrs); in some cases the participants are children (see also section 3.1.3). The results of these studies are rather consistent; most studies find a positive association between noise and annoyance.

3.1.2 Factors affecting the association with noise

The annoyance literature shows that 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. 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 (Guski, 1999) (Job, 1999) (Stallen, 1999) (Van Kamp et al., 2004). Examples of non-acoustical factors are individual noise sensitivity, fear with respect to the source, attitude towards the source, perceived control over the situation, and perceived economic or societal advantages of noise generating activity. It appears that these personal, social, and cultural factors explain about one third of the observed variance. From these, anxiety (fear of the noise source) and noise sensitivity are the most important non-acoustical factors of influence on exposure-response relationships (Fields, 1993) (Miedema and Vos, 1998) (Guski, 1999) (Job, 1999) (Stallen, 1999).

3.1.3 Annoyance measured in specific groups: children

While annoyance is one of the most frequently studied noise effects in adults, children's annoyance with noise sources appears to be an under researched area. Until now, only a few studies looked at it: the Munich Airport study (Evans et al., 1995) (Evans et al., 1998) (Hygge et al., 2002), the Heathrow studies (Haines et al., 2001a) (Haines et al., 2001b) (Haines et al., 2001c), the Tyrol study (Lercher et al., 2000) (Lercher et al., 2002) and the RANCH-study (Van Kamp et al., 2003) (Stansfeld et al., 2005). In these studies children were consistently found to be annoyed by chronic noise exposure. Most studies focus on aircraft noise.
A recent study (Haines et al., 2003) found indications that child noise annoyance is the same construct as adult noise annoyance: the emotional response of children to describing the annoyance reaction was consistent with adult reactions. Some see it as an affective response that points to a chronic decline in well being. For both parents and teachers steeper exposure-response curves were observed than for children (Van Kamp et al., 2003) (Lercher, 2002). Recently, Boman and Enmarker found that teachers were more annoyed than their pupils. It appeared that the teachers perceived the noise to be more unpredictable than the pupils (Boman and Enmarker, 2004).

3.1.4 Available exposure-effect-relationships

The first generalised exposure-effect-relationship for annoyance was published by Schultz (1978). To this end the data of 11 studies from the UK, France, Germany, Sweden, Switzerland and the USA, published between 1961 and 1972, investigating the effects of road, rail and air traffic noise were pooled. For this analysis, studies were only included in case it was possible a) to translate the reported noise levels into day-night average A-weighted sound levels (L_{DN}) in a reliable way and b) to make a consistent choice to who were 'highly annoyed'. In this way 161 data-points were generated, on which a curve was fitted by means of the least-squares procedure. The resulting curve had the shape of a third order polynomial (see Figure 2). It is important to note that the relationship does not apply to prediction of annoyance from high energy impulsive noise exposure and that the function should not be used outside of the range 45 dB(A) < L_{DN} < 85 dB(A).
In 1991, Fidell et al presented an update of the Schultz-curve. Since the publication of this curve, some 15 new studies were published. In order to increase the uniformity/homogeneity,

the participating studies had to meet the following criteria: a) at least one questionnaire item had to inquire directly about long-term annoyance per se, rather than activity interference or other noise effects from which annoyance might arguably be inferred; b) the noise source under study had to be a transportation noise source, and actual acoustic measurements of noise exposure were strongly preferred; c) the reported noise levels, if not reported in units of day-night (L_{DN}) average sound level, had to be convertible into such units with reasonable confidence; d) sample sizes had to be adequate for estimating the prevalence of annoyance with reasonable precision; and e) the scale used for quantification of annoyance had to permit numbers of respondents describing themselves as 'highly annoyed' (Fidell et al., 1991). Eventually 27 studies (453 data-points) were included resulting in a quadratic curve (see figure 2). Like Schultz, Fidell used the least-squares procedure to fit the data-points (Fidell et al., 1991).
In 1994, Finegold and colleagues decided to re-analyse the data-set of Fidell. Additional to the five inclusion criteria that Fidell used, they applied an extra criterion: whether or not a significant correlation exists between the day-night average sound levels and the related population annoyance ratings (Finegold et al., 1994). As a consequence they decided to exclude 6 datasets (53 data-points) that did not find a significant association between L_{DN} and annoyance. Again the least-squares procedure was used to fit the data-points. The curve of Finegold was adopted by the U.S. Federal Interagency Committee on Noise for use by federal agencies in aircraft noise-related environmental impact analyses and was recommended for predicting the effects of general transportation noise on people.

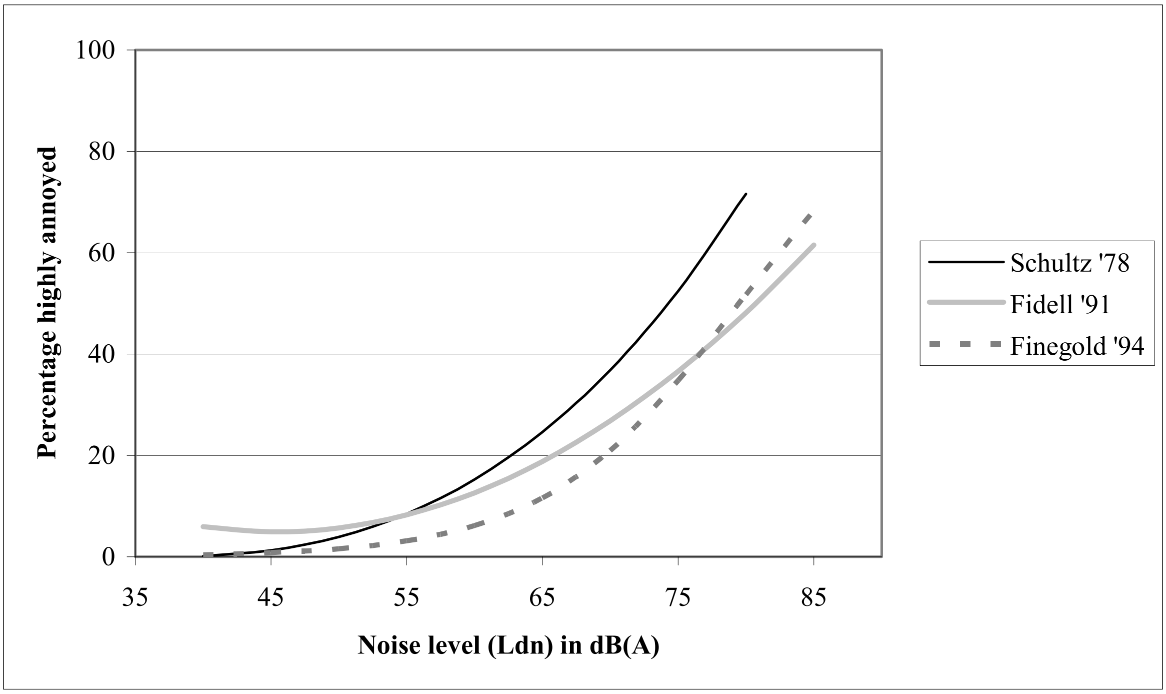

Figure 2. Exposure-effect-relationships for the relation between noise exposure and annoyance derived by Schultz (1978), Fidell (1991) and Finegold (1994).

Contrary to the curves that were presented until that time, Miedema and Vos (1998) decided to present source-specific curves. Kryter (1982) already showed that noise from urban street

and road traffic caused less annoyance than the noise from aircrafts with both equal level of L_{DN}. For the relationship between air traffic noise (L_{DN}) and severe annoyance by air traffic noise, Miedema and Vos (1998) made use of the data of 20 surveys
(including 34,214 respondents) published between 1965 and 1992 from different European countries, Australia, USA and Canada; for road traffic noise 21,228 data-points, derived from 26 studies (period 1971-1994 from different European countries and Canada) were available. For railway noise only 9 studies were available. These were published in France, Germany, the Netherlands, Sweden and the UK between 1972 and 1993. To be included into the analyses, acceptable L_{DN} and percentage highly annoyed had to be derived. Because there was no practical need for information concerning the annoyance at extreme levels
($L_{DN} < 45$ dB(A) or $L_{DN} > 75$ dB(A)), these were excluded from the analyses. At these levels the assessment of noise exposure and/or annoyance is relatively inaccurate (Miedema and Vos, 1998). In order to derive exposure-effect-relationships, the authors made use of both the ordinary least squares regression and multilevel procedure. Later, Miedema and Oudshoorn (2001) re-analysed the data, using a model of the relationship between exposure and annoyance that was more sophisticated and better suited for the data. Almost the same data were included: For aircraft noise 19 studies (27,081 data points), for road traffic noise 26 studies (19,172 data points) and for railway traffic 8 studies (7,632 points). Again, the multilevel procedure was used. In addition to the relationships between L_{DN} and annoyance, relationships that use another noise metric (the L_{den}) and other degrees of annoyance (% annoyed and the % a little annoyed) were presented.

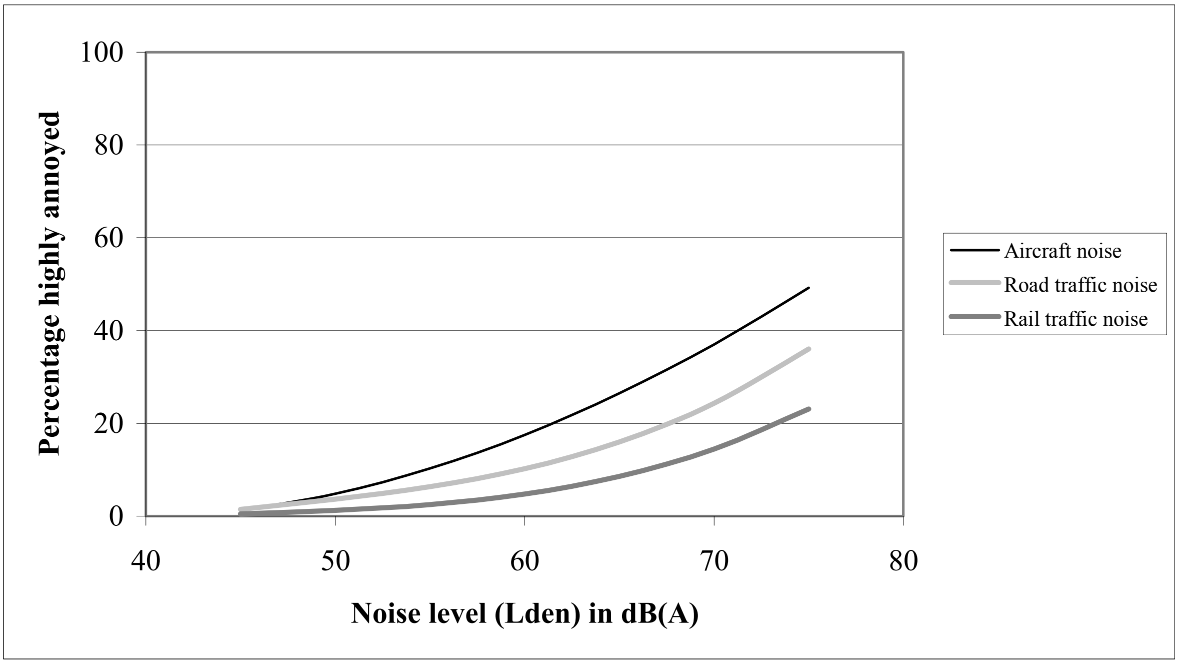

Figure 3. Exposure-effect-relationships for the association between noise (expressed as L_{den}) from different sources and annoyance derived by Miedema and Oudshoorn (2001).

Miedema and Oudshoorn were also the first to present 95% confidence intervals. Although no simple formulas are available, in their article they clearly explained how these intervals can be derived (Miedema and Oudshoorn, 2001). The exact formulas for the relationships that have been found, involved the formula for a normal distribution. Unfortunately, the covariance matrices, which are essential for calculating the intervals, were not presented. However, one has to keep in mind that the 95% confidence interval that was presented in the article of Miedema and Oudshoorn describes the uncertainty in the line (2001). This is different from the uncertainty in the underlying model.[3]

Although the Miedema-curves have been derived from probably the most extensive international database currently available, methodological differences in the original studies may have influenced the observed relationships. For example: the selected studies did not adjust for possible modifiers such as insulation, noise sensitivity and situational factors. Although we already know how some of these factors affect the association between noise and annoyance, the influence of some of these factors needs closer examination through additional research. Furthermore, some of the studies included in the Miedema curves were rather outdated.

Another comment is that rather broad inclusion criteria were applied: studies were only included if the reported noise levels could be translated into day-night average A-weighted sound levels in a reliable way and if a consistent choice about the percentage of 'highly annoyed' could be made. More specific inclusion and exclusion criteria would be desirable and not only with respect to exposure and outcome, but also with respect to the context of the study. When comparing the analyses of Schultz, Fidell and Miedema, it appeared that 13 data sets were excluded from Miedemas' analyses that were originally included in the Schultz and Fidell analyses, without explanation.

3.1.5 Approaches to assess the number of annoyed people in the Netherlands

At the moment the fraction of annoyed people in the Netherlands is assessed in two ways: (i) directly, on the basis of survey data, or (ii) on the basis of generalised exposure-effect relationships.

Ad (i). The fraction of (severely) annoyed people is assessed directly by means of national or local surveys. In 1977, 1987, 1993, 1998 and 2003 TNO and RIVM carried out national face-to-face interviews on a representative sample of the Dutch population (persons of 18-16 years and older). The results of 1998 showed that 27% was severely annoyed by road traffic noise. For a population of 20 years and older, this means that about 3.2 million people is severely annoyed by noise of road traffic (RIVM, 2000). In 2003 it appeared that 3.7 million people of 16 years and older were severely annoyed by road traffic noise (29%) (Franssen et al., 2004).

Ad (ii). The fraction of (severely) annoyed people can also be estimated using generalised exposure-effect-relationships for the association between air-, road-, and rail traffic noise and

[3] In their article, Miedema and Oudshoorn (2001) presented a model of the distribution of noise annoyance with the mean, varying as a function of noise exposure. The confidence interval was only related to the variation of the mean, which is different from the uncertainty in the underlying modelled annoyance distribution.

annoyance, such as the curves derived by Miedema and Oudshoorn (2001) (see appendix III for the corresponding formulas of these relationships). These can be utilized for strategic assessments, in order to estimate the effects of noise on populations on annoyance. The curves have been derived for adults; they are not recommended for specific sources such as helicopters, military low-flying aircraft, train shunting noise, shipping noise or aircraft noise on the ground (EU, 2002). Furthermore, they are not applicable to local, complaint-type situations, or to the assessment of the short-term effects of a change of noise climate.
Table 1 illustrates how the Miedema-relationships can be used to assess the number of annoyed people in the Netherlands. Because the relationships are only valid in the range between 45-75 (L_{DN}), people exposed to L_{DN} < 45 dB(A) or L_{DN} > 75 dB(A) were not included. First it is necessary to obtain information on the population exposure distribution for the Netherlands. In this case this was generated by means of EMPARA: a GIS-based noise-propagation model, combining source information with population and built environment data (Dassen et al., 2001). The exposure data were combined with demographic data. Subsequently, the exposure information was combined with the corresponding relationship in the way as is shown in Table 1. In this way it was estimated that 1.8 million people are annoyed by road traffic noise and about 600,000 (500,000 – 850,000) people are severely annoyed by road traffic (Table 1).

Table 1. The percentage of people exposed to and severely annoyed by road traffic noise in the Netherlands (adults only).

Exposure category, $L_{Aeq\ 24hr}$ (dB(A))	**Average $L_{Aeq\ 24hr}$ (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

(Sources: Staatsen et al., 2004 and Knol et al., 2005)

The Miedema-relationships have to be applied with great care on local situations. This becomes clear when comparing the percentage of severely annoyed people due to aircraft noise estimated using the Miedema-curve with the percentage of severely annoyed people due to aircraft noise estimated using the results of a survey around Schiphol carried out in 2002 (Breugelmans et al., 2005). This survey was executed among approximately 6,000 persons of 18 years and older living in an area of 25 x 25 km around the airport. Annoyance was measured as part of a questionnaire, using an eleven-point scale (Breugelmans et al., 2005). Persons that scored 8, 9 or 10 were defined as severely annoyed. Using the results of this survey, it was estimated that the percentage severely annoyed persons in 2002 was 13%. This was higher than can be expected on the base of the Miedema-relationship (see also Figure 4). The formula, based on the results of the survey of 2002 can be found in the appendix.

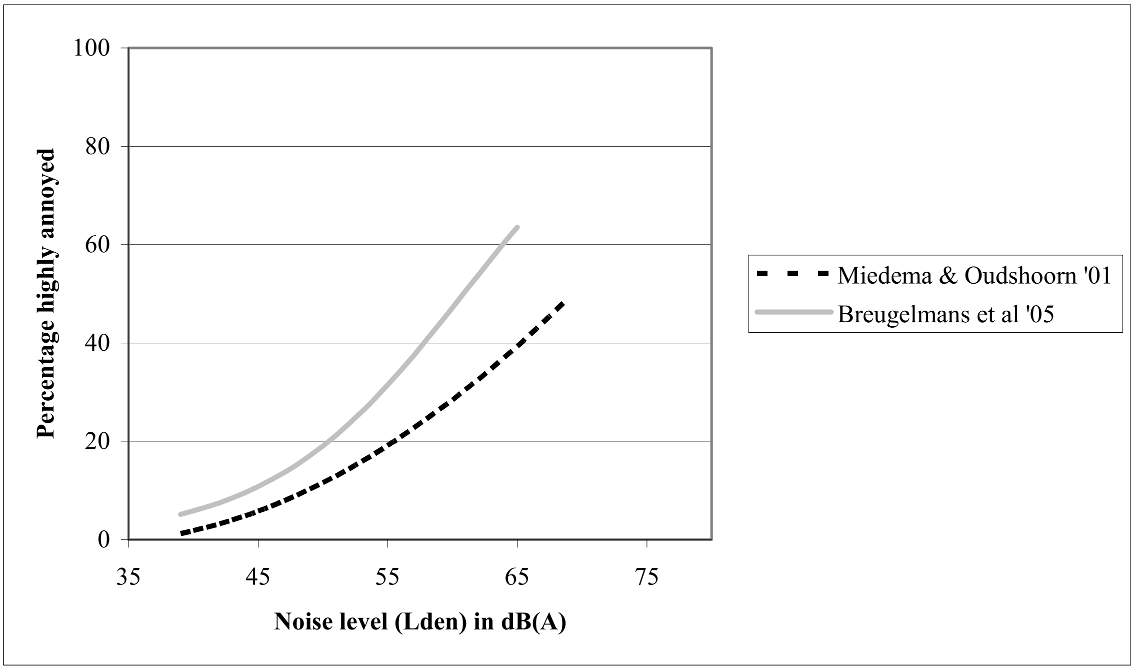

Figure 4. Comparison between the exposure-effect relationships derived in a survey around Schiphol Airport (Breugelmans et al., 2005) and the Miedema-curve (2001) for aircraft noise annoyance.

3.2 Effects on sleep

Sleep is an active physiological process and defined ‘as a state of the brain and body governed neural systems and characterized by periodic, reversible loss of consciousness; reduced sensory and motor functions linking the brain with the environment; internally generated rhythmicity; homeostatic regulation; and a restorative quality that cannot be duplicated by food, drink or drug.’(Aldrich, 1999). Several reviews have shown that night-time noise can affect people’s sleep (Lukas, 1975) (HCN, 1994) (Carter, 1996) (Porter et al., 2000). These effects may manifest itself in various ways: in the sleeping behaviour (e.g. increasing the time awake during the night), in the structure of the sleep (as measured by an Electroencephalogram, EEG), as physiological responses or as effects in the period after sleep (Van Dormolen et al., 1988). Several effects of noise on sleep, varying in severity (and evidence burden), have been measured:

- 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);
- secondary 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.

In Figure 5, a model for the different causes and effects of night-time noise exposure on sleep is presented. It shows that the relationship between community noise, sleep, behaviour and health is a rather complex one. Several research teams have tried to get more insight into this matter. The general picture of these attempts is that sleep disturbance is seen as an intermediate effect: It is assumed to be an initiator of diseases and/or it aggravates existing disease. Whether this will happen depends on the person’s vulnerability and/or sensitivity (Cohen et al., 1986) (Berglund et al., 1999) (Van Kamp et al., 2004) (Staatsen et al., 2004).

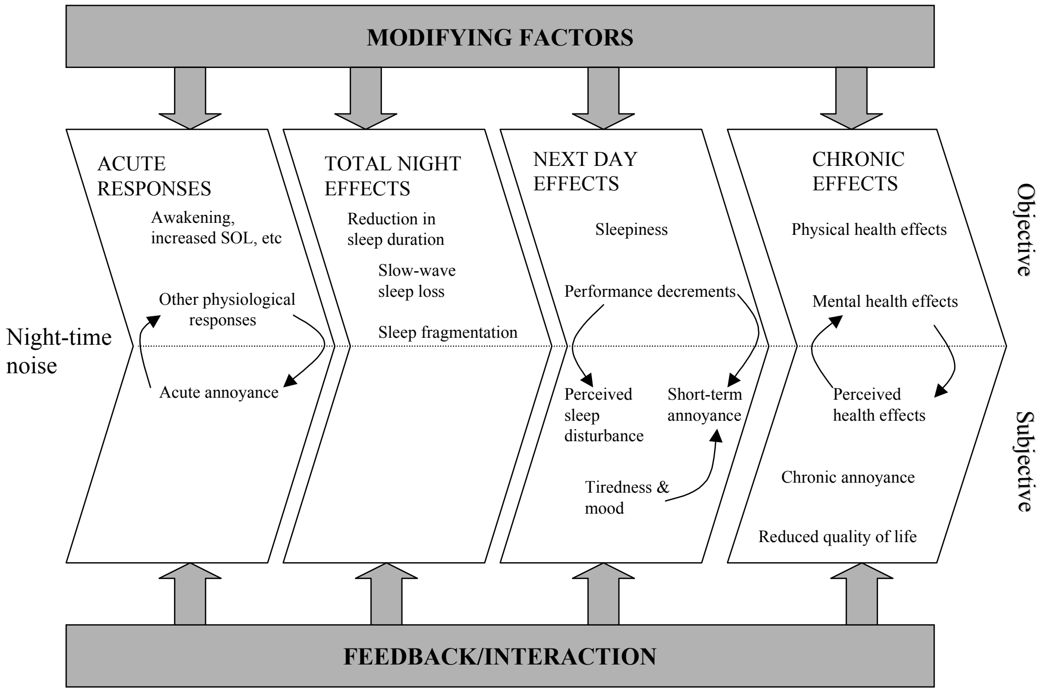

Figure 5. The potential impact of night-time noise: a model framework (Porter et al., 2000).

3.2.1 Studies investigating the effects on sleep

The effect of noise exposure on people's sleep is mainly investigated in laboratory studies. Field studies were carried out to a lesser degree and involved cross-sectional studies and intervention studies (investigating the effect of noise reductions or increases in noise). These studies were primarily focussed on the effects of transport related sources (air, road –and rail traffic). Little research has been conducted into sleep disturbance from localised sources such as factories, firing ranges, shunting yards, wind turbines, climate control systems, building or demolition work (HCN, 2004). Roughly, we can distinguish three types of studies (see also Figure 5):

- studies (primarily laboratory studies) investigating the reactions on noise events (e.g. an over-flight);
- studies (both field and laboratory studies) investigating the effects before, during and after a night of sleep of mainly road -and air traffic noise. These studies investigate how night-time noise during a sleep period affects the duration of the different sleep stages, sleep quality, awakening and getting to sleep, and the mood and performance the next day;
- field studies investigating the effects of long-term noise exposure on health and well-being: decreased sleep quality, sleep disturbance, health complaints, the use of sleeping pills and sleepiness.

More details can be found in the Appendices (I and II).

In most cases the study population consisted of adults; some studies investigated the effects on elderly (> 60 yrs) and children.

3.2.2 Factors affecting the association with sleep disturbance

Age, sex, season, annoyance from other environmental factors, medical condition and medication are important factors of influence with regard to the level of sleep disturbance (Staatsen et al., 2004). Personal factors of assumed importance are anxiety and noise sensitivity. Social factors that play a role are the attitude and expectancies regarding possible changes of the source (e.g. growth of the airport, or increase of noise levels, number of flights) (Job, 1999) (Stallen, 1999).

3.2.3 Effects in specific groups: children and the elderly

Until 1991 only 5 studies were carried out investigating the effects in elderly (> 65 yrs) (Hoffman, 1994). Effects under investigation were sleep latency, awakenings, sleep stage changes, sleep structure, heart rate, sleep quality, mood and sleep disturbance.
A recent report of the Dutch Health Council showed that the number of studies among children regarding the relation between noise and sleep is limited (HCN, 2004). To our knowledge there are only three studies available investigating the effects of night-time noise on children's EEG. These laboratory studies involved only a small number of respondents (Muzet et al., 1980) (Busby and Pivik, 1985) (Eberhardt, 1990). From these studies it can be concluded that children seem to have better perceived sleep quality than adults. Sleep impairment in children seems to start at higher noise levels than in adults (Öhrström et al., 2003). On the one hand young children are less prone to awakenings due to noise than adults (Lukas, 1972), but on the other hand the autonomic nerve system of children is more easily activated during their sleep (Semczuk, 1967) and children seem to have higher cardiovascular responses than adults (Muzet et al., 1980).

3.2.4 Available exposure-effect-relationships

For several combinations of exposures (expressed in several metrics) and outcomes, exposure-effect-relationships have been derived. In most cases, laboratory studies were included. In comparison to field studies, these find a stronger association with noise. One of the possible explanations for this difference is habituation, which cannot be accounted for in the laboratory (Berglund et al. 1999). Inside both the group of field and laboratory studies there are large discrepancies which account for a very large variability of the results. Because the aim of this report is to select exposure-effect-relationships that can be used in health impact assessment, only the most recent curves based on field studies are presented here. An overview of all the available exposure-effect-relationships is presented in Appendix IV.
Analogue to the exposure-effect-relationships derived for the association between road, rail and air traffic noise and annoyance, Miedema et al. (2003) carried out a re-analysis of individual data for sleep disturbance. Included were: (i) studies where L_{night} was included in

the data-set or the probability to calculate/estimate this metric on the basis of information regarding the included sites; and (ii) studies using questions regarding waking up or being disturbed by noise during the night. Studies using questions regarding disturbance of sleep or resting were excluded. In the opinion of Miedema and colleagues (2003), resting is different from sleeping and does not need to take place during the night only. Furthermore, low exposure levels (L_{night} < 45 dB(A)) were excluded from the analyses because, according to Miedema et al. (2003), in general, the assessment of those noise levels is relatively inaccurate and in situations with these low levels, other sources may be more important. High exposure levels (L_{night} > 65 dB(A)) were also excluded, because in areas with very high exposure levels there is a relatively high risk of self-selection of persons not bothered by noise. However, data dealing with this hypothesis are lacking. Eventually, 11 European studies, 2 Canadian and a Japanese study from the period 1975 – 2001 (8,459 subjects) for road traffic noise and 6 European studies and a Japanese study (period 1983-2001) (4,098 subjects) for rail traffic noise were included in the analysis. In order to derive exposure-effect-relationships for sleep disturbance, the same statistical model was used that was already developed for the analysis of the relationship between noise exposure and noise annoyance (Miedema and Oudshoorn, 2001). The relationships give the percentage highly sleep disturbed (*%HSD*), sleep disturbed (*%SD*), and (at least) a little sleep disturbed (*%LSD*) by road and railway noise as a function of the outdoor L_{night} at the most exposed façade (Miedema et al., 2003). Because the estimated variance of the normal distribution of the sleep disturbance scores was very high for aircraft, no exposure-effect-relationships for aircraft noise were presented. In a follow-up analysis, Miedema and Oudshoorn (2004) investigated whether there were reasons not to include particular data-sets used in the earlier analysis. As a result the researchers decided to include an American and a British study (DORA, 1967) (Hazard, 1971) that were not used in the earlier analysis. Two new studies with data on L_{night} and self-reported sleep disturbance were also included (DORA, 1980) (Wirth et al., 2004). A total of 8 studies (one American and 7 European studies) from the period 1967 to 2004 were now included for the analysis. Eventually, two curves were presented: a curve with and a curve without study effect. Because, this has also been included in the analyses for road traffic and railways, the researchers recommended to use the model that incorporated a study effect.

When using the exposure-effect-relationships for sleep disturbance, we have to take into account that these curves are not adjusted for other factors. Therefore, further verification of these relationships is needed with attention to the construction of dwellings (insulation, position of the bedroom) and other use of windows. The 95% confidence interval describing the uncertainty of the line can be estimated in a way similar to the exposure-effect-relationships for annoyance. Because Miedema (2003) worked with distributions, no simple formulas were available.

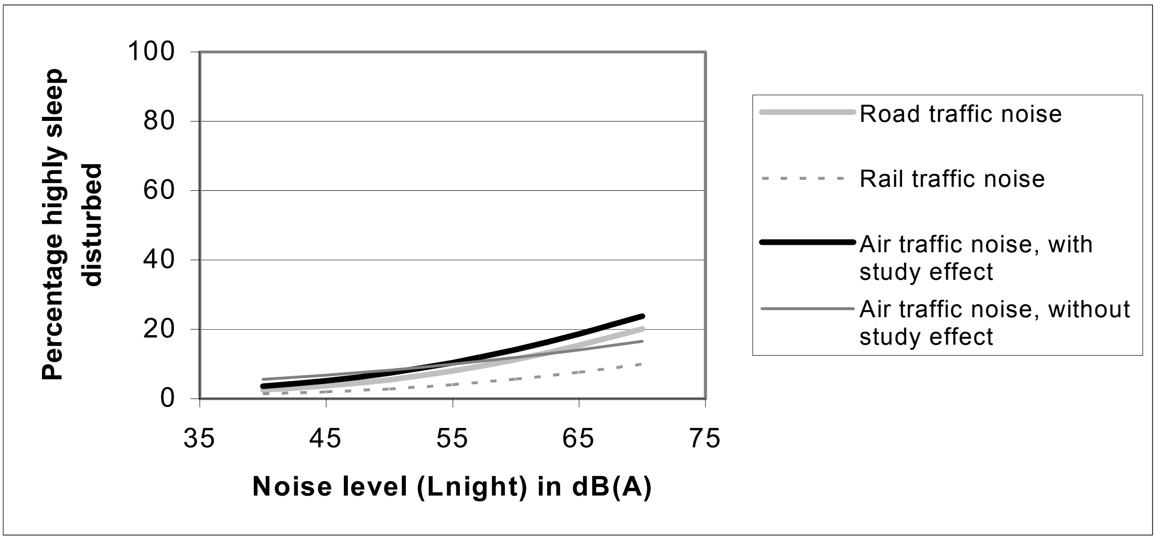

Figure 6. Exposure-effect-relationships between night-time noise exposure (L_{night} , outside at most exposed façade) and self-reported sleep disturbance for exposure to road, rail, and air traffic noise, derived by Miedema et al. (2003) and Miedema and Oudshoorn (2004).

Apart from sleep disturbance, the Working Group on Health and Socio-Economic Aspects (WGHSEA) also recommended relationships for a) awakenings and b) instantaneous and mean motility (Passchier-Vermeer, 2002) (Miedema et al., 2003) (Passchier-Vermeer, 2003) (WGHSEA, 2003).

Ad a) The relationship describing the association between aircraft noise events during the night and behavioural awakenings was derived by means of a meta-analysis. To this end 110 aggregated data points, derived from eight field studies were pooled by means of a regression analysis. The participating studies were carried out in Europe (France, the United Kingdom and the Netherlands) and in the USA between 1973 – 2002 (Passchier-Vermeer, 2003). According to Passchier this relationship is applicable to the general population exposed to commercial aircraft noise events during night-time. However, a few comments can be made: Whether a person awakes from an aircraft noise event depends on the noise level of that event. A L_{night} can be calculated by combining the SELs of the noise events that took place in the period between 23 and 7 hr. This means that many noise events with a low noise level can lead to the same value for L_{night} as a few very loud noise events. At a given value for L_{night} a person awakes more often in the case of many noise events with low Sound Exposure Levels compared to as situation with only a few noise events with high sound exposure levels. Furthermore, one should take into account that people awake spontaneously: According to the Dutch Health Council this 1.5 to 2 times a night (average); next to this, the level of consciousness can be so high that an aircraft event can be heard (this happens about 10-12 times a night) (HCN, 2004). When more noise events occur during the night, the chance that a person hears such an event when he/she is awake is higher. A complicating factor is that until now, studies only investigated the effects at individual level. At a certain level of L_{night} the individual chance that a person awakes due to aircraft noise events is smaller in case there are only a few events with high sound levels. It is unknown how this works on population level. This makes that the relationship is not applicable yet. A recent German study does an attempt to quantify the effect on population level (Basner et al., 2004) (Fast, 2004).

Ad b) The relationships describing the association between noise and motility were based on actimetry[4] data of a Dutch study that was carried out around Schiphol Airport. Participants were 418 subjects (aged 18-81 years), who were measured during 11 nights (63242 aircraft noise events). The probability of motility has been considered at the 15-s interval at which the maximal indoor equivalent sound level in a 1-sec interval during an aircraft noise event occurs. Using the data of this study, relationships between aircraft noise-induced increase in probability of motility and indoor L_{max} and indoor SEL of aircraft noise events were obtained by using a random effects logistic regression model with a random subject factor.
With regard to the applicability of these motility-curves the following can be said: Since the study has sufficient power and because several shortcomings of earlier studies have been accounted for, the derived relations are applicable for the indoor situation for $L_{max\ indoor}$ -values up to 70 dB(A) or SEL_{indoor} values up to 80 dB(A). With the exclusion of children, persons with night-time shifts, severely diseased people and persons who recently started to use sleep medication, the curves are generally applicable. The relationships can not be applied in situations where the difference between night-time aircraft noise levels (L_{night}) and the equivalent aircraft noise levels for the 24hrs period differ from the differences found in the study of Passchier-Vermeer (2002). She showed that the equivalent aircraft noise levels for a period of 24 hrs affected the relationship between night-time noise exposure and motility (Passchier, 2003). An important factor influencing this relationship is the individual long-term aircraft noise exposure during sleep. As the aircraft noise levels become higher, the probability of instantaneous aircraft-noise related increase in motility is much lower (Passchier-Vermeer, 2002).

3.2.5 Approaches to assess the number of sleep disturbed people in the Netherlands

Despite the large number of available exposure-effect-relationships, only the relationships between air, road, -and rail traffic noise and sleep disturbance from Miedema (2003, 2004) are in use to assess the impact of noise exposure on people's sleep in the Netherlands. These were also recommended by the Working Group on Health and Socio-Economic Aspects to be incorporated into the European Noise Directive (END) (WGHSEA, 2003).
An example: The calculation of the fraction of sleep disturbed people was similar to the calculation of the fraction of annoyed people, applying the exposure-effect-relationships on a given population noise exposure distribution (derived by means of EMPARA). Because the relationship is only applicable for the range 45 – 65 dB(A) (L_{night}), persons outside this range

[4] Actimitry is carried out by means of a device (a kind of watch) that the respondent has to ware around his/her wrist: the actimeter which detects body movements (motoric activity, motility). Per time interval (e.g. 15 seconds) they register the time and a value that gives an indication for the strenght of the body movements during the time interval.

were not included in the calculation. Again, only adults were included. By doing this, it was estimated that 300,000 (200,000 – 450,000) (2%) people may experience severe sleep disturbance due to road traffic noise in 2000 (Staatsen et al., 2004) (Knol et al., 2005).
Similar to annoyance, the fraction of (severely) sleep disturbed people can also be assessed directly. The results of the survey of TNO/RIVM among a representative sample of the Dutch population showed that in 2003 about 1.5 million Dutch people of 16 years and older (12%) were severely sleep disturbed by road traffic noise. In 1998 8% was severely disturbed. Furthermore, it was estimated that in 2003 about 890,000 persons were severely sleep disturbed by moped noise (Franssen et al., 2004)

Like annoyance, the application of the sleep disturbance curves on local situations has to be done with great care. In the surveys around Schiphol that were mentioned already in section 3.1.5, sleep disturbance was also measured, using an eleven point scale. Persons that scored 8, 9 or 10 were defined as severely sleep disturbed (Breugelmans et al., 2005). On the base of the results of the survey carried out in 1996, it was estimated that approximately 7% (about 190,000 persons) were severely sleep disturbed. In 2002, this decreased to 5% (about 130,000) persons). In both situations, the estimated percentages were higher than was expected on the base of the curve derived by Miedema (2004) but similar to what was found on the base of the national surveys of TNO/RIVM. In 2003 and 1998 3% of the respondents were severely sleep disturbed. For 1998 it was estimated that about 180,000 persons in the Netherlands were severely sleep disturbed due to air traffic noise from Schiphol (Fast, 2004) (Breugelmans et al., 2005).

3.3 Effects on the cardiovascular system

Several literature reviews have suggested that noise exposure is associated with blood pressure changes and ischemic heart disease risk (HCN, 1994) (Morrell et al., 1997) (Babisch, 1998) (Porter et al., 1998) (Van Kempen et al., 2002). A broad range of effects on the cardiovascular system related to exposure to noise, have been reported: (i) systolic and diastolic blood pressure differences (expressed in mmHg); (ii) changes in the occurrence (prevalence, incidence) of hypertension, myocardial infarction and angina pectoris, and (iii) changes in the number of hospital admissions and/or mortality due to cardiovascular disease. Usually the effects that were found are small; transient stress-related hemodynamic responses that are harmless on an individual level may result in slight shifts in blood pressure on population level. In a smaller, susceptible proportion of the population this may lead to increases in hypertension and, eventually, prevalence of ischemic heart disease, including angina pectoris and myocardial Infarction.

Although very complex, the biologic mechanism of noise exposure leading to cardiovascular effects seems plausible. The literature has suggested that noise-induced cardiovascular effects must be seen as the consequence of stress, which can arise in several ways in relation to noise. In experimental studies, investigating the short-term of noise exposure, acute biochemical, physiologic, and cardiovascular changes have been found, which mark a common physiologic stress reaction of short duration that occurs as a consequence of the activation of the autonomous nervous system and hormone system. Another possibility is that the effect of noise on the auditory system is transmitted to the Reticular Arousal System (RAS) and the hypothalamus, where both neuronal and hormonal activity may be activated. Stress can also be the consequence of noise appraisal (Van Kempen et al., 2002).

3.3.1 Studies investigating the effects on the cardiovascular system

The field studies investigating the impact of noise on the cardiovascular system were mainly cross-sectional. We can distinguish studies investigating the effects of occupational noise exposure and studies investigating the effects of community noise exposure (road, rail and air traffic noise). The occupational studies were performed among a great variety of industries throughout the world within a broad exposure range. Most studies were carried out among adults (Van Kempen et al., 2002). For more information see also Appendix II and III.

3.3.2 Factors affecting the association with noise

When investigating chronic diseases in cross-sectional studies there is the problem of self-selection in community studies and the healthy worker effect in occupational studies. In community studies, somewhat sensitive subjects may move out of the polluted areas, diluting the effect of interest. In occupational studies, subjects may leave the job because of cardiovascular disease due to noise or because of the noise itself. These effects tend to diminish the magnitude of the effects found in studies (Babisch, 1998).

It might be possible that physiological effects as blood pressure, are not the result of the noise exposure itself, but that psychological factors might also play a role. This assumption is not new: Until now several studies have looked into the impact of variables on the relation between noise and health, that are assumed to be connected with the processes that determine whether environmental sounds are noise. The basic assumption of these studies were stress models assuming that the presence of noise is not always enough to explain the occurrence of health effects. The appraisal of the noise by the individual is also important. This process of appraisal could have affected the occurrence of health effects as blood pressure elevations and symptoms.

Another explanation for the weak association between noise exposure and health might be that the effects of noise exposure especially appear in risk groups. Indications for this assumption come from some recent studies: In a study among adults, investigating the effects of road traffic noise on ischemic heart disease, Babisch and colleagues found that the association between road traffic noise exposure and ischemic heart disease was modified by pre-existent disease (Babisch et al., 2000).

3.3.3 Specific groups: children

Only a few studies investigated the effects of (military) air traffic, road traffic and rail traffic noise on blood pressure and heart rate in children aged 3-17 years. These included mainly cross-sectional studies, and 2 follow-up studies with sample sizes varying from approximately 100 to 1,500 children. More characteristics of these studies are presented in Appendix VI. Until the Eighties the results of the studies that have investigated the effects of aircraft, road traffic and rail traffic noise exposure on systolic and diastolic blood pressure and heart rate in children are difficult to interpret, since limited quantitative data were presented. In these studies, very crude data regarding more blood pressure abnormalities in children living in the vicinity of airports or attending schools in areas with high noise exposure, were reported. The results of the later studies were rather inconsistent: although often an association with systolic blood pressure was found, the results for diastolic blood pressure and heart rate were contradictory. The conclusions that can be drawn from these studies are limited, because of a number of methodological problems (e.g. small differences in noise levels between the exposure groups, potential selection bias, a lack of control for socio-economic status factors, insulation and parental history of high blood pressure).

3.3.4 Available exposure-effect-relationships

In 1993, Passchier-Vermeer published the results of a systematic review evaluating 21 occupational studies and some community noise studies. After analysing the data, increases of the mean systolic and diastolic blood pressure of 3.9 and 1.7 mmHg, respectively, were observed for persons in exposed groups compared to persons in reference groups. Also a significant increase in the risk of hypertension was found: a relative risk (RR) of 1.7 for noise levels exceeding 85 dB(A). The observation threshold for hypertension was estimated to correspond to an L_{DN} value of 70 dB(A) for environmental noise exposure (Passchier-Vermeer and Passchier, 2000).

The second meta-analysis was carried out by Duncan et al. (1993). They found an increase in the odds of developing hypertension as a function of increasing noise levels above 20 KE. However, in this meta-analysis the effect of different exposure sources were combined.

To gain more insight into the potential health impact of noise exposure, a meta-analysis of 43 epidemiological studies published between 1970-1999 and investigating the relation between noise exposure (both occupational and community), blood pressure and/or ischemic heart disease (ICD-9: 410-414) was conducted in 2000. A wide range of effects, varying from blood pressure changes to a myocardial infarction, was studied. Quantitative summaries were obtained by means of a random effects model. Only estimates from studies adjusting for at least age and gender were included into the analysis. Because it was not possible to indicate the shape of the curve and a threshold value on the base on the available data, it was decided to use two models for the meta-analysis: an exponential and an additive model. The latter assumes that the increase in prevalence per unit of noise is constant, while the first assumes a constant relative risk (RR) per unit of noise (in other words the relation between the exposure and the prevalence of the effect concerned is exponential). Eventually, both models seemed to fit the data (Van Kempen et al., 2002). With respect to the association between noise exposure and blood pressure, small blood pressure differences were noticed. A significant association for both occupational noise exposure and air traffic noise exposure and hypertension was recorded: RRs of 1.14 (1.01 – 1.29) and 1.26 (1.14 – 1.39) per 5 dB(A) noise increase were estimated, respectively. Air traffic noise exposure was positively associated with the consultation of a GP or specialist, the use of cardiovascular medicines and angina pectoris. In cross-sectional studies, road traffic noise exposure increases the risk of myocardial infarction and total ischemic heart disease (see also Table 1 of Appendix V) (see also Figure 7).

The study of Van Kempen included studies carried out in the period 1970-1999. However, in the period 2000 until now the results of new community noise studies investigating the effects of road traffic, air traffic and rail traffic noise on cardiovascular disease have come out. The conclusions from these studies did not really differ from what is already found in the published reviews on this topic. New is that now also 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). A 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).

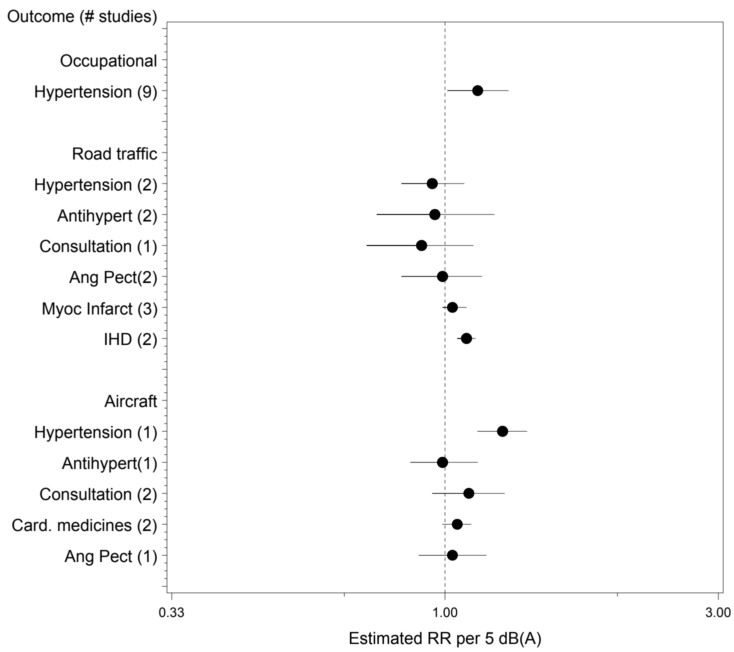

Figure 7. Summary estimates, expressed as relative risks (RR) per 5 dB(A), for the association between noise exposure, hypertension, and ischemic heart diseases, adjusted for gender and age. The black circle and the horizontal line correspond to the estimated $RR_{5\ dB(A)}$ and 95% confidence interval. The dotted vertical line corresponds to no effect of noise exposure (Source: Van Kempen et al., 2002).

3.3.5 An approach to estimate cardiovascular disease attributable to noise

On the ground of the relationships derived by Van Kempen, some estimates were available for the risk on cardiovascular disease attributable to noise from air- and road traffic among adults (Staatsen et al., 2004) (Knol et al., 2005). When doing this, it was assumed that 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 1), it was assumed 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, the annual hypertension mortality that may be attributed to noise exposure (population attributable risk or PAR) was calculated. No distinction was made 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, PARs were calculated by combining the exposure population distribution with quantitative exposure-effect 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),
$L_{cut-off}$ = cut-off or reference level,
β = the risk function estimate (per 5 dB(A))
p_i = exposure probability in class i.

The exposure-effect-relationship used, was derived from Van Kempen et al. (2002). 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). It was assumed that the relation between noise exposure and the prevalence of hypertension is exponential. Because, the studies investigated the effects on the cardiovascular system were carried out in the range between 50 – 75 dB(A), only these were included; a cut-off point of 50 dB(A) was used.
After applying equations 1 and 2, a population attributable risk of 0.06 for noise-induced hypertension was found. 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 don't experience problems in their daily functioning, this health state is normally not incorporated in the calculation of the burden of disease. Therefore the fraction of noise-related mortality attributable to hypertension (0.0043) was estimated. This was done 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 it was estimated that maximum 1,100 people may die annually due to noise attributive hypertension (Staatsen et al., 2004) (Knol et al., 2005).

3.4 Cognition

Although it has been documented in several studies that noise adversely affects cognitive performance, this paragraph is entirely focussed on children.
In children, the possible effects of noise on cognitive functioning were studied the most. In studies investigating the effects of chronic noise- exposure to air-, rail-, and road traffic, effects were found on reading, attention, problem solving and memory. In summary, the following results have been found in children exposed to high levels of environmental noise, 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). The general finding was that mainly performance on the complex tasks is affected.

With regard to cognition, not much is known about the underlying mechanisms. Only some hypotheses are known (Stansfeld et al., 2000) (François and Vallet, 2001).
One hypothesis is that at least in the school context, noise affects the intelligibility of speech communication. This can lead to difficulties in written and spoken language, and adverse effects on language development and the acquisition of reading skills. As a consequence children's reading may be impaired and their vocabulary is reduced. Disturbed speech communication may have serious repercussions on the education and intellectual development of young people. If a message is degraded, they cannot reconstitute the fragments that may be masked by the noise. Due to noise there is a loss of meaning in the content of teachers' instruction, children may have problems with the intelligibility of letters, words and even entire sentences. It has been shown that, in a noisy environment, children confuse certain consonants and that sound distortion makes certain parts of words (particularly endings) unintelligible. A hypothesis that is related to this, is the tuning out response: children may adapt to noise interference during activities by filtering out the unwanted noise stimuli. However, researchers think that this strategy may 'over-generalise' to situations where noise is not present, such that children tune out stimuli indiscriminately. It is hypothesised that mechanisms as auditory discrimination and speech perception are important mediators of the possible association between noise and performance.

3.4.1 Studies investigating the effects on cognition

Most research on noise and cognition has been carried out in primary school children, aged 5 to 12 years. The effects of mainly aircraft noise were investigated by means of cross-sectional studies or laboratory studies. Detailed information can be found in Appendix I and II. Tables 2 and 3 give an overview of the most recent studies investigating the effects of noise exposure on children's cognition. For reading ability consistent results were observed, indicating a negative association between chronic (long-term) noise exposure and reading acquisition. The results of the studies that looked at the association between noise exposure and attention deficits varied. Nearly all studies have involved a cross-sectional design, small sample sizes, and a lack of adjustment for potential confounders such as socio-economic status.

Table 2. Characteristics of recent field studies investigating the effects of noise on cognition in children.

Study[a)]	Design	# Schools	# children	Exposure Source	Noise metric	Noise range
LA-study	Cross-sectional/ 1-yr follow-up	7	262	Air	Peak noise level	High: 95 dB
Munich	Before-after study	-	326 (9-10 yrs)	Air	$L_{Aeq, 24hr}$	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 yrs)	Air	$L_{Aeq,16hr}$	High: >66 dB Low: <57 dB
WLSS	Cross-sectional	20	451 (8-9 yrs)	Air	$L_{Aeq,16hr}$	High: >63 dB Low: <57 dB
Tyrol	Cross-sectional	26	1230 (8-11 yrs)	Rail & road	L_{dn}	High: >60 dB Low: <50 dB
RANCH	Cross-sectional	89	2844 (9-10 yrs)	Air & road	$L_{Aeq, 7\text{-}23\ hr}$	Air: 30 -77 dB Road: 32 – 71 dB

a) LA-study: Los Angeles Airport study (Cohen et al., 1980), (Cohen et al., 1981) (Cohen et al., 1986); 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., 2001a) (Haines et al., 2001c); WLSS: The West London Schools Study: (Haines et al., 2001b); Tyrol: The Tyrol Study: (Lercher et al., 2000), (Lercher et al, 2002). RANCH: RANCH-study (Stansfeld et al., 2003) b) Gr 1: Noise levels of group "old airport, noise before/after removal –exposed"; Gr 2: noise levels of group 2 "old airport- not exposed"; Gr 3: Noise levels group 3 "new airport –exposed"; Gr 4: Noise levels of group 4 "new airport – not exposed".

Table 3. Overview of the results of recent field studies investigating the effects of noise on cognition in children.

Study → Outcome ↓	LA-study	Munich	SEHS	WLSS	Tyrol	RANCH air	RANCH Road
Motivation	+	+	0	NI	NI	NI	NI
Reading	0	+	+/0	+/0	NI	+	0
Long term memory	NI	+	+/0	0	NI	+	0
Working memory	NI	+	NI	0	NI	+	0
Attention	+/-	+	+	0	0	0	0

NI = Not investigated; +: a positive association was found; 0: no association was found; -: a negative association was found

3.4.2 Factors affecting the association with cognition

There is still uncertainty as to how much the observed cognitive impairments can be attributed to noise, because these cognitive tasks might also be influenced by the quality of the school and the level of social deprivation of the area in which the children live. The Environment and Health School study suggested that chronic aircraft noise exposure was associated with school performance after adjustment for school effects, but that this association might be influenced by socio-economic factors (Haines et al., 2001a) (Haines et al., 2001c). Noise exposure and social class might be inter-related and possibly act together to influence performance. On the other hand social disadvantage was also associated with low school achievement.

3.4.3 Available exposure-effect-relationships

Until now three studies have tried to derive an exposure-effect-relationship for the effects of air traffic noise exposure on reading. These involved two ecological studies (Green et al., 1982) (Haines et al., 2002) and a multi-centre study (Stansfeld et al., 2003) (Stansfeld et al., 2005). In the study of Green et al. (1982), noise exposure levels were based on noise exposure contours for New York City Airports. The outcome was the percentage of students reading below grade level (1 yr, more than 1 year, more than 2 year): to this end the aggregate results from annual nationally standardized tests of reading ability given in the New York City Public schools were obtained from the New York City Board of Education. Next to this data on racial composition, socio-economic level and various educational factors for each school were obtained, for the years 1972 to 1976. Eventually, the results of 362 schools were included. A summary coefficient of 0.62 (0.51 - 0.74) was estimated, suggesting 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. The mean difference in the percent reading one or more years below grade level in noisy school compared to the quietest schools was 3.6% (1.5 - 5.8 %).

Later, Haines et al. (2002) 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. Aircraft noise exposure was assessed by the Civil Aviation Authority dB(A) Leq 16hrs contour maps. Each school was classified into one of 8 exposure levels (<54; 54-56; 57-59;60-62; 63-65; 66-68; 69-72; > 72). Both on school and on individual level, several factors were taken into account. At school level, this concerned the percentage of pupils eligible for a free school meal, the percentage of pupils with special needs; the percentage of pupils with English as a second language and the type of school. At individual level sex, year of testing and date of birth have been taken into account. In order to take into account the hierarchical data structure, multilevel modelling was applied. The researchers found that 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.

In the RANCH-study an effect of air traffic noise exposure on reading was found among 2010 children attending 89 primary schools in airports in the UK, the Netherlands and Spain. The data were pooled using multi-level modelling (regression analysis). After adjustment for age, gender, insulation, longstanding illness, parental support, and socio-economic state, it appeared that children in schools with higher aircraft noise exposure scored statistically significant lower on the reading comprehension test than children in schools with lower aircraft noise exposure. A linear exposure-effect-relationship was found: there was no significant departure from linearity when comparing a straight line with a fractional polynomial curve. It was estimated that a 5 dB(A) increase in noise was associated with a 2-month and 1-month impairment in reading age in the UK and the Netherlands, respectively (Stansfeld et al., 2003) (Stansfeld et al., 2005).

3.4.4 Approaches to assess the number of children with impaired cognitive functioning attributable to noise

In order to assess the impact of noise on children's reading ability, the results of the RANCH-study can be used. The results of this study are robust because of the use of data from three countries with contrasting socio-demographic profiles, detailed noise assessments, extensive measurement of confounding factors (Stansfeld et al., 2003) (Stansfeld et al., 2005). A problem is that the outcome is difficult to interpret at population level. What does a '1 to 2-month impairment in reading age' mean and when is an impairment in reading age 'clinically' relevant ? Outcomes such as 'the chance that a child has a low score on his/her reading test' would be more meaningful. Before the relationship as derived in the RANCH-study is applicable for health impact assessment, greater specification is necessary.

4 Discussion

For the assessment of health impacts related to traffic noise exposure in the Netherlands exposure-effect relationships were available describing the association between noise exposure and annoyance, sleep disturbance and the cardiovascular system. For these outcomes the evidence was sufficient and they were likely to occur at typical levels of community noise. The relationships evaluated in this report, were derived either by means of a quantitative summary of published data or a re-analysis of individual data based on primary studies.

4.1 Annoyance

The relationships for the association between noise and annoyance derived by Miedema and Oudshoorn (2001) are at the moment the best currently available. They are based on a re-analysis of individual data from 45 different studies, which makes them rather unique. Recently they were recommended for use in the EU Directive on Noise (EU 2002). Their applicability was demonstrated in section 3.1.5. Differences were observed when comparing the number of annoyed people in the Netherlands estimated by means of national surveys with the number of annoyed people estimated by means of combining exposure-distributions with the Miedema relations. In order to get the exposure-distribution, the EMPARA-model is used (Dassen et al., 2001). An important explanation can be found when looking at the differences in the classification of the sources: national surveys ask for annoyance due to traffic, mopeds, or lorries separately. The EMPARA model does not calculate the noise levels for these sources separately. Other explanations are: (i) the noise exposure that is attributed to persons that participate in the national survey is estimated in a different way from what is done by EMPARA; (ii) usually, national surveys are investigating representative samples of the Dutch population (e.g. 18 years or older); EMPARA does not take age into account (Kruize and Staatsen, 1998). Therefore, where risk estimates for annoyance based on national surveys are available, this is preferred (WHO, 2003). Other issues that make the reason for this recommendation clear will be addressed in the next paragraphs.
The studies that were included in the Miedema relationships were carried out in the period 1965-1992, leaving a gap of about 15 years. In a recent publication, Guski (2004) showed on the basis of the Miedema data that there has been a trend in the last decades: the number of people that is highly annoyed increases at lower day-night levels (see also Figure 8). If it is true what Guski finds, then possible explanations for this trend might be found in the fact that the composition of aircraft noise has changed over the years: the single noise events become less loud, but the number of events increased considerably. Furthermore, sounds get their meaning in relation with other sounds. It is possible that the changing composition of sound pressure levels evokes differences in perception (Wirth, 2004).

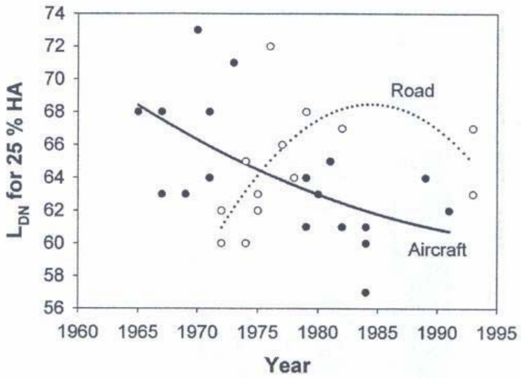

Figure 3. Noise levels for a constant proportion (25%) of highly annoyed residents. Data from Miedema and Vos (1998).

Figure 8. *Figure derived from Guski (2004) indicating a possible trend in annoyance due to aircraft noise exposure.*

The relationships derived by Miedema (2001) are only to be used for stable situations when noise levels are not changing (e.g. due to new runways): they can be utilized 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 climates. Whether this is realistic, is the question. Looking at major airports (such as Schiphol), a stable situation is hardly ever matched any more: expansions, alterations planning procedures and public contentions are more or less ongoing (Wirth and Bröer, 2004).

4.2 Sleep disturbance

Several exposure-effect-relationships were derived for the effects of several noise sources for a range of effects: awakenings, sleep stage changes, motility and sleep disturbance. From these, the relationships for the association between noise from road, rail and air traffic and sleep disturbance derived by Miedema (2003 and 2004) are best applicable for health impact assessment, for they were derived on a re-analysis of individual data from different studies. The curves describe the level of annoyance due to night-time noise, which is not the same as perceived sleep quality. However, they have to be used with great care. This is especially the case for aircraft noise: in comparison with the curves for road and rail traffic noise, the variance of the responses at a given exposure level was relatively large for aircraft noise. This means that the uncertainty regarding the curve for aircraft noise is large. Several causes are suggested: (i) the time pattern of noise exposures around different airports varies considerably due to specific night-time regulations; (ii) the sleep disturbance questions for aircraft noise show a large variation; and (iii) the most recent studies show the highest self-reported sleep disturbance at the same L_{night} level, which suggests a time trend (Miedema,

2004). Therefore the curve for aircraft noise must be considered as indicative only. Further verification of the proposed sleep disturbance curves is needed with attention to construction of the dwellings and other use of the windows (Miedema, 2003) (Staatsen et al., 2004).
When estimating the number of sleep disturbed people, smaller numbers were found than was expected on the base of national surveys (see also section 3.2.5). Explanations were already addressed in the annoyance-paragraph. As for annoyance, where risk estimates for sleep disturbance based on national/local surveys are available, this is preferable.

Next to sleep disturbance, relationships for awakenings and instantaneous and motility have also been proposed (WGHSEA, 2003). The relationship for motility was based on the results of a Dutch study with sufficient power; furthermore, several short-comings of earlier studies have been accounted for (Passchier, 2002). The applicability for health impact assessment is limited, for the relationship can not be applied in situations where the difference between night-time aircraft noise levels (L_{night}) and the equivalent aircraft noise levels for the 24hrs period differ from the differences found in the study of Passchier (2002). Although the relationship describing the association between aircraft noise events during the night and behavioural awakenings was derived by means of a meta-analysis, the applicability of this relationship was limited. An important reason was that the effect was only studied at individual level.

4.3 Cardiovascular system

In relation the effects of noise on the cardiovascular system, only the results of the meta-analysis of Van Kempen (2002) can be used for estimating the number of people with an effect on their cardiovascular system that is attributable to noise exposure.
A problem that is difficult to overcome by means of a meta-analysis, is the fact that the different studies used different exposure groups to which they refer to: some studies used a reference group consisting of persons exposed to noise levels less than 60 dB(A), while others used a reference group consisting of persons exposed to a noise range of 45-50 dB(A). Because of this, there is much uncertainty about the shape of the relationship. There are several views ranging from studies suggesting ‘u-shaped’ curves or a continuous increase in risk of noise with increasing noise exposure for noise levels to studies suggesting no associations. There is also discussion about the threshold or observation value: some overviews concluded that there is an observation threshold for hypertension and ischemic heart disease corresponding to an L_{dn} value of 70 dB(A) (WHO, 2000) (Passchier-Vermeer, 2000). According to the meta-analysis of Van Kempen (2002), the existing database is not fit to determine the shape of a possible relation with cardiovascular effects or a threshold value. This has also to do with the observation range of the studies: this is most of the time limited to 70, 75 dB(A). How the choice of the value of a reference group affects the number of people with cardiovascular disease, was demonstrated in Van Kempen et al. (2001). Part of the problems regarding the shape of the relationship between noise and

effects on the cardiovascular system could be tackled by constructing a database with results from individual studies investigating the effect of noise exposure on the cardiovascular system. Preferably, exposure data have to be available at individual level. In the near future, the EU-sponsored project Hypertension and Exposure to Noise near Airports (HYENA, 2003-2006) can provide these data for aircraft noise: The overall goal of this project 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 will be 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.

4.4 Limitations of the underlying studies

As already mentioned, the number of usable exposure-effect-relationships is limited in contrast with the amount of research that is carried out until now. This has several reasons that can be found in the underlying studies. First some general problems that emerge more often in environmental epidemiologic studies are addressed.
Most single studies used in the studies deriving an exposure-effect-relationships were cross-sectional. A problem that arises in relation to the interpretation of cross-sectional studies in general is that this confounds both the determination of the direction of the causation and the accurate estimation of the exposure (Stansfeld et al., 2000). Another problem that is related to cross-sectional studies is the problem of self-selection (healthy worker effect), which was already addressed in 3.3.3. However, until now, the existence of such an effect in the field of noise research has not been proven yet. In relation to annoyance and sleep disturbance due to aircraft noise, expansions, alterations etcetera are more or less ongoing; this makes that these studies are dealing with some kind of change-situation. So, the moment at which the cross-sectional study takes place is very important.
Another problem that emerges often in environmental epidemiological studies involves the estimate of the exposure and the inability to apply individual exposure estimates (if available) to larger study populations. It appears that health outcomes of people living in proximity tend on average to be more alike than those form other areas. This may also be the case in noise studies investigating the effects on health and well-being (Pattenden, 2001).

4.4.1 Exposure-characterization

Environmental noise exposure is usually expressed as the average of the noise events over a certain time (T), expressed in dB(A). The equivalent noise level is an exponential average of the noise levels over a certain time (Björkman et al., 1998). By means of this way of averaging, higher levels get more weight than the lower levels.
A deficiency in the studies investigating the impact of noise on health and well-being involved the estimate of the exposure and the inability to apply individual exposure estimates (if available) to larger study populations. To date, most assessments of the impact of noise

exposure have involved between-group comparisons (high vs low): noise levels were measured or modelled for a school, a residential area, a neighbourhood or a city. Subsequently, this noise level was assigned to everybody who is a member of that group: the children attending the particular school or the respondents living in that particular neighbourhood, residential area. Such comparisons between groups were subject to exposure misclassification. Another problem is that noise exposure variables have not been directly validated for their use as exposure measures in epidemiological studies.

In mainly all noise studies the level of exposure was estimated by means of noise models incorporated in Geographic Information Systems (GIS). These models were able to predict equivalent noise levels in function of traffic data provided by a traffic model, provided that a number of parameters, describing the characteristics of the road network, of the town buildings and of the site of the environment and the meteorological conditions as well, are known and acquired as input data. A noise model will predict hourly equivalent noise levels at fixed outdoor points. From this, adopted noise indicators will be calculated. However, even exposure assessments based on geographic models may be inadequate unless these models have been validated.

In other cases, noise exposure was based on noise measurements by means of sound level meters. Usually these meters were situated in representative parts/spots of the study area. The measurements were carried out during a certain time period on different periods of the day. The results are used to estimate the equivalent noise levels for the area. According to a recent study, it appeared that there are differences between measured and modelled aircraft noise levels around Schiphol (Fast, 2004) (Commissie Deskundigen Vliegtuiggeluid, 2004). At the moment there is still discussion about the magnitude and direction of these differences. Possible explanations for these differences have not investigated yet. How possible differences between modelled and measured aircraft noise levels affect the derived exposure-effect relations is not clear yet, for most were based on studies where the noise levels are measured and/or modelled.

The resulting noise exposure metrics primarily describe the noise energy in different time periods and indicate the average exposure of an individual and/or school and/or residential area for the period of 1 year. This makes that they are not very specific: e.g. within a year the aircraft noise levels can vary. So differences between the metrics are (i) the extent in which the number of noise events take place and (ii) the time periods they occur and how the individual events are ‘summed up’. How this affects the exposure-effect relation is not clear. It is important to get more insight into this matter. Looking at aviation for example, we can observe that individual airplanes produce less noise, while at the same time the number of flight events increases.

4.4.2 Statistics

Most observational noise studies have not been able to make adjustments for important factors such as noise sensitivity, insulation, body mass index (BMI), ethnicity, etcetera. Furthermore, in most studies investigating the impact of noise on people’s health and well-being, group-level noise exposure data are combined with individual level data. Because data

are often not available at individual level, noise levels are measured or estimated for a residential area, a neighbourhood or a city. Subsequently, this noise level is assigned to everybody who is a member of that group: the people living in that particular neighbourhood, residential area. The statistical models used assume independence between the observations, but in a hierarchical study, observations with areas are not independent. Studies don't always adjust for this in their model (Pattenden, 2001). However, when using group-level exposure data, this should be accounted for in the statistical analysis.
Another problem is that instead of investigating the population that is distributed over a certain exposure range by using a continuous noise exposure measure, studies have tended to create noise categories (e.g. high, medium, low) by using indicator terms for ordered polytomuous exposure categories. However, it is recognized that the results may be sensitive to decisions about cut-points used to categorize continuous exposure variables and the method used to assign scores to exposure categories. It might introduce exposure misclassification which can be differential with respect to the outcome and, consequently lead to biased exposure-effect trend estimates (Richardson and Loomis, 2004).

4.5 Generalizability/transferability

Health impact assessments usually apply exposure-effect estimates derived from a study in one population to estimate impacts in another. Such assessments assume that the exposure-effect estimates are transferable. The validity of this assumption implicitly requires that the two populations be similar with regard to factors that influence the magnitude of the exposure-effect estimate, such as structure of the morbidity, basic health status, or noise type (Krzyzanowski et al., 2002).

For annoyance and sleep disturbance, the generalizability of the derived exposure-effect curves to different countries and different areas has not been well established. What makes it complicated is that not only personal but also situational problems play a role: it is not unlikely that there are substantial differences in terms of susceptibility to noise. It is hypothesized that the annoyance responses of people in different countries deviate from the established curves because of differences in cultural expectations about the acceptability of transportation noise exposure, differences in climate and the adequacy of housing sound insulation techniques (Staatsen et al., 2004). With regard to sleep disturbance, individuals differ from one another both in terms of their biological responses to night-time noise and in terms of the effects on their health and well-being. Much depends on the extent to which a variety of inherent and acquired personal factors interact with environmental factors (HCN, 2004). An example of this is demonstrated in Figure 9: the results of a recent survey (2002) around Schiphol Airport in Amsterdam (Netherlands) (Breugelmans et al., 2004) are compared with the curve derived by Miedema (2004) describing the association between aircraft noise exposure and severe sleep disturbance. Although the shapes of the exposure response curves are highly comparable, the percentage of severe sleep disturbance along all noise levels is still much higher than one would expect on the basis of available generalized exposure-effect curve of Miedema and Oudshoorn (2001). Similar results are found for annoyance when comparing the results of the Schiphol survey (see also section 3.1.5) or the results of a survey around Maastricht with the Miedema curve for aircraft noise annoyance (Van Dongen and Vos, 2003) .

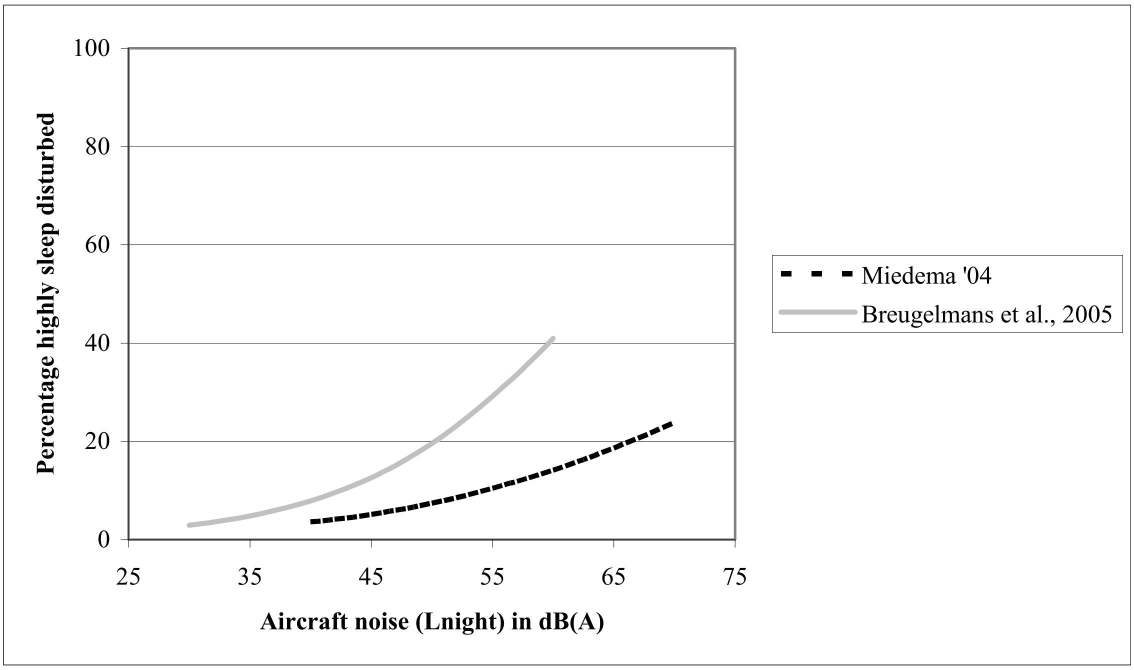

Figure 9. Comparison between the exposure-effect relationships derived in a survey around Schiphol Airport (Breugelmans et al., 2004) and the Miedema curve (2001) for sleep disturbance due to aircraft noise .

Other factors that may produce bias in terms of transferability to other populations are differences in daily pattern of activity, climatic conditions, housing, and different importance of confounding factors that might not have been properly controlled for in the epidemiological studies. But also differences in flight patterns and the composition of the aircraft- and road traffic fleet between the countries can be of importance (Van Kempen et al., 2003).

The relationships presented for the effects on the cardiovascular system in the meta-analysis were often based on the results of one or two relative old studies. It is questionable whether it is valid to extrapolate the results of these studies to the actual situation in the Netherlands. We already addressed the fact that the composition of aircraft noise has changed over the years. Nowadays houses are much better insulated against noise. As an alternative for the meta-analysis, one could also decide to use the results of a more recent and better study, when estimating the noise impact on cardiovascular disease. When doing this, one should check whether the study has sufficient power and whether the investigated sample is comparable with the Dutch population (WHO, 2002).

4.6 Children versus adults

The available exposure-effect- relationships are in most cases valid for adults. In the WHO guidelines for noise, children are regarded as a vulnerable group for the effects of noise. It is suggested that children are more sensitive to noise than adults because of noise exposure during critical developmental periods (organ development of foetuses, babies, learning of children). In addition, children may have less possibilities of controlling the noise or a less developed coping repertoire as compared to adults (Stansfeld et al., 2003).
However, judging from earlier daytime studies of children and adults doing the same cognitive tasks while exposed to noise, children are not more sensitive than adults to noise (Boman and Enmarker, 2004), but they perform at a lower level than the adults in noise and in quiet.
A recent study (Haines et al., 2003) found indications that child noise annoyance is the same construct as adult noise annoyance: the emotional response of children to describing the annoyance reaction was consistent with adult reactions. In comparison with children, for both parents and teachers steeper exposure-effect relationships were observed than for children (Lercher, 2002) (Van Kamp et al., 2003). Although it is very difficult to say whether children are more vulnerable to adults in relation to the effects on sleep, one has to realize that there are differences in sleep patterns between adults and children (Fast, 2004).
From the above it is questionable whether children are more vulnerable to noise in relation to health and cognition, but since so much more of cognitive work is expected from children while in school, their learning environment and their cognitive tasks can be said to be more noise vulnerable than corresponding environments for adults (WHO, 2002).

Conclusions and recommendations

This report is a background document that can be used to assess the health impact attributable to noise in the Netherlands. To this end the available exposure-effect-relationships in the field of noise and health were evaluated. This evaluation reveals that the following relationships are suitable for health impact assessment purposes at this stage. These are:

- the relationships for the association between noise from road, rail and air traffic and **annoyance** derived by Miedema and Oudshoorn (2001);
- the relationships for the association between noise from road, rail and air traffic and **sleep disturbance** derived by Miedema (2003 and 2004); and
- the relationships describing the effects of noise on the **cardiovascular system** derived by Van Kempen (2002).

With regard to the effects on sleep, also relationships for awakenings and motility have been proposed. However, the applicability of these curves is rather limited. An important reason is that the effects were only studied at individual level. In order to assess the impact of noise on children's reading ability, the results of the RANCH-study can be used. Although the results of this study are robust, the interpretation at population level is difficult.

Because the responses regarding annoyance and sleep disturbance of people in different countries might be different due to differences in cultural expectations about the acceptability of transportation noise exposure, differences in climate and the adequacy of housing sound insulation techniques, the use of the annoyance and sleep disturbance curves for local situations should be applied with great care. For aircraft noise exposure there are indications that the annoyance and sleep disturbance response increased during the last years. Therefore we recommend the use of national reference data if available. If this is not possible, the generalised relations published by Miedema (2001) (2003) (2004) could be used to estimate annoyance and sleep disturbance levels.

Despite the fact that the underlying mechanisms are plausible and the large amount of available data, the epidemiological evidence for an association between noise and cardiovascular disease is limited. At the moment, some risk estimates for road traffic and aircraft noise are available for adults which can be used. The thresholds of no-effect to be used and the shape of the curve are still debatable. In order to get a feeling how these uncertainties might affect the estimates, we recommend that they are accompanied with a sensitivity analysis. As an alternative for the meta-analysis, one could also decide to use the results of a more recent and better study, when estimating the noise impact on cardiovascular disease. When doing this, one should check whether the study has sufficient power and whether the investigated sample is comparable with the Dutch population.

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Appendix I. Noise exposure in studies investigating the effects of noise exposure on health and well-being

Exposure metrics
To judge noise levels and their possible impacts on health, several noise metrics are available. These measures start from a physical quantity to which corrections are applied that account for the human noise sensitivity. These corrections depend on the frequency, noise characteristics, and the noise source. Noise measures relevant for this report are explained below (Staatsen et al., 2004):

- ***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 10 dB(A) since noise during the evening and the night is 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 hours) with 10 dB(A).
- ***L_{night}*** The equivalent sound level over night-time (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. Only in some sleep disturbance studies indoor noise measurements have been taken.

In studies investigating the effects on annoyance, noise was usually expressed in L_{DN} or L_{den}; studies investigating the effects on the cardiovascular system and cognition, usually used the $L_{Aeq,7\text{-}23\ hr}$ or $L_{Aeq,\ 6\text{-}22\ hr}$. In the different studies investigating the effects of noise exposure on sleep a lot of different noise exposure metrics were used: ranging from time-averaged noise exposure ($L_{Aeq, T}$) to peak noise level (L_{Amax}) (Franssen and Kwekkeboom, 2003). Which metric is used in which case depends a little of which effects are investigated:

- **studies investigating the reactions on noise events (e.g. an over-flight)**: the noise level of an event is usually expressed by means of a Sound Exposure Level (SEL) and/or by the maximal noise level of the event (L_{Amax}). Both metrics are highly correlated (Fast et al., 2004).
- **studies investigating the effects before, during and after a night of sleep**: noise levels during a night are usually expressed by means of the equivalent noise level for the night period (23 to 7 hr) (L_{night}). The L_{night} is calculated from the SELs of the noise events that took place in the period between 23 and 7 hr (Fast et al., 2004). For health impact assessment purposes, L_{night} is not defined as an inside level because insulation quality and window-behaviour differs considerably between individuals and between countries; in Europe a large proportion of the population likes to sleep with windows open to some extent (WGHSEA, 2003). In most cases the noise levels are calculated at the most exposed side of the house. As people will try to avoid high noise levels by choosing a bedroom on the least exposed side, research results may get biased if only the most exposed value is taken (WGHSEA, 2003).
- **in studies investigating the effects of long-term noise exposure** on health and well-being, the L_{night} is also often used.

Operationalisation

In studies investigating the possible impact of noise exposure on health and well-being, exposure was based on the results of noise measurements or on the results of calculations with noise exposure models.

In these studies, measurements are generally carried out by means of sound level meters that are situated in representative parts/spots (usually outdoors) of the study area. The measurements are carried out during a certain time period on different periods of the day. The results are used to estimate noise levels for a study area / location:

- the L_{den} or L_{DN} (often used in relation to annoyance);
- the L_{night} (in relation to sleep disturbance);
- equivalent noise levels for the day and evening ($L_{Aeq, 7\text{-}23\ hr}$ or $L_{Aeq, 6\text{-}22hr}$) (in relation to effects on the cardiovascular system or cognition);

In some cases the output of monitoring systems is used. It has to be realised that the results of noise measurements strongly depend on the measurement area: such the shielding and reflection of buildings, and meteorological conditions (Fast et al., 2004).

Other studies estimate the level of exposure by means of noise models. These models are able to predict equivalent noise levels in function of traffic data provided by a traffic model, provided that a number of parameters, describing the characteristics of the road network, of the town buildings and of the site of the environment and the meteorological conditions as well, are known and acquired as input data. A noise model will predict hourly equivalent noise levels at fixed outdoor points: this might be e.g. the centroid of a study area or sometimes at address-level. From this, adopted noise indicators (such as L_{DN} and L_{den}) will be calculated.

With regard to the use of different noise models one has to take note of the fact that different countries / regions have different calculation 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 the fleet and composition of road/rail in the different countries. Apart from these real existing differences, other (and bigger) differences in outcome between national assessments are due to (undesired) methodological artefacts. In the end, all these issues can cause differences up to 10 dB(A). So, for health impact assessment purposes, we prefer to use the same model.

Appendix II. Operationalisation of effects in studies investigating the effects of noise exposure on health and well-being.

Annoyance

Annoyance is usually measured by means of one or more questions that are part of a questionnaire including many other items. A problem is that the questions and the answer categories from which the respondents can choose for reporting their degree of annoyance, differ among the studies. Although usually the percentage highly annoyed is reported (in relation to noise) studies sometimes report the mean annoyance, or the percentage annoyed. 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.

With regard to children, less uniform methods were used to assess annoyance: In the Munich Airport study, annoyance was measured using Visual Analogue Scales (VAS) (Evans et al., 1995) (Evans et al., 1998); other studies used multi-item lists (Evans et al., 1995) (Evans et al., 1998) (Lercher et al., 2000). Both the Heathrow studies and the RANCH study used standardised questions who are similar to the questions used for adults (Haines et al., 2001a) (Haines et al., 2001b) (Haines et al., 2001c) (Van Kamp et al., 2003) (Lercher, 2003) (Stansfeld et al., 2003).

Effects on sleep

Different methods were used to investigate the possible effects of noise on sleep. We can distinguish between questionnaires and physiological examinations (see also Table A.1).

Table A.1. Overview of physiologic examinations used in studies investigating the possible effects of noise on sleep.

Type of examination	What is examined ?	Indicator for
Electroencephalograph (EEG)[1)]	The sleep stages	Total sleep time, total time spent overnight in Slow Wave Sleep (SWS; deeper sleep) and in the stage of Rapid Eye Movement (REM; dream sleep)
EMG[1)]	Muscle tonus	
EOG[1)]	Eye-movements	
Electrocardiography (ECG)	Cardiac function	Heart rate
Plethysmography	Heart rate and blood pressure	
Actimetry	Motility	Total sleep time, time of falling asleep, wake-up time, Number of awakenings
Overnight cortisol in blood or fluvia	Level of circulating catecholamines	Sympathetic nervous activity
Overnight urinary catecholamines	Level of total catecholamines released during sleep, not taken up by sympathetic nerve endings	

1) The measurement of brain activity by means of EEG, EMG and EOG is also called polysomnography.

The table shows that awakenings can be measured and defined in several ways. We distinguish between arousals (or EEG awakenings) and behavioural awakenings. An arousal is defined as an EEG response that has all the characteristics of an awake individual; behavioural awakening is confined to a verbal or motor response, indicating the subject is awake. Behavioural awakenings are often measured by use of a switch mounted on the headboard of the bed or by a micro-switch taped on the hand (Lukas, 1975). It has to be kept in mind that behavioural awakenings are only a rough evaluation of sleep disturbance, because changes in sleep architecture (such as sleep stage changes and short lasting EEG awakenings) can be totally missed out. Awakenings can be related to both noise events and noise levels during a whole night. For analysis purposes, noise events are typically grouped into bins or ranges. Within each range the percentage of awakenings is determined (Pearsons, 1998).

The quality of the sleep can also be measured on a subjective way. This is done by means of an interview or questionnaire in which standardised questions are included. Box 1 gives some examples. Like annoyance, 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 scales are translated into a scale from 0 to 100. Cut-off points to assess the percentage of highly sleep disturbed persons are used, analogue to the definitions of percentage (highly) annoyed persons.

If aircraft noise wakes you up at night during weekday/weekend, how much does this annoy you ?

Not at all, a little, quite, very much.

Does <source> interrupt sleeping ?
No, yes.

Box 1. Examples of sleep disturbance questions (Miedema et al., 2003)

Effects on the cardiovascular system

Table A.2 shows how the different cardiovascular parameters that can be related to noise exposure have been operationalised. In case risk elevations are found for one or more of these parameters, this can be supposed as an indication of a (small) contribution to total cardiovascular disease prevalence; all parameters are part of the cardiovascular disease process (De Hollander, 2004).

Table A.2. Parameters measured in observational studies investigating the association between noise exposure and cardiovascular disease.

Effect measured	**Definition**	**Population where effect is measured**	**Method**
Blood pressure (mmHg)	Blood pressure changes	Especially in males who are (not) treated for hypertension, children	Sphygmomanometer, automatic blood pressure meter
Hypertension	SBP ≥ 160 and/or DBP ≥ 95mmHg and/or use of anti-hypertensives	Mainly healthy men	Blood pressure measurement and by means of a questionnaire
Use of anti-hypertensives	Use of anti-hypertensives	Mainly males	Questionnaire
Use of cardiovascular medication	Use of anti-hypertensives and/or cardiovascular medication	Both men and women	Questionnaire
Angina Pectoris	Prevalence of angina pectoris	Middle-aged persons	WHO/Rose-questionnaire or LSH-pain questionnaire
Myocardial infarction	Prevalence of myocardial infarction (incl mortality, hospital admission and ECG); incidence MI-cases	Middle-aged men	LSH-pain questionnaire and ECG (Minnesota code)
Ischemic heart disease	Prevalence consultation specialist, hospital admission, incidence angina (typical/a-typical), MI and ECG-ischemia	Mainly middle-aged persons	Questionnaire and ECG (Minnesota code)
Mortality due to ischemic heart disease			Data of records/administrations

abbreviations: SBP = Systolic blood pressure; DBP = diastolic blood pressure; MI = myocardial infarction.

Cognition

After listing the most recent field studies, it appeared that the following cognitive domains have been mainly investigated: attention, memory and intellectual achievement (reading and mathematics) (Stansfeld et al., 2000).

With regard to attention, we can distinguish tests measuring sustained and visual attention and vigilance. In sustained attention tests the children have to search (for a certain time-period) for target geometric stimuli (e.g. letters) from among fields of similar objects. In visual attention tests target pictorial stimuli have to be located within an array of pictures with verbal or key press response. Sometimes target stimuli have to be located within an array of other sounds (auditory attention). Examples of vigilance tests are reaction time tests.

Several aspects of memory are measured: long term memory and short term memory. In order to test the first, children usually have to listen to or read a passage/text. After a certain time recall and recognition is assessed. This involves multi-choice questions and written recall questions. An aspect of short term memory that is mainly measured is working memory. During several trials digit or letter sequences are presented visually or auditory (audio cassette). After that the children are requested to recall, in order, as many digits/letters as possible.

One of the indicators of intellectual achievement that are investigated most is reading. In some cases, effects on mathematics are also investigated. In order to investigate the effects on reading, standardized tests are usually used. However, due to cultural differences, differences in the school systems, the results of cognitive tests are not easy to compare between countries. There is also the problem of test-leader dependency because of the interaction between the tester and the child. This can cause observer bias. Furthermore, it appears that children have a lower score on a test if they know that nothing depends on the result of the test (Cito, 2004). How this phenomena affects the association with noise is not known.

Appendix III. Available exposure-effect relationships for annoyance

Table A.3. Characteristics of the available exposure-effect-relationships for annoyance.

Author (yr)	**Methods[a)]**	**Characteristics underlying studies**					**Derived curve**	
		Studies	**data points**	**Period**	**Location[b)]**	**Source(s) investigated**	**Exposure range**	**Formula**
Schultz 1978	B/C, LSM	11	161	1961-72	EU, NA	Road, rail and air	L_{DN}: 45-85 dB(A)	$\%HA = 0.8553L_{DN} - 0.0401L_{DN}^2 + 0.00047L_{DN}^3$
Fidell 1991	B/C, LSM	27	453	1961-90	EU, NA	Transportation	L_{DN}: 45-85 dB(A)	$\%HA = 78.9181 - 3.2645L_{DN} + 0.0360L_{DN}^2$
Finegold 1994	B/C, LSM	22	400	1961-90	EU, NA	Transportation	L_{DN}: 40-85 dB(A)	$\%HA = 100/(1+ e^{(11.13 - 0.141LDN)})$
Miedema 1998	C, LSM	20	34214	1965–92	EU, NA, AUS	Aircraft	L_{DN}: 45-75 dB(A)	$\%HA = 0.53 (L_{DN} - 42) + 0.0285 (L_{DN} - 42)^2$
		26	21228	1971-94	NL, EU, NA	Road traffic		$\%HA = 0.03 (L_{DN} - 42) + 0.0353 (L_{DN} - 42)^2$
		9	8527	1972-93	NL, EU	Rail traffic		$\%HA = 0.01 (L_{DN} - 42) + 0.0193 (L_{DN} - 42)^2$
Miedema 1998	C ML	20	34214	1965–92	EU, NA, AUS	Air	L_{DN}: 45-75 dB(A)	$\%HA = -0.02 (L_{DN} - 42) + 0.0561 (L_{DN} - 42)^2$
		26	21228	1971-94	NL, EU, NA	Road		$\%HA = 0.24 (L_{DN} - 42) + 0.0277 (L_{DN} - 42)^2$
		9	8527	1972-93	NL, EU	Rail		$\%HA = 0.28 (L_{DN} - 42) + 0.0085 (L_{DN} - 42)^2$
Miedema 2001	C, ML	19	27081	1965–92	EU, NA, AUS	Air	L_{DN}: 45-75 dB(A)	$\%LA = -5.741x10^{-4} (L_{DN} - 32)^3 + 2.863x10^{-2} (L_{DN}-32)^2 + 1.912 (L_{DN} -32)$ $\%A = 1.460x10^{-5} (L_{DN} - 37)^3 + 1.511x10^{-2} (L_{DN}-37)^2 + 1.346 (L_{DN} -37)$ $\%HA = -1.395x10^{-4} (L_{DN} - 42)^3 + 4.081x10^{-2} (L_{DN}-42)^2 + 0.342 (L_{DN} -42)$
	C, ML	19	27081	1965–92	EU, NA, AUS	Air	L_{den}: 45-75 dB(A)	$\%LA = -6.158x10^{-4} (L_{den} - 32)^3 + 3.410x10^{-2} (L_{den}-32)^2 + 1.738 (L_{den} -32)$ $\%A = 8.588x10^{-6} (L_{den} - 37)^3 + 1.777x10^{-2} (L_{den}-37)^2 + 1.221 (L_{den} -37)$ $\%HA = -9.199x10^{-5} (L_{den} - 42)^3 + 3.9321x10^{-2} (L_{den}-42)^2 + 0.2939 (L_{den} -42)$
	C, ML	26	19172	1971-94	NL, EU, NA	Road	L_{DN}: 45-75 dB(A)	$\%LA = -6.188x10^{-4} (L_{DN} - 32)^3 + 5.379x10^{-2} (L_{DN}-32)^2 + 0.723 (L_{DN} -32)$ $\%A = 1.732x10^{-4} (L_{DN} - 37)^3 + 2.079x10^{-2} (L_{DN}-37)^2 + 0.566 (L_{DN} -37)$ $\%HA = 9.994x10^{-4} (L_{DN} - 42)^3 - 1.523x10^{-2} (L_{DN}-42)^2 + 0.538 (L_{DN} -42)$
	C, ML	26	19172	1971-94	NL, EU, NA	Road	L_{den}: 45-75 dB(A)	$\%LA = -6.235x10^{-4} (L_{den} - 32)^3 + 5.509x10^{-2} (L_{den}-32)^2 + 0.6693 (L_{den} -32)$ $\%A = 1.795x10^{-4} (L_{den} - 37)^3 + 2.110x10^{-2} (L_{den}-37)^2 + 0.5353 (L_{den} -37)$ $\%HA = 9.868x10^{-4} (L_{den} - 42)^3 - 1.436x10^{-2} (L_{den}-42)^2 + 0.5118 (L_{den} -42)$
	C, ML	8	7632	1972-93	NL, EU	Rail	L_{DN}: 45-75 dB(A)	$\%LA = -3.343x10^{-4} (L_{DN} - 32)^3 + 4.918x10^{-2} (L_{DN}-32)^2 + 0.175 (L_{DN} -32)$ $\%A = 4.552x10^{-4} (L_{DN} - 37)^3 + 9.400x10^{-3} (L_{DN}-37)^2 + 0.212 (L_{DN} -37)$ $\%HA = 7.158x10^{-4} (L_{DN} - 42)^3 - 7.774x10^{-3} (L_{DN}-42)^2 + 0.163 (L_{DN} -42)$
	C, ML	8	7632	1972-93	NL, EU	Rail	L_{den}: 45-75 dB(A)	$\%LA = -3.229x10^{-4} (L_{den} - 32)^3 + 4.871x10^{-2} (L_{den}-32)^2 + 0.1673 (L_{den} -32)$ $\%A = 4.538x10^{-4} (L_{den} - 37)^3 + 9.482x10^{-3} (L_{den}-37)^2 + 0.2129 (L_{den} -37)$ $\%HA = 7.239x10^{-4} (L_{den} - 42)^3 - 7.851x10^{-3} (L_{den}-42)^2 + 0.1695 (L_{den} -42)$
Miedema 2004	A, ML	1	1751	2004	NL	Seasonal	L_{den}: 45-65 dB(A)	$\%LA = 39.156 - 2.146 L_{den} + 0.03096 L_{den}^2$ $\%A = 32.137 - 1.635 L_{den} + 0.02124 L_{den}^2$ $\%HA = 18.123 - 0.887 L_{den} + 0.01091 L_{den}^2$
						Shunting yards	L_{den}: 45-65 dB(A)	$\%LA = -69.963 + 3.171 L_{den} -0.0176 L_{den}^2$ $\%A = -27.629 + 0.722 L_{den} + 0.01265 L_{den}^2$ $\%HA = 16.980 - 1.367 L_{den} + 0.02980 L_{den}^2$

						Other	L_{den}: 45-65 dB(A)	%LA = 11.477 – 1.130 L_{den} + 0.02815 L_{den}^{2} %A = 36.854 – 2.121 L_{den} + 0.03270 L_{den}^{2} %HA = 36.307 – 1.886 L_{den} + 0.02523 L_{den}^{2}
Breugelmans 2005	A	1	5873	2002	NL	Air	L_{den}: 39-65 dB(A)	Logit(%HA) = -8.1101 + 0.1333 * L_{den} % HA = (exp(logit(%HA))/(1+(exp(logit(%HA)))*100 ++)

a) method applied to derive an exposure-effect relationship where A = data of a single study, B = Meta-analysis, C = re-analysis of individual data; Statistics used: LSM = Least Squares Method, ML = Maximum Likelihood method. B) Location where studies were carried out where NA= North-America, EU = Europe; ++) The derived exposure-response relationship is based on modelled flight paths

Appendix IV. Available exposure-effect relationships for the effects of noise on sleep

Table A.4.a. Characteristics of the available exposure-effect relationships for the effects of noise on sleep.

Study (year)	Characteristics underlying data						Exposure		
	Method [a)]	Statistics [b)]	# studies / design [c)]	Subjects included		Period/location [d)]	Noise source(s)	Noise parameter [e)]	Noise range
				N [f)]	Gender/age				
Lukas 1975	C		7 / lab	>62	M&F / 20-55	EU, Sixties	Stimuli from road traffic, air traffic, sonic booms	EPNL	
Lukas 1975	C		7 / lab	>62	M&F / 20-55	EU, Sixties	Stimuli from road traffic, air traffic, sonic booms	EPNL	
Griefahn 1976	C		10 / lab	158	5-75		Stimuli from airplane noise, sonic booms, pink noise and traffic noise	L_{Amax} and number of stimuli	40-120 dB(A) 7-32 stimuli per night
Griefahn 1976	C		10 /lab	158	5-75		Stimuli from airplane noise, sonic booms, pink noise and traffic noise	L_{Amax} and number of stimuli	40-120 dB(A) 7-32 stimuli per night
Pearsons et al., 1989	C	LSM	21 / lab & cross			1988		L_{Amax} of a noise event and SEL (indoor)	
Pearsons et al., 1989	C	LSM	21 / lab & cross					L_{Amax} of a noise event and SEL (indoor)	
Finegold, 1994	C		21 / lab & cross			See Pearsons et al, 1989		$ASEL_{indoor}$	
Hoffman 1994	B		NA/lab & field	134	5-75	1964 – 1991	Stimuli from aircraft noise	L_{Amax} and number of stimuli	At least 58 dB(A) 7-100 stimuli per night
Hoffman 1994	B		NA/lab & field	134	5-75	1964-1991	Stimuli from road traffic noise		40-100 dB(A) 7-100 stimuli per night

Fidell 1998	B		8 / field	100		EU, NA /1972 - 1998	Comm & military aircraft, ambient noise, sonic booms, heavy truck, railway	SEL	
Finegold and Elias 2002	B		8 / field	100		EU, NA / 1973 - 1998	Comm & military aircraft, ambient noise, railway	ASEL	
Passchier, 2003	B		8 / field	110	18-81	EU, NA /1973 - 2002	Community aircraft	SELi	54 – 90 dB(A)
Passchier et al., 2002	A	ML	1 / field	418	18-81	NL	Aircraft noise	$L_{max\ indoor}$, SEL_{indoor}	63242 aircraft noise events
Passchier et al., 2002	A	ML	1 / field	418	18-81	NL	Aircraft noise	$L_{night,\ outdoor}$	
Miedema et al., 2003	B	ML	14 / field	8459		EU, NA, AS / 1975 – 2001	Road traffic noise	$L_{night\ outdoor}$ at most exposed façade	45-65
Miedema et al., 2003	B	ML	7 / field	4098		EU, AS / 1983 – 2001	Rail traffic noise	$L_{night\ outdoor}$ at most exposed façade	45-65
Miedema et al., 2004	B	ML	8 / field	9734		1967 – 2004 / EU, NA	Aircraft noise	$L_{night\ outdoor}$ at most exposed façade	45-65
Breugelmans et al., 2005	A		1 / field	5873	> 18 yr	2002 / NL	Aircraft noise	L_{night}	30–60

a) Method applied to derive an exposure-effect relationship where A = data of a single study, B = Meta-analysis, C = Re-analysis of individual data; b) LSM: Least squares method; ML = Maximum Likelihood method (usually applied in multi-level models); c) Design: cross = cross-sectional study, lab = laboratory studies; d) location where studies were carried out, where NA = North America; EU = Europe, excluding the Netherlands, AS = Asia, NL = the Netherlands; e) noise parameters EPNL = Effective Perceived Noise Level; ASEL = A-weighted Sound Exposure Level;

Table A.4.b Characteristics of the available exposure-effect relationships for the effects of noise on sleep.

Study (year)	Outcome		Derived curve	
	Definition	**Measurement**	**Formula** [a)]	**Remarks**
Lukas 1975	Average % of behavioural awakenings	Behavioural (button)	% persons without sleep disturbance = -1.552 * L + a	
Lukas 1975	Average % of sleep stage changes	EEG patterns		
Griefahn 1976	Awakening		Probability awake = 1.32 * L_{Amax} – 79.67	Source: Hoffman '94
Griefahn 1976	Sleep stage changes			
Pearsons et al., 1989	Percentage of awakenings		% Aroused = 0.1159*L_{Amax} – 4.7249	Sound levels in bedroom, derived from field studies
Pearsons et al., 1989	Percentage of sleep stage changes		% Sleep disrupted = 0.7748*L_{Amax} – 22.0715	Sound levels in bedroom, derived from field studies
Finegold, 1994	Awakenings		% Awakenings = 7.1 x 10^{-6} * $L_{AE}^{3.5}$	Indoor sound levels
Hoffman 1994	Probability awake _ air	EEG patterns	Probability awake = 0.43462 * L_{Amax} – 9.1415	
Hoffman 1994	Probability awake_road	EEG patterns	Probability awake = 1.0357 * L_{Amax} – 42.749	
Fidell 1998	Percentage of subjects awake	EEG patterns, behavioural, actimetry		
Finegold and Elias 2002	Percent of awakenings	EEG patterns, behavioural, actimetry	% awakenings = 0.58 + (4.30 x 10^{-8}) x SEL $^{4.11}$	
Passchier 2003	Percentage noise induced awakenings	EEG patterns, behavioural, actimetry	% noise-induced awakenings = -0.564 + 1.909x10^{-4} x SEL_i^2	Indoor, in bedroom
Passchier et al, 2002	Noise-induced mean motility	Actimetry	Noise-induced mean motility = 0.000192 x (L_{night} – 21)	The difference between L_{night} and the similar L_{Aeq} at the façade of the bedroom is assumed 0 dB(A); The difference between the night-time L_{Aeq} outdoors at the façade of the bedroom and in the bedroom during the sleep period is assumed to be 21 dB(A)
Passchier et al., 2002	Probability of motility	Actimetry	Max expected number of intervals with motility = N x [0.0001233 x (L_{night} + 70.2 – 10 lgN – 21)^2 – 0.007415 x (L_{night} + 70.2 – 10 lgN – 21) + 0.0994]	The difference between L_{night} and the similar L_{Aeq} at the façade of the bedroom is assumed 0 dB(A); The difference between the night-time L_{Aeq} outdoors at the façade of the bedroom and in the bedroom during the sleep period is assumed to be 21 dB(A)

Miedema et al., 2003	Percentage highly sleep disturbed, sleep disturbed and a little sleep disturbed	Questions regarding waking up or being disturbed by noise during night	$\%HSD = 20.8 - 1.05 * L_{night} + 0.01486 * L_{night}^2$ $\%SD = 13.8 - 0.85 * L_{night} + 0.01670 * L_{night}^2$ $\%LSD = -8.4 + 0.16 * L_{night} + 0.01081 * L_{night}^2$
Miedema et al., 2003	Percentage highly sleep disturbed, sleep disturbed and a little sleep disturbed	Questions regarding waking up or being disturbed by noise during night	$\%HSD = 11.3 - 0.55 * L_{night} + 0.00759 * L_{night}^2$ $\%SD = 12.5 - 0.66 * L_{night} + 0.01121 * L_{night}^2$ $\%LSD = 4.7 - 0.31 * L_{night} + 0.01125 * L_{night}^2$
Miedema et al., 2004	Percentage highly sleep disturbed, sleep disturbed and a little sleep disturbed	Questions regarding waking up or being disturbed by noise during night	$\%HSD = 18.147 - 0.956 * L_{night} + 0.01482 * L_{night}^2$ $\%SD = 13.714 - 0.807 * L_{night} + 0.01555 * L_{night}^2$ $\%LSD = 4.465 - 0.411 * L_{night} + 0.01395 * L_{night}^2$
Breugelmans et al. 2005	Percentage highly sleep disturbed	Question regarding being disturbed by noise during night (past year; 11 pointscale)	Logit (%HSD) = $-6.642 + 0.1046 * L_{night}$ %HSD=(exp(logit(%HSD)/(1+(exp(logit(%HSD))))*100 ++)

a) % HSD = Percentage highly sleep disturbed; % SD = Percentage sleep disturbed; %LSD = percentage (at least) a little sleep disturbed. ++) The derived exposure-response relationship is based on modelled flight paths

Appendix V. Available exposure-effect relationships for the effect of noise exposure on the cardiovascular system

Table A.5. Characteristics of the available exposure-effect relationships describing the effects of noise on the cardiovascular system.

Author	Characteristics of the studies included # studies	Period	Location [a)]	Source	Outcome	Valid range	Formula
HCN 1994	20	-	-	Occup noise	Hypertension		RR = 1.7
	6	1976 – 89	NL, EU	Air & road	Hypertension	70-80	$RR = 0.05 + 0.007 * L_{den}$
	-	-	-	Air & road	Ischeamic heart disease	70-80	$RR = 0.05 + 0.008 * L_{den}$
Duncan 1993	9	1976–88	NL, EU, AS	Occup and community	Hypertension (males)	$L_{DN} > 61$	ß = 0.021 (95%CI = 0.013 – 0.029)[± b]
	2	1977-79	NL, EU	Occup and air	Hypertension (females)	$L_{DN} > 61$	ß = 0.031 (95% CI = 0.006 – 0.056) [± b]
Van Kempen 2002	9	1976 – 99	SA, NA, EU	Occup	Hypertension	$L_{Aeq\ 8hr}$: 55 – 116 [c)]	$RR_{5\ dB(A)}$ = 1.14 (95% CI = 1.01 –1.29)[± d]
	2	1976 – 84	NL	Road	Hypertension	$L_{Aeq\ 6\text{-}22\ hr:}$ <55 – 80 [c)]	$RR_{5\ dB(A)}$ = 0.95 (95% CI = 0.84 –1.08) [d]
	2	1976 – 80	NL, EU		Use of anti-hypertensives	$L_{Aeq\ 6\text{-}22\ hr:}$ >50 – 73 [c)]	$RR_{5\ dB(A)}$ = 0.96 (95% CI = 0.76 –1.22) [d]
	1	1976	NL		Consultation GP/specialist	$L_{Aeq\ 6\text{-}22\ hr:}$ 55 – 70 [c)]	$RR_{5\ dB(A)}$ = 0.91 (95% CI = 0.73 –1.12) [d]
	2	1993	EU		Angina Pectoris	$L_{Aeq\ 6\text{-}22\ hr:}$ 51 – 70 [c)]	$RR_{5\ dB(A)}$ = 0.99 (95% CI = 0.84 –1.16) [d]
	3	1993 -94	EU		Myocardial infarction [e)]	$L_{Aeq\ 6\text{-}22\ hr:}$ 51 – 80 [c)]	$RR_{5\ dB(A)}$ = 1.03 (95% CI = 0.99 –1.09) [d]
	2	1993 – 99	EU		IHD-total [e)]	$L_{Aeq\ 6\text{-}22\ hr:}$ 51 – 70 [c)]	$RR_{5\ dB(A)}$ = 1.09 (95% CI = 1.05 –1.13) [± d]
	1	1976	NL	Air	Hypertension	$L_{Aeq\ 6\text{-}22\ hr:}$ 55 – 72 [c)]	$RR_{5\ dB(A)}$ = 1.26 (95% CI = 1.14 –1.39) [± d]
	1	1976	NL		Use of anti-hypertensives	$L_{Aeq\ 6\text{-}22\ hr:}$ 55 – 72 [c)]	$RR_{5\ dB(A)}$ = 0.99 (95% CI = 0.87 –1.14) [d]
	2	1976, 1998	NL		Consultation GP/Specialist	$L_{Aeq\ 6\text{-}22\ hr:}$ 55 – 77 [c)]	$RR_{5\ dB(A)}$ = 1.10 (95% CI = 0.95 –1.27) [d]
	2	1976	NL		Use cardiovasc medicines	$L_{Aeq\ 6\text{-}22\ hr:}$ 38 – 77 [c)]	$RR_{5\ dB(A)}$ = 1.05 (95% CI = 0.99 –1.11) [d]
	1	1976	NL		Angina Pectoris	$L_{Aeq\ 6\text{-}22\ hr:}$ 55 – 72 [c)]	$RR_{5\ dB(A)}$ = 1.03 (95% CI = 0.90 –1.18) [d]

a) Location where studies were carried out, where NA = North America, EU = Europe, excluding the NL, AS = Asia, NL = the Netherlands. $^{\pm}$ = significant at $p < 0.05$. b) By means of this ß the authors calculated odd ratios for several L_{DN} levels; c) these are measurement ranges and should not be confused with threshold values; d) after adjustment for gender and age; e) only prevalence estimates

Appendix VI.Characteristics of the studies investigating the association between noise exposure and blood pressure in children.

Table A.6.a. Study characteristics of the studies investigating 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 – 4^{th} 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 – 4^{th} grade	262/ 163	Air traffic	$L_{Amax, mean}$ 74 dB(A) $L_{Amax, mean}$ 56 dB(A)	SLM, interior	22; 24, 26 – 32
Ising et al. ,1990	Germany	Cross	9-13 yr	94	Military air	Low altitude flight zones 150 m area vs 75 m area		3, 23
Ising et al., 1990	Germany	Cross	9-13 yr	433	Military air	Low altitude flight zones 150 m area vs 75 m area		3, 23
Ising et al., 1991	Germany	Cross	12-17 yr	467	Military air	Low altitude flight zones Control area vs 75 m area		3, 23, 33
Schmeck, 1993	Germany	Cross	4-17 yr	376	Military air	Low altitude flight zones Low vs high		3, 24, 36
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); 36 = psycho-social factors

Table A.6.b. Study characteristics of the studies investigating the 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 a 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
Ising et al., 1990	-	-	-	-	Children living in the 75 m area had a lower bp than children living in the 150m area; non-significant differences of 1 mmHg were found
Ising et al., 1990	-	-	-	-	Higher readings up to 9 mmHg in systolic blood pressure and up to 3 mmHg in diastolic blood pressure were found in children living in extreme low-flying zones
Ising et al., 1991	-	-	-	-	Although differences up to 2 mmHg for both systolic and diastolic blood pressure were found, these were not significant
Schmeck, 1993	-	-	-	-	Non-significant differences of up to 2 mmHg 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 proximate 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, acclimating	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 klass 10, when comparing the quiet school with the most noisiest school; b) only qualitative results were available