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Good practice guide on noise exposure and potential health effects

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The Expert Panel on Noise (EPoN) advises the EEA on noise policy issues.

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Preface

The Expert Panel on Noise (EPoN) is a working group that supports the European Environment Agency and European Commission with the implementation and development of an effective noise policy for Europe.

The group aims to build upon tasks delivered by previous working groups, particularly regarding Directive 2002/49/EC relating to the assessment and management of environmental noise.

This good practice guide is intended to assist policymakers, competent authorities and any other interested parties in understanding and fulfilling the requirements of the directive by making recommendations on linking action planning to recent evidence relating to the health impacts of environmental noise and, among others, the Night Noise Guidelines for Europe as recently presented by the World Health Organisation.

The contents should not be considered as an official position statement of the European Commission. Only the text of the directive is applicable in law at Community level. If in any circumstance, the guidance contained in this good practice guide seems to be at variance with the directive, then the text of the directive should be applied.

1 Introduction

1.1 Scope of this paper

The main purpose of this document is to present current knowledge about the health effects of noise. The emphasis is first of all to provide end users with practical and validated tools to calculate health impacts of noise in all kinds of strategic noise studies such as the action plans required by the Environmental Noise Directive ([i]) (END) or any environmental impact statements. The basis of this is a number of recent reviews carried out by well known institutions like WHO, National Health and Environment departments and professional organisations. No full bibliography is provided but the key statements are referenced and in the reference list, some documents are highlighted which may serve as further reading.

Noise is normally defined as 'unwanted sound'. A more precise definition could be: *noise is audible sound that causes disturbance, impairment or health damage*. The terms 'noise' and 'sound' are often synonymously used when the purely acoustical dimension is meant (e.g. noise level, noise indicator, noise regulation, noise limit, noise standard, noise action plan, aircraft noise, road traffic noise, occupational noise). Noise annoyance, in contrast, is a term used in general for all negative feelings such as disturbance, dissatisfaction, displeasure, irritation and nuisance ([ii]). Adverse effects of noise occur when intended activities of the individual are disturbed. The sound level of the acoustic stimulus, its psycho-acoustical sound characteristics, the time of its occurrence, its time course, its frequency spectrum and its informational content modify the reaction. During sleep, however, unconscious activation of the autonomous nervous system takes place without cortical (cognitive) control, due to direct interaction between the hearing nerve and higher structures of the central nervous system. Noise indicators such as L_{den} and L_{night}, regardless of any weighing factors, describe the exposure situation. The link between exposure and outcome (other terms: endpoint, reaction, response) is given by reasonably well-established exposure-response curves which are derived from research into noise effects. Large parts of this document deal with exposure-response curves that can be used for impact assessment. The content of this document was finalised in June 2010. The EPoN reserves the right to issue an update to the advice contained in the document at a time when the members consider it appropriate to do so.

1.2 Definitions of health

For the purpose of this document the larger definition of health is used. Although several other definitions of health are in use or have been proposed (see Annex I for an overview), the one that comes close to the intentions of this document is the original definition from the WHO-charter:

Health is a state of complete

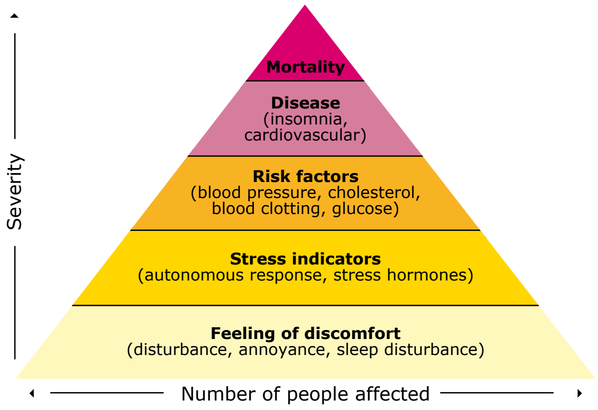
- *physical,*
- *mental, and*
- *social well-being*

and not merely the absence of disease or infirmity (WHO, 1946).

Figure 1.1 illustrates how exposure to noise affects health and wellbeing. If a certain population is exposed to substantial noise, many people will notice it and develop adverse feelings to this. Within a part of this exposed population, stress reactions, sleep-stage changes and other biological and biophysical effects may occur. These may in turn increase risk factors like blood pressure. For a relatively small part of the population these factors may then develop into clinical symptoms like insomnia and cardiovascular diseases which, as a consequence, can even increase the death rate.

The various reviews depict complex models for the relations between noise and stress and noise and

Figure 1.1 Pyramid of effects (WHO 1972 — modified)[iv]



Source: Babisch, W, 2002[XVII].

sleep disturbance. Most of the steps in the models have been verified experimentally, although for some only qualitatively. In general these models are accepted as describing the relations between noise and health.

1.3 Definitions of noise indicators used in this document

As the noise level of the sources addressed here varies with time, some way to aggregate the data in order to describe a situation is needed. The best analogy is the variation in temperature: according to use the daily minimum or maximum is presented, or the daily, monthly, or even yearly average. As the effects described here range in time scale from instantaneous to chronic, so do the noise indicators range from a (split) second to a year. The averaging method used for noise is the energy equivalence, hence this is called the equivalent continuous sound level, abbreviated L_{eq}. Without indication of the averaging time this is by itself quite meaningless.

Table 1.1 Noise indicators

Indicator *	Description	Time-constant
L_{max}	Maximum sound pressure level occurring in an interval, usually the passage of a vehicle	125 ms **
SEL	Sound exposure level = Sound pressure level over an interval normalised to 1 second.	1 s
L_{day}	Average sound pressure level over 1 day. This day can be chosen so that it is representative of a longer period — for example, L_{day} occurs in the END; if used in that context, a yearly average daytime level is intended.	12 or 16 hrs
L_{night}	Average sound pressure level over 1 night. This night can be chosen so that it is representative of a longer period — L_{night} also occurs in the END; if used in that context, a yearly average night time level is intended. This is the night time indicator defined in EU-directive 2002/49 and used by WHO.	8 hrs
L_{24h}	Average sound pressure level over a whole day. This whole day can be chosen so that it is representative of a longer period.	24 hrs
L_{dn}	Average sound pressure level over a whole day. This whole day can be chosen so that it is representative of a longer period. In this compound indicator the night value gets a penalty of 10 dB.	24 hrs
L_{den}	Average sound pressure level over all days, evenings and nights in a year. In this compound indicator the evening value gets a penalty of 5 dB and the night value of 10 dB. This is the 'general purpose' indicator defined in EU-directive 2002/49.	Year

Note: * Noise levels refer to the outside façade of buildings if not otherwise specified.
** If sound level meter setting 'fast' is used, which is common.

(⁶) Strictly speaking, the decibel is not a unit but the logarithmic ratio of the sound pressure, in a unit such as pascals, to a standard reference pressure in the same units.

Just as for temperature the unit is the degree Celsius (at least according to ISO), the unit for sound level is the well known decibel, dB ([1]). Although there are exceptions, the levels are normally corrected for the sensitivity of the human ear. This is called the A-weighting, for which reason an 'A' is often added to the dB: dB(A) or dB A. ISO rules now prefer to add the A as a suffix to the indicator, e.g. $L_{A,day}$. In this document all levels are A-weighted.

In order to keep the text as simple as possible, all indicators have been converted to the ones in Table 1.1. This is much helped by the fact that studies show a high correlation between most indicators, and for transport sources after conversion only a small numerical difference remains. Annex V gives background information on this issue.

1.4 Population exposure indicators

To study the effect of noise abatement measures the risk analysis approach as described in Chapter 4 is recommended. The results of this approach are estimates of the (additional) health effects due to noise in the population. Subsequently these can be reduced to one figure by converting the effects to disability adjusted life years (DALY). This can be seen as the health based population indicator.

In Annex V this is shown to be a special case of the more general noise population indicator. In its simplest form this is:

$$L_{den,dwelling} = 10 \lg (\sum n.10 \exp ^{((Lden,i / 10))})$$

Where n is the number of dwellings and $L_{den,i}$ the L_{den} value of each dwelling i.

If the number of inhabitants per dwelling is used in the equation, the total noise load of the population is calculated:

$$L_{den,population} = 10 \lg (\sum n.p.10 \exp^{((Lden,i / 10))})$$

Where p = the number of inhabitants per dwelling.

These overall indicators can be used to rank situations in order to prioritize action plans ([v]). Other population indicators have been used (like the average noise load, or the numbers exposed above a certain value), but they cannot be considered as comprehensive population indicators because they address only part of the population and hence of the problem. These type of indicators might have relevance when used to assist in narrowing policy options to what is politically felt to be a vulnerable part of the population.

2 Health endpoints

As a broad definition of health is used, this paper will cover a relatively large number of relevant endpoints. Some endpoints may also be qualified as intermediary effects. These can be used to assess special situations where the uncertainty in relation to the endpoints in terms of health and wellbeing is large (e.g. noise sources for which exposure-response relationships have not been established).

Table 2.1 Effects of noise on health and wellbeing with sufficient evidence

Effect	Dimension	Acoustic indicator *	Threshold **	Time domain
Annoyance disturbance	Psychosocial, quality of life	L_{den}	42	Chronic
Self-reported sleep disturbance	Quality of life, somatic health	L_{night}	42	Chronic
Learning, memory	Performance	L_{eq}	50	Acute, chronic
Stress hormones	Stress Indicator	L_{max} L_{eq}	NA	Acute, chronic
Sleep (polysomnographic)	Arousal, motility, sleep quality	$L_{max, indoors}$	32	Acute chronic
Reported awakening	Sleep	$SEL_{indoors}$	53	Acute
Reported health	Wellbeing clinical health	L_{den}	50	Chronic
Hypertension	Physiology somatic health	L_{den}	50	Chronic
Ischaemic heart diseases	Clinical health	L_{den}	60	Chronic

Note: * L_{den} and L_{night} are defined as outside exposure levels. L_{max} may be either internal or external as indicated.
** Level above which effects start to occur or start to rise above background.

3 Exposure-response relationships and thresholds for health endpoints

3.1 Annoyance

General

Annoyance is an emotional state connected to feelings of discomfort, anger, depression and helplessness. It is measured by means of the ISO 15666 defined questionnaire. This uses a 11 point numerical scale with end point 'not annoyed' up to 'extremely annoyed'. Although the quantity of highly annoyed as a cut-off point (72 % of scale length) is widely used, relations for annoyed (50 %) and average score are available. Annoyance is source dependant. For transport noises the thresholds are taken to be the same (42 L_{den}), but this is definitely not true for special noise sources like wind-turbines and shunting yards. Dose-effect relations for the transport noises road, rail and air traffic were set out in the EU-Position Paper (2002) ([vi]). Exposure-response relations for other noises (e.g. industry) are not available on EU-level, but useful data is available from other sources ([vii]).

Road traffic noise

The relations provided by the EU-position paper on dose-effect relations ([iii]) have largely been confirmed by later studies. Use of these relations is therefore recommended. The relationship for percentage annoyed (% A, 50 % of the scale) is:

$$\% A = 1.795 * 10^{-4} (L_{den} - 37)^3 + 2.110 * 10^{-2} (L_{den} - 37)^2 + 0.5353 (L_{den} - 37);$$

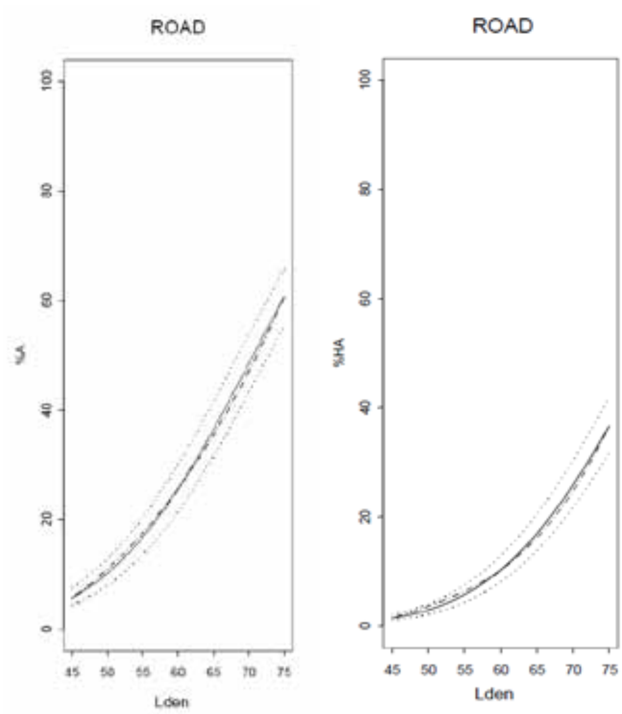
And for highly annoyed (% HA, 72 % of scale length):

$$\% HA = 9.868 * 10^{-4} (L_{den} - 42)^3 - 1.436 * 10^{-2} (L_{den} - 42)^2 + 0.5118 (L_{den} - 42);$$

In Figure 3.1 the relations are illustrated together with their 95 % confidence intervals.

It should be noted that these are average relations for road traffic noise without particular characteristics. Research indicates that some factors may influence the position of the relation. A few of these:

Figure 3.1 % Annoyed (% A) and % highly annoyed (% HA) for road traffic noise with 95 % confidence



- interrupted flow: shifts of + 3 dB are reported;
- increased low frequency noise: increase in annoyance;
- quiet road surfaces: decrease reported (that is, lower annoyance then expected on the basis of the physical decrease alone).

Railway noise

The EU-position paper ([iii]) relations for railway noise are valid for most types of railway. Studies in Japan and Korea (some on high speed lines) sometimes show higher annoyance, but a systematic review is missing.

Percentage of annoyed:

$\% A = 4.538 * 10^{-4} (L_{den} - 37)^3 + 9.482 * 10^{-3} (L_{den} - 37)^2 + 0.2129 (L_{den} - 37);$

Percentage highly annoyed:

$\% HA = 7.239 * 10^{-4} (L_{den} - 42)^3 - 7.851 * 10^{-3} (L_{den} - 42)^2 + 0.1695 (L_{den} - 42);$

In Figure 3.2 the relations are illustrated together with the 95 % confidence intervals. As with road traffic, these are relations for average rail traffic, but particular characteristics may have an influence.
A few known:

- proximity: very close to the tracks (ca 50 meters) relation may be shifted by + 5 dB;
- high speed: relations could be shifted (effect could be partly due to proximity);
- additional low frequency noise or vibrations could increase annoyance (e.g. steel bridges, diesel engines);

Figure 3.2 Percentage annoyed (left) and highlighly annoyed

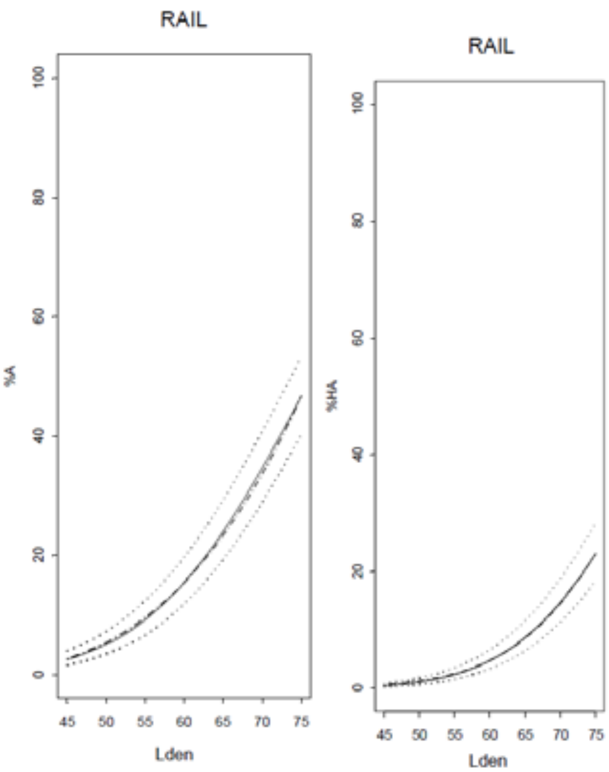
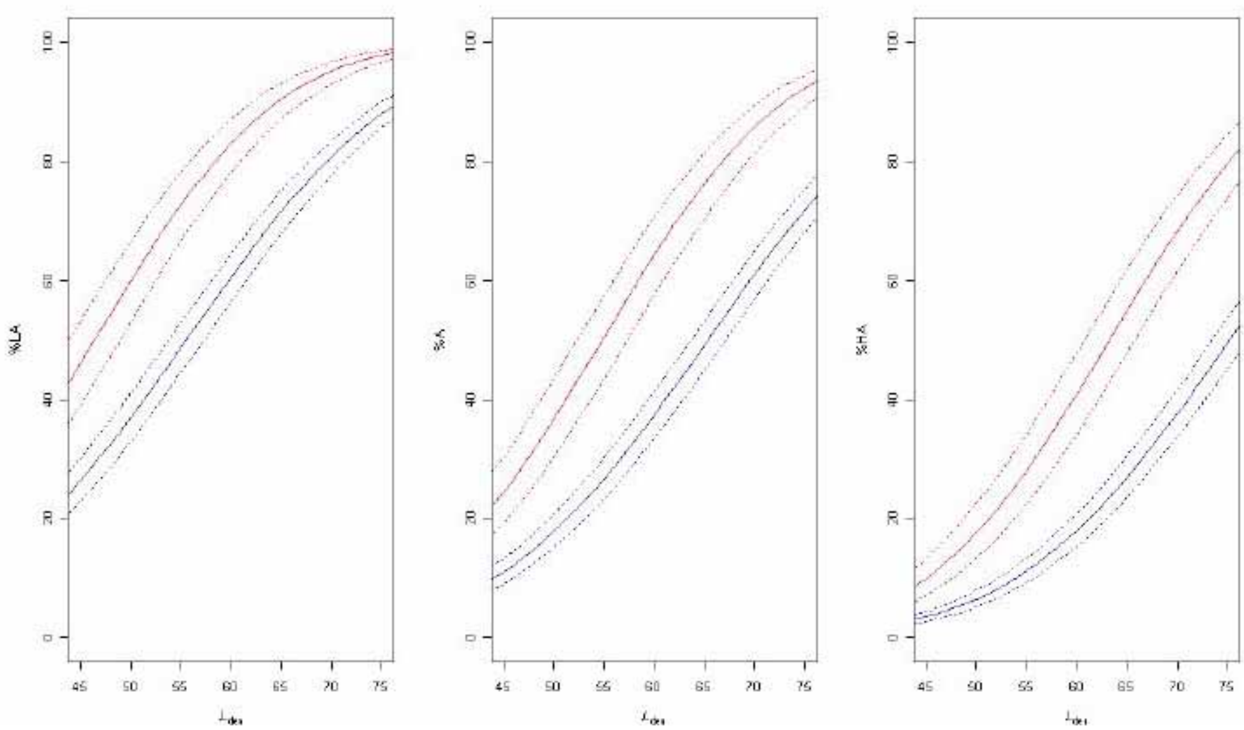


Figure 3.3 % LA, % A, % HA for aircraft noise



Note: Lower curves (blue) pre 1990 dataset, high curves post-1990 dataset.

- share of freight traffic does not seem to have an additional influence;
- squeal noise (such as happens in small radius curves), impulse noise and pure tones increases annoyance.

Aircraft noise

The EU-relations for aircraft noise have been criticized by Guski ([viii]), who noted in series of recent surveys a decrease of the level needed to cause 25 % highly annoyed over time. Subsequent analyses ([iv]) seemed to confirm this, but could not find an explanation. Recent detailed study on the entire dataset failed to find a single cause, but confirmed a trend breach around 1990. This coincides with the introduction of the ISO-standard questionnaire, but it is doubtful that this actually caused the increase. A recent multi-centered study (HYENA) ([x]) showed that this change of annoyance was only found for aircraft noise but not for road traffic noise.

The relations from the EU-position paper are:

Percentage annoyed:

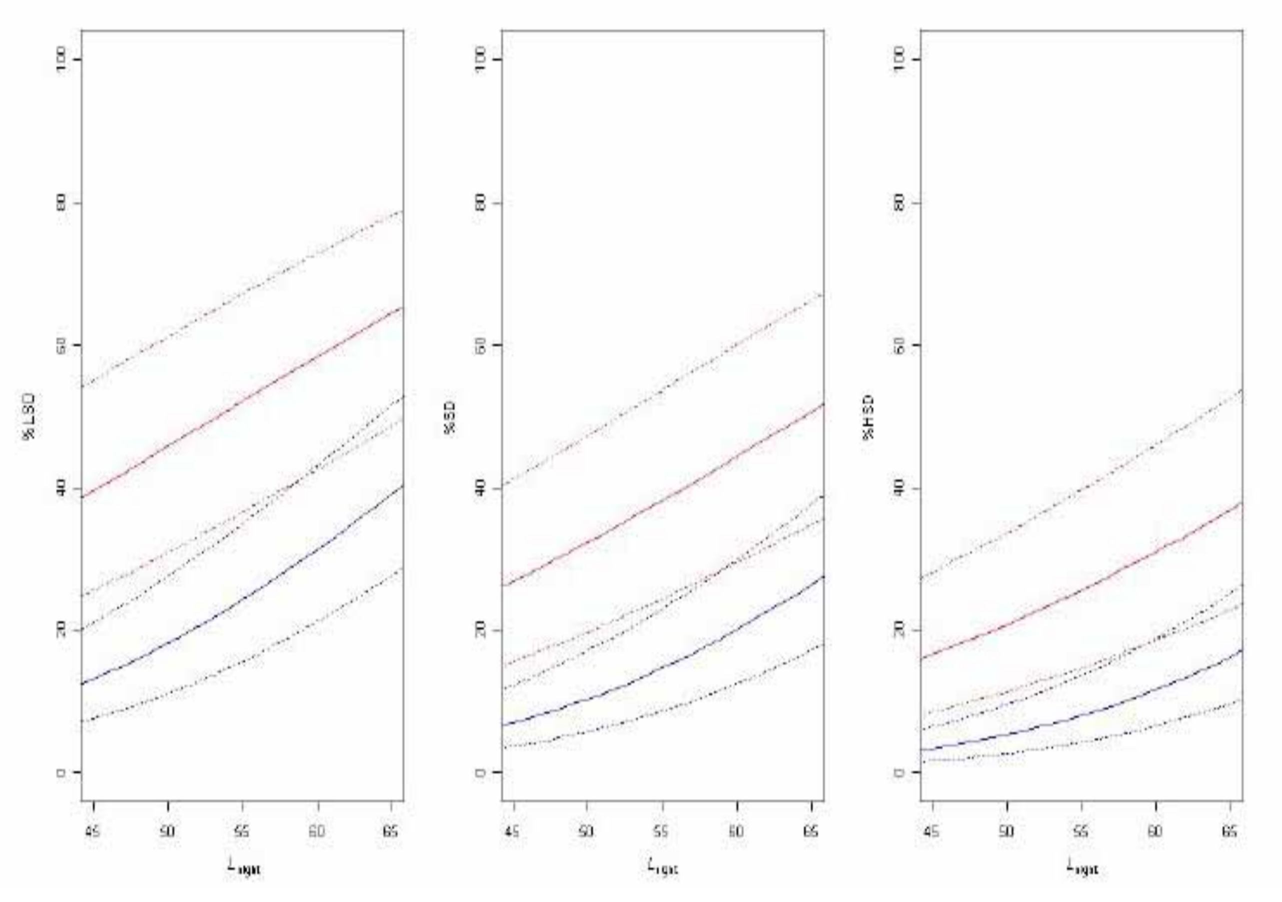
$\% A = 8.588 * 10^{-6} (L_{den} - 37)^3 + 1.777 * 10^{-2} (L_{den} - 37)^2 + 1.221 (L_{den} - 37)$;

Percentage highly annoyed:

$\% HA = -9.199 * 10^{-5} (L_{den} - 42)^3 + 3.932 * 10^{-2} (L_{den} - 42)^2 + 0.2939 (L^{den} - 42)$.

In a recent report an estimate was made of the average of aircraft noise studies carried out after 1990. These were all European studies (Switzerland, Germany, Netherlands) and so may give a better impression for the EU than the pre-1990 studies which are mainly from USA and Australia. Figure 3.3 shows the pre-1990 relations as well as the estimate for post-1990 EU studies. In the annex, the numerical table for the post-1990 relationships is included. Although it is recommended to use the post-1990 data in impact assessment, one should be aware that

Figure 3.4 % LSD, % SD and % HSD for aircraft noise



Note: Lower curves (blue) before 1990 dataset. Dotted lines confidence intervals.

the exact values might change under the influence of further studies. Using the old values in the context of the END would be formally valid, but leads to a conservative approach.

3.2 Sleep disturbance

The WHO-Night Noise Guidelines (2009) ([xii]) discusses in great detail the relations between, noise, sleep quality and health. The report states that sleep is an important biological function and impaired sleep — which is considered a health effect by itself — is related to a number of diseases. Although the function of sleep is still somewhat obscure, sleep deprivation is definitely a condition that deeply afflicts health. Animal experiments show that sleep deprived animals live less, and sleep deprived humans typically show dramatic function loss after a few days. As it can be demonstrated that noise disturbs sleep, the inference is that noise, via the sleep pathway, causes the same diseases. The recommendations are expressed in terms of L_{night} (the night time noise indicator from the END), and the report describes also a number of exposure-response relationships for instantaneous reactions. In part the relationships in the WHO-document are derived from the EU-position paper on night time noise.

3.2.1 Self-reported sleep disturbance

This effect is measured, like annoyance, by questionnaire. Details about the derivation of the relations can be found in the EU-position paper on night time noise ([xiii]).

Road traffic noise:

$$\% \text{ HSD} = 20.8 - 1.05\, L_{night} + 0.01486\, L_{night}^{2}$$

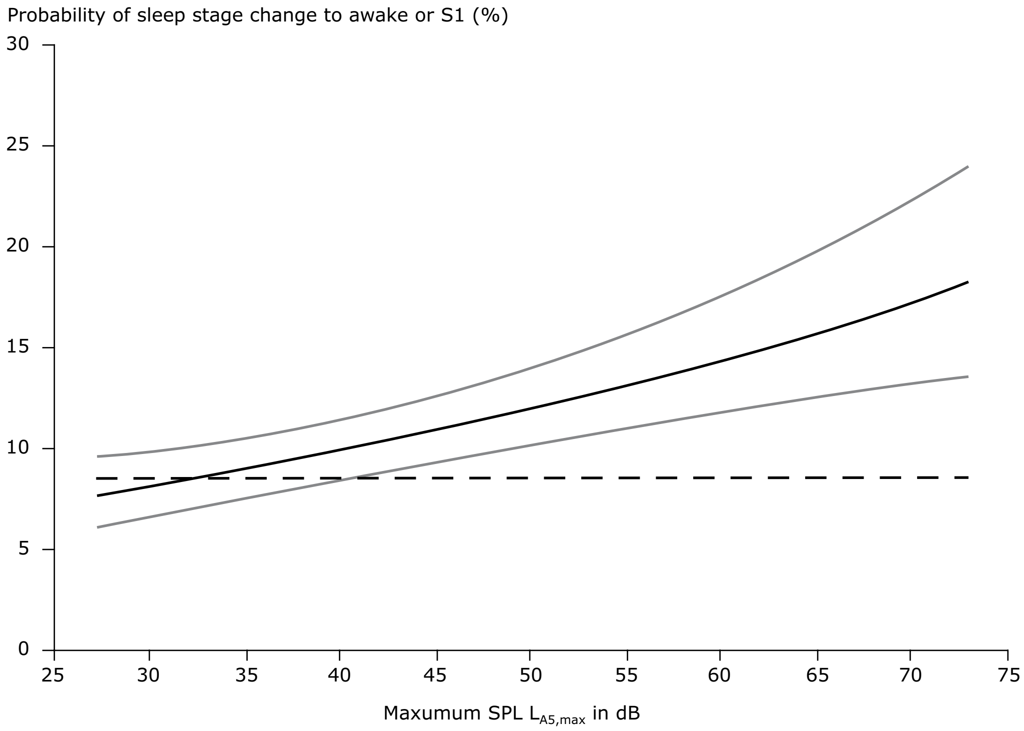
$$\% \text{ SD} = 13.8 - 0.85\, L_{night} + 0.01670\, L_{night}^{2}$$

Railway noise:

$$\% \text{ HSD} = 11.3 - 0.55\, L_{night} + 0.00759\, L_{night}^{2}$$

$$\% \text{ SD} = 12.5 - 0.66\, L_{night} + 0.01121\, L_{night}^{2}$$

Figure 3.5 Probability of sleep stage change to stage S1 or awake depend on maximum SPL $L_{a,max}$



Note: Probability of sleep stage change to stage S1 or awake depend on maximum SPL $L_{a,max}$. Point estimates black line, 95 % confidence gray lines and spontaneous reaction probability without noise (dashed line).

Source: Adopted from: Basner, M., Samel, A. and Isermann, U., 2006. Aircraft noise effects on sleep: Application of a large polysomnographic field study. *Journal of the Acoustical Society of America*, 119(5), 2 772–2 784, with permission from the author.

Aircraft noise

In the former Section 3.1 it was indicated that aircraft noise was found to be more annoying in post-1990 studies. This is also true — though to a lesser extent — for self reported sleep disturbance. The pre-1990 relations as in the EU-position paper[x] are:

$$\% \text{ HSD} = 18.147 - 0.956\, L_{night} + 0.01482\, (L_{night})^2$$

$$\% \text{ SD} = 13.714 - 0.807\, L_{night} + 0.01555\, (L_{night})^2$$

Figure 3.4 shows the difference between the EU-relationships and the estimated post-1990 curves. In the annex, the numerical values for sleep disturbance for aircraft noise for the post-1990 studies can be found.

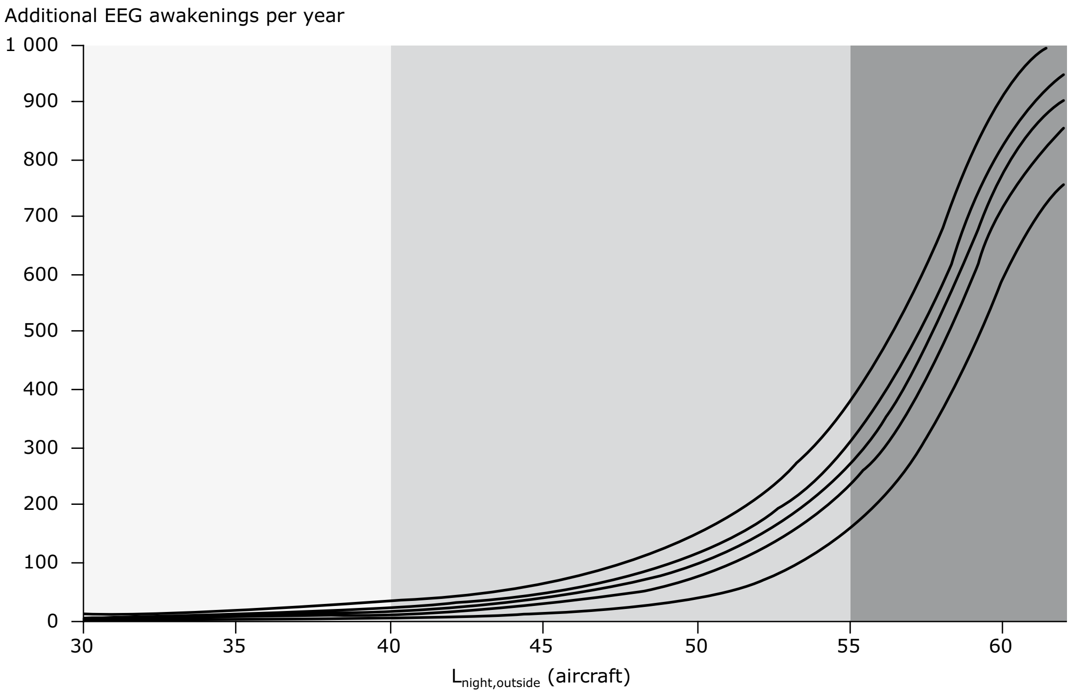
3.2.2 *Polysomnographic sleep (EEG-reactions)*

At very low sound level (33 dB(A) L_{max} as measured inside the bedroom) the body starts to react to intruding sounds. In terms of evolution, this is probably a useful adaptation in the human species. The changes in brainwave pattern are measurable by EEG-machines, and are categorized in arousals, sleep stage changes, or EEG-awakenings (EEG and EMG activations that last for at least 15 seconds which are classified as 'awake').

Although natural biological effects like sleep stage changes or awakenings can not be considered a health effect by themselves they are considered significant early warning signals when the incidence starts to rise above background (spontaneous non-noise related reactions). The best quantitative assessment of EEG-awakenings available is from the DLR-studies into aircraft noise ([xiv]). The curve in Figure 3.5 shows the increase in the probability of a noise induced EEG awakening with L_{max} relative to spontaneous awakenings (circa 24 awakenings usually occur even during undisturbed 8 hour nights).

The exposure-response relationship on the single event level was used to predict the expected degree of sleep fragmentation depending on L_{night} (outside the bedroom), using data from the DLR-field study (Cologne/Bonn airport, 135 nights at 32 measurement locations, inside and outside

Figure 3.6 The average number of additionally aircraft noise induced awakenings per year ([xv])



Note: The average number of additionally aircraft noise induced awakenings per year[xv]. Altogether, 10 million 8-hour nights with 1 to 200 (1, 2, 3,..., 200) noise events randomly drawn from the DLR field study[xiv] were simulated. The lines represent (from below to above) 2.5, 25, 50, 75, and 97.5 percentiles. The gray shaded areas represent Night Noise Guidelines L_{night} ranges (30–40, 40–55, > 55 dB(A).

Source: Applied Acoustics 71(6): 518–222, Basner, M., Müller, U., Griefahn, B. Practical guidance for risk assessment of traffic noise effects on sleep, Copyright 2010, doi:10.1016/j.apacoust.2010.01.002. Reprinted with permission from Elsevier.

noise measurements, and simulation techniques, 1 to 200 noise events randomly drawn, window opening habits 69.3 % tilted windows, 18 % closed windows, 12.6 % open windows) to estimate the average number of EEG awakenings additionally induced by aircraft noise per year ([xv]). This is shown in Figure 3.6 ([xvi]).

3.2.3 Body movements

Like EEG-reactions, an increase in body movements starts at low sound levels. Also in this case the direct health effect is not clear.

The Night Noise guideline report presents relations for instantaneous increases of body movements related to single events, as well as the average increase related to long term exposure. As at the moment this effect cannot be properly used in health assessment, the relations have not been reproduced here.

3.2.4 Reported awakening

Experimental and sociological evidence shows that people awake between 1 and 2 times per night. These awakenings may be defined as reported, conscious, remembered or confirmed awakenings depending from the setting.

Any increase in awakenings is therefore to be taken seriously. As sleep is so important, the organism tends to suppress awakenings. This is the reason why the occurrence remains low even at high noise levels.

The NNGL ([xii]) provides this relation between reported awakenings and noise level:

Percentage of noise-induced awakenings = $-0.564 + 1.909 * 10^{-4} * (SEL_{inside})^2$

3.3 Cardiovascular effects

Ischaemic heart disease (including myocardial infarction) and hypertension (high blood pressure) have been much investigated with respect to noise. The hypothesis that chronic noise affects cardiovascular health is due to the following facts (biological plausibility):

1) Laboratory studies in humans have shown that exposure to acute noise affects the sympathetic and endocrine system, resulting in nonspecific physiological responses (e.g. heart rate, blood pressure, vasoconstriction (the narrowing of the blood vessels), stress hormones, ECG).

2) Noise-induced instantaneous autonomic responses do not only occur in waking hours but also in sleeping subjects even when no EEG awakening is present. They do not fully adapt on a long-term basis although a clear subjective habituation occurs after a few nights.

3) Animal studies have shown that long-term exposure to high noise levels leads to manifest health disorders, including high blood pressure and 'ageing of the heart'.

4) Although effects tend to be diluted in occupational studies due to the 'healthy worker effect', epidemiological studies carried out in the occupational field have shown that employees working in high noise environments are at a higher risk of high blood pressure and myocardial infarction.

3.3.1 Biological factors including stress hormones

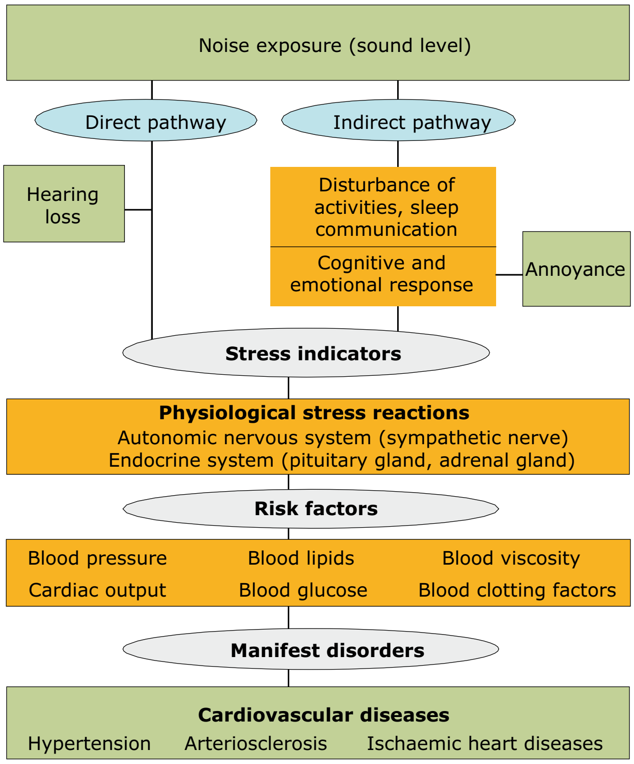
The general stress theory is the rationale for the non-auditory physiological effects of noise. Noise affects the organism either directly through synaptic nervous interactions, or indirectly through the emotional and the cognitive perception of sound. The objective noise exposure (sound level) and the subjective noise exposure (annoyance) may both be interacting predictors in the relationship between noise and health endpoints.

Short-term changes in circulation including blood pressure, heart rate, cardiac output and vasoconstriction as well as the release of stress hormones, including adrenaline and noradrenalin and cortisol have been studied in experimental settings. Classical biological risk factors have been shown to be elevated in subjects who were exposed to high levels of noise.

Acute noise effects do not only occur at high sound levels in occupational settings, but also at relatively low environmental sound levels when certain activities such as concentration, relaxation or sleep are disturbed.

As shown in Figure 3.7, the long-term exposure to noise may lead to health effects through the pressure on the organism via the stress effects. Laboratory, field and animal experiments suggest a biological pathway between the exposure to noise, via the stress mechanism to cardiovascular diseases ([xvii]).

Figure 3.7 Simplified noise effects reaction scheme



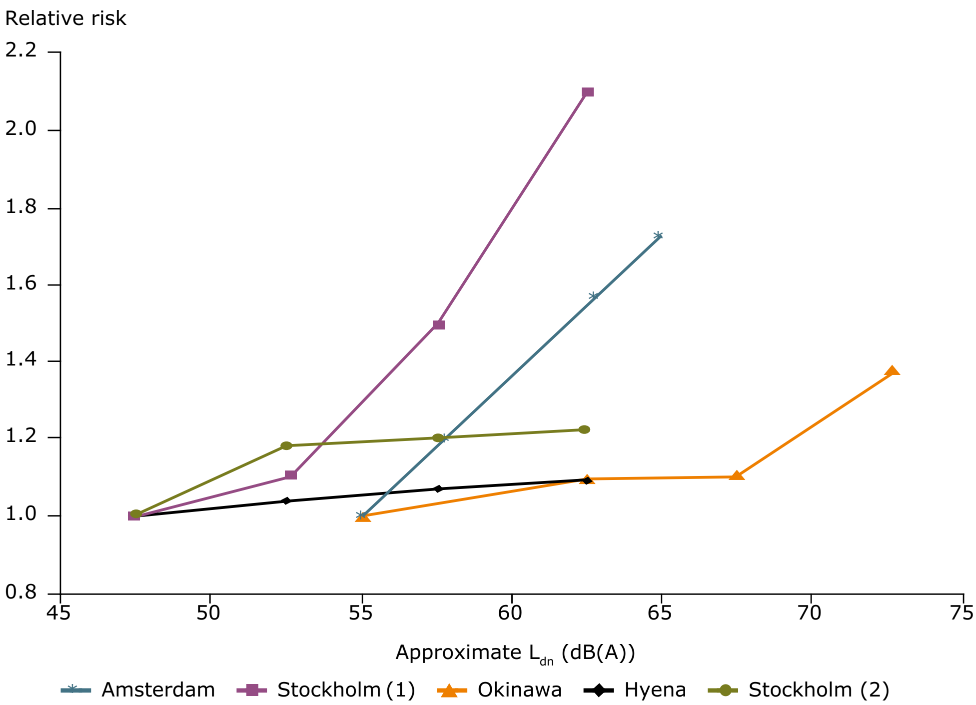
Source: Babisch, W., 2002 (xvii).

Noise either directly or indirectly affects the autonomous nervous system and the endocrine system, which in turn affects the metabolic homeostasis (physiological balance) of the organism, including biological risk factors, and thus increasing the risk for manifest disorders in the long run. Indirect — in this respect — means that the subjective perception of sound, its cognitive interpretation and the available coping abilities play a role in physiological reaction. Direct, on the other hand, means that the activation of the regulatory system is determined by direct interaction of the acoustic nerve with other parts of the central nervous system (e.g. hypothalamus, amygdala). This is particularly relevant during sleep, where autonomous responses to single noise events, including changes in blood pressure and heart rate, have been shown in subjects who were subjectively not sleep disturbed (xviii).

3.3.2 Hypertension

Based on a meta-analysis, an exposure-response function is derived from pooled data of 5 aircraft noise studies, see Figure 3.4. Since this effect estimate is based on different studies with different noise level ranges, no clear cut-level for the onset of the increase in risk can be given. It is therefore

Figure 3.8 Relative risk for hypertension in 5 studies



suggested to use either $L_{den} \leq 50$ or $L_{den} \leq 55$ dB(A) (2) as a reference category (relative risk = 1). The respective relative risks for subjects who live in areas where L_{den} is between 55 to 60 dB(A) and between 60 to 65 dB(A) would then approximate to 1.13 and 1.20, or 1.06 and 1.13, respectively (xix).

Exposure-response function for hypertension:

OR per 10 dB(A) = 1.13, 95 % CI = 1.00 – 1.28, range = 50–70 dB.

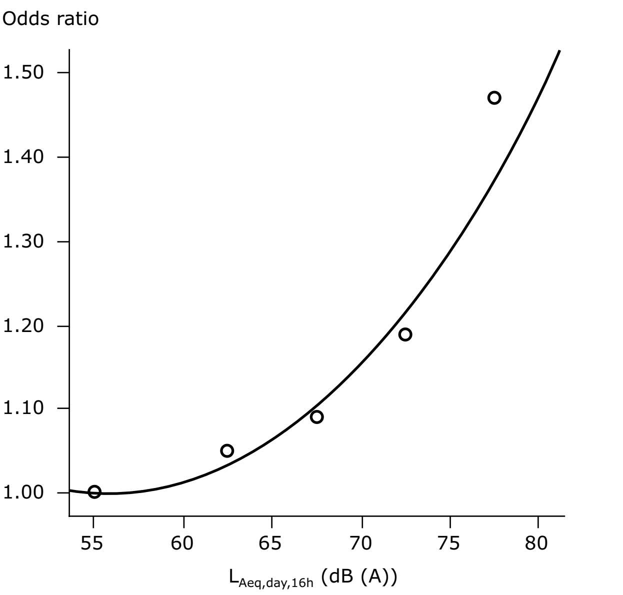
3.3.3 Ischaemic heart disease (IHD)

Similarly, for myocardial infarction (MI) the results are based on pooled data for 5 studies, in this case road traffic noise. The studies considered in the meta-analysis (xx) refer to myocardial infarction as a marker of ischaemic heart diseases (IHD). A cubic exposure-response function and a linear trend function are given for the increase in risk per increment of the noise level. Annex III shows the relative risks (odds ratios) for single noise levels.

Exposure-response function for myocardial infarction:

Cubic model: $OR = 1.629657 - 0.000613 * (L_{day,16h})^2 + 0.000007357 * (L_{day,16h})^3$, $R^2 = 0.96$.

Figure 3.9 Odds ratio for myocardial infarction



Conversions L_{den} = = > $L_{day,16h}$ (for inclusion into the formula) can be made using Annex III and the following approximation: $L_{day,16h} = L_{den} - 2$ dB(A) for urban road traffic.

$L_{day,16h} \approx L_{den} \leq 60$ dB(A) is considered as a reference category (relative risk = 1).

3.4 Cognitive impairment

A number of laboratory studies indicate that noise may influence learning and performance, but the relation is complex, as people usually try to keep performance up. This kind of research was primarily carried out in schoolchildren. The RANCH study, first convincingly demonstrated in a multinational field study that there is relation between learning (measured as reading ability) and noise exposure (xxi). Figure 3.10 shows the association between the average noise level outside schools during lesson hours and a standardized score ('z-score') of a standardized reading test in children (reading comprehension). The higher the score the better is the performance of the children in the reading test.

From this and other studies Hygge derived a hypothetical exposure-response for percentage cognitively affected (xxii). This is shown in Figure 3.11, showing an increase in the risk of cognitive impairment with increasing noise exposure assuming that 100 % of the noise exposed are cognitively affected at a very high noise level, e.g. 95 L_{dn}, and that none are affected at a safely low level, e.g. 50 L_{dn}. A straight line (linear accumulation) connecting these two points, can be used as basis for approximation. The cut-off level here is $L_{den} = 50$ dB[5]. This straight line is an underestimation of the real effect. Since for theoretical reasons based on an (assumed) underlying normal distribution, the true curve should have the same sigmoidal function form as the two curves in the figure. Within the noise exposure bracket 55–65 L_{dn} the straight line and the solid line sigmoidal distribution agree on approximately 20 % impairment. In the bracket 65–75 L_{dn} the number should be in the range of 45–50 % and above 75 L_{dn} in the range of 70–85 % according to the results of individual studies. It should be noted that the RANCH study revealed deficits in cognitive performance at aircraft noise levels even below ~ 50 dB(A) (3).

(2) Assumed that $L_{dn} \approx L_{den}$.
(3) Ranch noise data refer partly to $L_{day,16h}$ and measured sound levels during school-times.

Figure 3.10 Association between average aircraft noise level outside schools (L_{Aeq}) and reading comprehension in schoolchildren

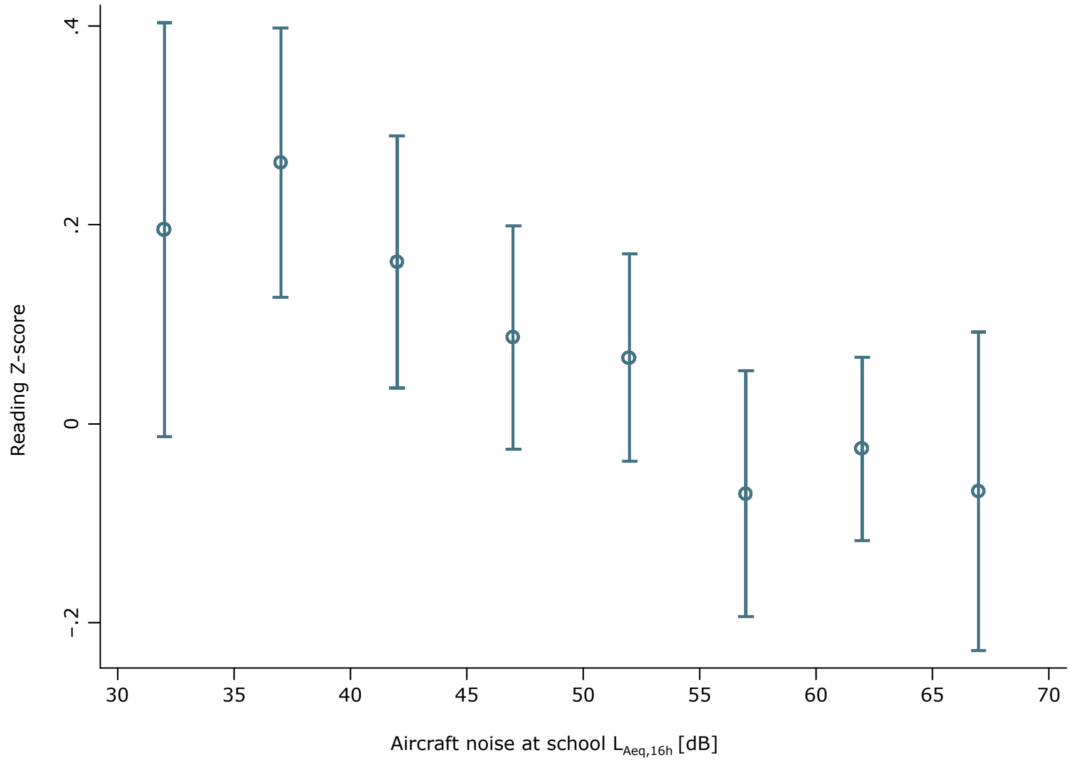
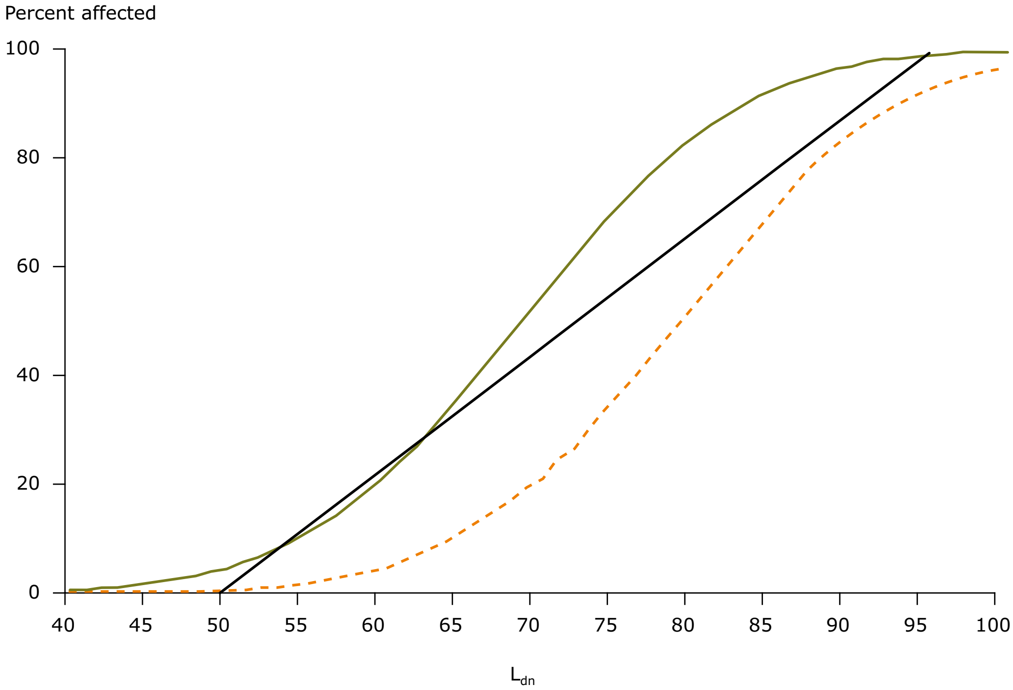


Figure 3.11 Hypothetical association between aircraft noise level (L_{dn}) and cognition impairment in schoolchildren (Hygge)



4 Risk assessment

In a serious document with the witty subtitle 'Death, DALY's or Dollars', de Hollander ([xxiii]) showed that for the political decision process it does not matter very much if environmental impact is evaluated in terms of money, health or mortality risk. The choice between one method or another depends on cultural and/or political preferences. In this chapter recommendations are given for risk assessment based on health and on monetary valuation.

From the technical point of view, validated methods are available to assess environmental impact of an activity. Even if it is not possible to assess absolute impacts, at least the ranking order of the alternatives can be established.

The following formula defines the assessment of the *attributive fraction* (WHO). The attributive fraction (other terms: impact fraction, population attributable risk) describes the reduction in disease incidence that would be observed if the population were entirely unexposed, compared with its current (actual) exposure pattern.

$AF = \{\Sigma (P_i * RR_i) – 1\} / \Sigma (P_i * RR_i)$

where: AF = Attributive Fraction

P_i = Proportion of the population in exposure category i

RR_i = relative risk at exposure category i compared to the reference level.

An example may explain this more clearly.

In Table 4.1 the relative risk of myocardial infarction is calculated for the German population, based on a probabilistic estimate of exposure for the year 1999 and the exposure-response curve shown in Section 3.3.3. This leads to the conclusion that 2.9 % of MI cases (people with MI incidence) may be due to road traffic noise per year in Germany. Using general health statistics of the annual incidence rate of cases of ischaemic heart diseases (IHD), including myocardial infarction (MI), it was estimated that approximately 3 900 MI cases (or 24 700 IHD cases if the exposure-response curve is extrapolated to all IHD cases) would be due to the road traffic noise in that year.

4.1 Evaluation using disability-adjusted life years (DALY)

The DALY (and QALY) was developed by WHO and the World Bank to enable policy makers to make rational choices for medical treatment. To do this each clinical phenomenon is assessed to establish a weighting factor. According to the protocol designed to assess these weights, the factor takes into account

Table 4.1 Example of the use of the exposure-response curve for myocardial infarction and representative noise exposure data from Germany for the calculation of the attributive fraction of myocardial infarction due to road traffic noise

Average sound pressure level during the day (6-22 h) $L_{day,16hr}$ [dB(A)]*	Percentage exposed (%)	Relative risk of myocardial infarction (OR) *)
< = 60	69.1	1.000
> 60–65	15.3	1.031
> 65–70	9.0	1.099
> 70–75	5.1	1.211
>75	1.5	1.372

Note: * These calculations can also be made using END noise mapping data. In the textbox an example of such a calculation is presented.

In accordance with the Environmental Noise Directive (END) (Directive 2002/49/EC, 2002), the EU Member States have produced a large scale inventory of the noise situation in their area. The data were sent to the Commission and can be viewed on the Noise Observation and Information Service for Europe: http://noise.eionet.europa.eu/index.html.

The rough data for road traffic noise exposure are derived by combining exposures in major agglomerations with those from major roads outside agglomerations, and some additional published (but not reported) data from agglomerations is also added. At this stage the data covers 17 % of the total EU population. It is estimated from Austrian and Dutch data — where estimates are available for the entire population — that total exposure at least doubles if the relation in these countries between agglomeration results and country wide inventory are extrapolated to the total EU-population of 497 million. With this data it is possible to calculate, for example, the people highly disturbed in their sleep by road traffic noise in the EU.

Table 4.2 Number of people (in millions) exposed to L_{night}-classes in dB as reported by EU Member states 2009 and the calculated number of highly sleep disturbed

	50–54	55–59	60–64	65–70	> 70
Exposed	34	17	9	2	0.3
Number of highly sleep disturbed	2.4	1.6	1.2	0.3	0.066

mortality, (loss of) mobility, self-care, daily activities, pain/discomfort, anxiety/depression and cognitive function.

In principle the DALY is calculated as the sum of years of potential life lost due to premature mortality and the years of productive life lost due to disability:

DALY = YLL + YLD

YLL = ND (number of deaths) x DW (disability weight) x LD (standard life expectancy at age of death in years)

YLD = NI (number of incident cases) x DW (disability weight) x LI (average duration of disability in years)

Although the procedure is not without critics, it can be used to rank policy alternatives. One critical point is the choice of the disability weights. It can be suspected that different cultures or even different population groups come to different weights. Studies indicate that there is such an effect, but seems to be of modest magnitude. It is however important that weights be assessed according to the already mentioned protocol by medically experienced staff. Below is a list of relevant disability weights for the scope of this document.

These examples — see also preceding sections -also highlight the problem of evaluating the data. The number of people with myocardial infarction (cases) is relatively low, while the number of sleep disturbed (and annoyed) people is high. Another

Table 4.3

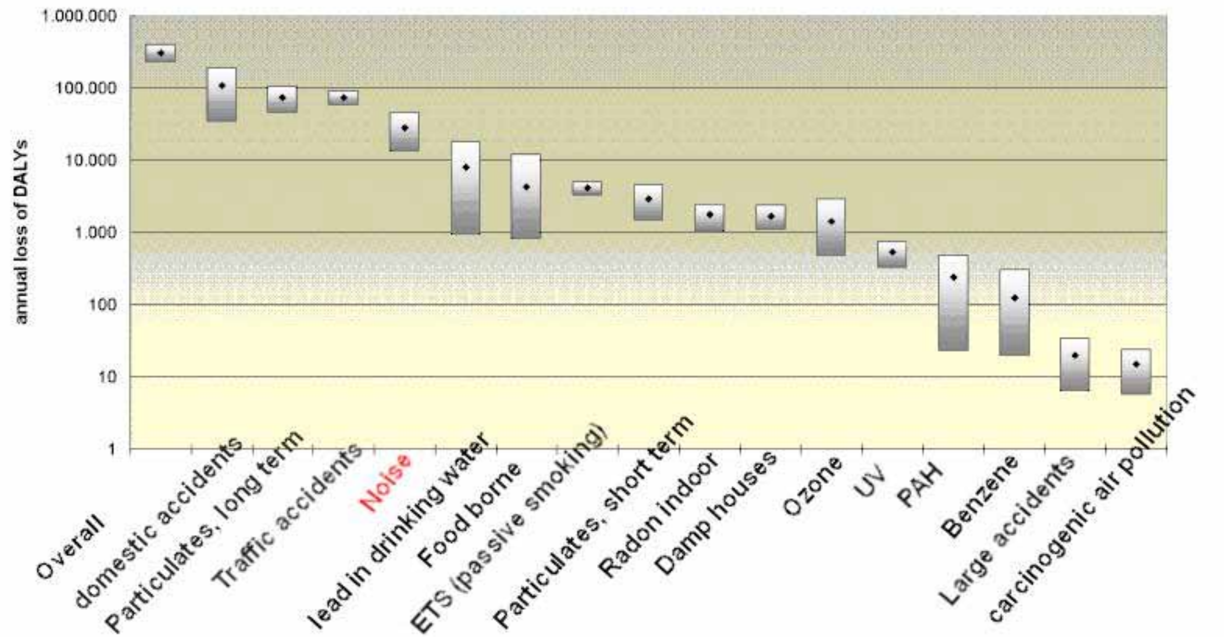
Health condition	Disability weight
Mortality	1.000
Non-fatal acute myocardial infarction	0.406 (WHO)
Ischaemic heart disease	0.350 (de Hollander, 1999)
High blood pressure	0.352 (Mathers, 1999)
Primary insomnia	0.100 (WHO, 2007)
Sleep disturbance	0.070 (WHO, 2009)
Annoyance	0.020 WHO (preliminary) 0.010 (Stassen, 2008); 0.033 (Müller-Wenk, 2005)
Cognitive impairment	0.006 (Hygge, 2009)

Note: Using these weights some Member States proceeded to carry out the calculations. Examples of the outcomes are listed in Table 4.4.

Table 4.4 Examples of DALY calculations in three countries

Example:	Severe annoyance Netherlands	Cognitive effects Sweden	Ischaemic heart diseases Germany
Subjects	Adults	Children 7–19 years	Adults
Total population	Ca. 14 mio	Ca. 1.5 mio	Ca. 70 mio
Exposure	Empara	EU estimate	Probabilistic UBA
Reference year	2000	2000	1999
Disability weight	0.02	0.006	0.350
DALYs per million of people	1 203	648	361

Figure 4.1 Estimate of DALY's from different environmental aspects



important observation is that the bulk of the affected population is in the medium high exposure range.

Using exposure data from the Netherlands, RIVM made a comparison in burden of disease from several environmental exposures ([xxiv]). Although the uncertainty in these estimates is large, it does provide a useful insight.

Recently the authors of this study stated that Environmental DALYs allow comparative evaluation of the environmental health risks of a multitude of pollutants and, consequently, the setting of priorities. The use of DALYs may also improve risk communication as their number can be expressed as a fraction of total burden of disease ([xxv]).

At the same time they caution against indiscriminate use (eg in small populations) and point to ethical objections. As these apply also to other evaluation systems (like cost-benefit analysis), the sensible advice is to discuss these with the partners in the project at hand.

4.2 Cost-benefit

Cost benefit analysis is often a standard procedure in policy making, and in the European Commission this is mandatory. A good example in the noise field is latest decision on tyre noise. The report from FEHRL ([xxvi]) leaned heavily on a cost-benefit analysis of this issue. Using the — modest — rate of 25 EUR/decibel/household/year, it was demonstrated that quieter tyres could produce benefits to the public of between EUR 48 and 123 billion in the period 2010–2022. Although in the end many other factors played a role (like safety), it was shown that the benefits largely exceeded the costs and this contributed to a decision taken in favour of the reduction of tyre noise.

An important factor in carrying out the analysis is an estimate of the benefits. Currently there are two methods for which sufficient proof is available: Contingent valuation and Hedonic pricing.

The European Commission Working Group Health and Socio-Economic Aspects (WG-HSEA) provided the position paper 'Valuation of noise' ([xxvii]) based on the willingness to pay data from Navrud (2002) ([xxviii]). The paper recommends the use of a benefit of EUR 25 per household per decibel per year above noise levels of L_{den} = 50–55 dB. Purchasing power parity (PPP) indices could be used to adjust the values for use in accession states. These are published indices which adjust the exchange rates between countries by differences in the cost of living. Even though this figure has been criticized as being too low, it appears that most noise abatement measures have a positive cost benefit ratio, as was demonstrated in the tyre noise study which used this figure.

Hedonic pricing data come from studies of the real estate markets: it is found that properties exposed to higher noise levels will have a lower value on the market than a similar building exposed to a lower noise level. This is valid for residential houses (for which there is extensive literature) but probably also for office buildings. The best estimate is that house prices loose 0.5 % of their value per decibel over 50–55 L_{den}. The range of research results is between 0.2 % and 1.5 %, with a tendency for higher values for aircraft noise.

5 Quality targets

Many countries have some form of noise policy and management, and noise limit values form the basis of a noise control system. A number of studies address the comparison of these limit values to see if a mutual understanding exists for a common noise quality level. Starting with the study by INRETS ([xxix]) for the preparation of the EU Green Paper, the paper by late Dieter Gottlob ([xxx]) and recently the International Institute of Noise Control Engineering ([xxxi]), all these studies show that this is not an easy task. Firstly the noise indicators differ considerably, and secondly the actual enforcement of the limits or actions taken may be quite different.

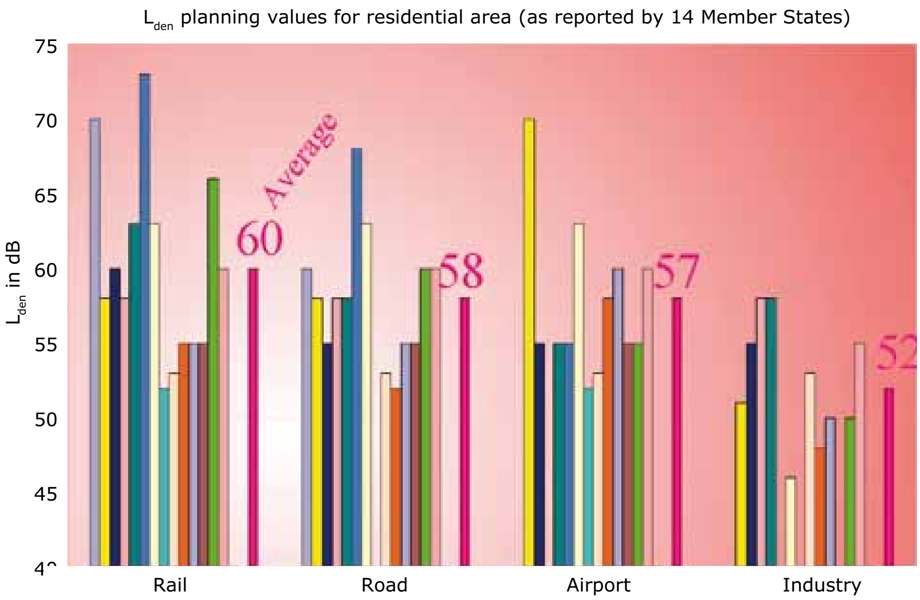
Recently the EU Member States reported to the Commission any relevant limit values in force or under preparation as requested by the END Art. 5.4. These have to be expressed in terms of L_{den} and L_{night}, eliminating at least one important source of bias. Figure 5.1 shows a comparison of L_{den} for planning purposes for residential areas from 14 Member States.

The limit values reported by this selection of Member States have often a long history, so the variation in values is not too surprising. It is interesting that limit values for industry are significantly lower, and for railway noise higher. The average difference is 8 dB, and the maximum limit value is 57 dB for industrial noise, and 73 dB for railway noise.

The Community Guidelines 2000 from the WHO ([xxxii]) recommends 50/55 $L_{Aeq,\ 16\ hrs}$ as health based threshold, which is in line with earlier recommendations and guidance from ISO and national and international environment agencies. Although more than half of the L_{den} limit values is close to these health based guidelines, some are considerably higher.

The same goes for the night time levels, although it seems that '10 dB lower' rule is almost universally adopted: the averages are L_{night} = 50 dB for railway and road noise, 46 for aircraft noise and 42 for industry.

Figure 5.1 Comparison L_{den} limit values



Source: Data reported in accordance with END up to 2009.

The recently issued WHO Night Noise Guidelines expanded the Community guidelines on the issue of sleep disturbance, and concluded that although biological effects kick in as low as L_{night} = 30 dB, L_{night} = 40 dB should be an adequate health protection value, but also recommends an 'interim target' of 55 L_{night}. An $L_{night,outdoor}$ of 30 dB is considered as LOEL (lowest observed effect level) and an $L_{night,outdoor}$ of 40 dB as LOAEL (lowest observed adverse effect level). The NNG uses the default year average insulation value of 21 dB, which is based on the well known fact that a large part of the population keep the windows (slightly) open for at least half of the time. In Annex II guidance is provided to calculate the year average in specific circumstances. The CALM network[xxxiii] considered L_{den}/L_{night} values of 50/40 dB as an optimum target that is defensible.

In conclusion, from the broad overview of the limit values in a large number of countries, and from the scientific evidence, as well as from some more political organisations, there seems to be a consensus that L_{den} around 50 dB (or the equivalent level in other units) would represent a good noise quality, and L_{night} < 55 dB should be respected to protect the population from serious health effects.

6 Implications for END

6.1 Sustainable action planning

All Action Plans should be sustainable in nature, in that the economic, social and environmental impact of the plans should be considered during their development. Although the information provided in this Good Practice Guide is not entirely new, it brings together in one place, information and tools from many sources that can be used to assess impacts of noise on people's quality of life, on their health, and on quality of the living environment, including economic costs. This information can feed into the development of a sustainable Noise Action Plan.

6.1.1 Dose-effect relationships

As per Annex III of the END, 'dose-effect' relations should be used to assess the effects of noise on populations. This assessment can in turn highlight the potential harmful effects of noise in the population (art. 6.3), which should then inform the 'action planning' process in order to mitigate or reduce any harm to the population under consideration.

Chapter 3 of this Guide sets out charts and algorithms by which annoyance, sleep disturbance, noise induced awaking, hypertension, and heart disease can be estimated in relation to L_{den} or L_{night} or SEL values. These are valuable indices for converting something abstract such as an Lden value into something that has meaning to most of the public. Knowing that between 4 and 27 % of the population exposed to an Lden level of 55 dB(A) from traffic sources will be highly annoyed, has much more meaning and is more tangible than just knowing the number of people exposed to that sound value.

Annex V. (3) in relation to minimum requirements for action planning states each action plan *should contain estimates in terms of the reduction of the number of people affected (annoyed, sleep disturbed or other)*. Therefore it is recommended that where possible these dose effect relationships should be referred to and used when drafting Noise Action Plans.

6.1.2 Quiet areas

Currently 'Quiet Areas' in agglomerations are defined by 'an appropriate noise indicator such as L_{den}'. If the aim of identifying quiet areas is to maintain or provide areas of calm or respite from noise, then perhaps one of the aims of an action plan would be to identify and quantify the number of people who benefit in terms of annoyance or improvement of the quality of the living environment. The development of 'Annoyance Maps' along with noise maps is also a possibility which could add an extra and meaningful dimension to any action plan.

6.1.3 Resource prioritisation

In most cases the development of noise action plans involves competing for limited resources. Chapter 4 sets out methods and figures for carrying out cost

Table 6.1 Comparison of L_{den} values for different sources with respect to annoyance

Percentages of highly annoyed

L_{den}	Road	Rail	Aircraft (revised estimate)	Industry	Windturbine
55 dB	6 %	4 %	27 %	5 %	26 %
50 dB	4 %	2 %	18 %	3 %	13 %
45 dB	1 %	0 %	12 %	1 %	6 %

benefit analysis and evaluations using 'disability adjusted life years', (DALY). With these parameters it should be possible to make a reasonable estimate as to the true costs of noise and to rank and prioritise noise against other environmental impacts. These parameters can put noise within an environmental context by which coherent and understandable arguments can be made for adequate resources to develop and carry out action plans.

6.1.4 END threshold values

The END threshold values are above the levels where effects start to occur as shown in the previous chapters. The lower thresholds for mapping (55 dB L_{den} and 50 dB L_{night}) delimit the area where the noise is considered to be a problem. As a first step this is understandable, as this kind of mega-scale mapping exercise is unique. Beyond that, the Member States are free to choose their own threshold from where to start action planning.

The WHO Night Noise guidelines give a clear advice
that from the health point of view the calculations of night time burden should start at 40 dB L_{night} and that action planning should at least contain actions to bring down level below 55 dB L_{night}. Lowering the actual threshold of L_{night} = 50 dB to L_{night} = 40 dB would give a better understanding of the magnitude of the problem, and consequently a better allocation of efforts.

6.1.5 Comparison of Sound Sources with respect to annoyance

As for the L_{den} threshold of 55 dB, it should be noted that this does not take into account the differences that exist between sources. L_{den} = 55 dB is a fair threshold for railway noise, but for other sources this leads to an underestimate of the actual burden. Table 6.1 shows the percentage of highly annoyed related to threshold values of 45, 50, and 55 dB L_{den}.

This shows that a sensible approach for action planning is to make a distinction between sources when assessing the magnitude of the impact on the population.

Again it should be highlighted that when using the methods, charts, algorithms and figures set out in this Good Practice Guide, there should be a certain amount of caution against indiscriminate use (e.g. in small populations) and alertness to ethical objections when dealing with DALY's, vulnerable groups and exposure to extreme noise levels. The sensible advice is to discuss these with the partners in the project at hand.

7 References and further reading

The amount of literature on noise and health is overwhelming, and in the text only the most important references are presented. Below the most relevant references to obtain more information are printed in **bold**.

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List of abbreviations

AF	Attributive fraction
CALM	Community Noise Research Strategy Plan
CI	Confidence interval
DALY	Disability adjusted life year
dB	Decibel
dB(A)	Decibel (A-weighted)
DLR	German Aerospace Center
DW	Disability weight
EC	European Commission
ECG	Electrocardiogram
EEA	European Environment Agency
EEG	Electroencephalogram
EMG	Electromyogram
END	Environmental Noise Directive (EU 2002/49)
EPA	Environmental Protection Agency of the United States of America
EU	European Union
FEHRL	Forum of European National Highway Research Laboratories
INRETS	French National Institute For Transport And Safety Research
IHD	Ischaemic heart disease
ISO	International Standardization Organization
JRC	Joint Research Centre (of the European Commission)
L	Sound level indicator
L_A	A-weighted sound level indicator
L_{den}	day-evening-night equivalent sound level
L_{night}	night equivalent sound level
LOEL	Lowest Observed Effect Level)
LOAEL	Lowest Observed Adverse Effect Level)
Max	Maximum
MI	Myocardial infarction
MS	Member State (of the European Union)
NNGL	Night Noise Guidelines for Europe (WHO)
OR	Odds ratio (estimate of the relative risk)
PPP	Purchasing Power Parity
QUALY	Quality adjusted life year
R^2	Coefficient of determination
RR	Relative risk
SEL	Sound exposure level
UBA	Umweltbundesamt (Federal Environment Agency, Germany)
WHO	World Health Organization

Annex I Alternative health definitions

- 'Good health is a major resource for social, economic and personal development and an important dimension of quality of life (WHO, 1986)'.
- Prerequisites for health: 'The fundamental conditions and resources for health are peace, shelter, education, food, income, a stable ecosystem, sustainable resources, social justice and equity. Improvement in health requires a secure foundation in these basic prerequisites (WHO, 1986)'.
- Advocate: 'Good health is a major resource for social, economic and personal development and an important dimension of quality of life. Political, economic, social, cultural, environmental, behavioural and biological factors can all favour health or be harmful to it. Health promotion action aims at making these conditions favourable through advocacy for health (WHO, 1986)'.
- 'Good health and well-being require a clean and harmonious environment in which physical, psychological, social and aesthetic factors are all given their due importance. The environment should be regarded as a resource for improving living conditions and increasing well-being (WHO, 1989)'.
- Adverse effects '… Change in morphology, physiology, growth, development or life span of an organism, which results in impairment of the functional capacity to compensate for additional stress or increase in susceptibility to the harmful effect of other environmental influences (WHO 1994)'.
- 'People are healthy until they are deemed not to be so. The relative health can be determined by comparative population measures of mortality, morbidity and impairment (Morell, 1997)'.
- 'Health is a dynamic condition of the organism which functions properly and mentally according to the individual's age, sex and general conditions of the population to which the individual belongs, and the current state of science and technology and the related objectives of health care and public health, the beliefs and the cultural patterns of society (Dutch Health Council, 1997)'.

Annex II Practical guidance on conversions

i. SEL (L_{AE}) to L_{Amax}

According to ground based measurements, the relation between SEL and Lmax for aircraft noise is:

SEL = 23.9 + 0.81 * L_{Amax}.[1]

A more general approach can be used to estimate SEL for transportation noise.

If the shape of the time pattern of the sound level can be approximated by a block form, then SEL ≈ L_{max} + 10 lg (t), where t (in seconds) is the duration of the noise event. This rule can be used inter alia for a long freight train that passes at a short distance. When t is in the range from 3 to *30 s, then SEL is 5 to 15 dB(A) higher than L_{max}. For most passages of aircraft, road vehicles or trains, the shape of the time pattern of the sound level can be better approximated with a triangle. If the sound level increase with rate a (in dB(A)/s), thereafter is at its maximum for a short duration before it decreases with rate — a, then SEL ≈ L_{max} – 10 lg (a) + 9.4. Depending on the distance to the source, for most dwellings near transportation sources the rate of increase is in the order of a few dB(A)/s up to 5 dB(A)/s. When a is in the range from 9 to 1 dB(A)/s, then SEL is 0 to 9 dB(A) higher than L_{max}.

ii. From outdoor levels to indoor exposure

As the L_{night} is an annual value, the insulation value is also to be expressed as such. This means that if the insulation value is 30 dB with windows closed and 15 dB with windows open, the resulting value is 18 dB if the window is open 50 % of the time. If these windows are closed only 10 % of the time, the result is little more than 15 dB. The issue is complicated by the fact that closing behaviour is, to a certain extent, dependent on noise level.

When data about effects are expressed with indoor noise levels (i.e. inside bedrooms) as the parameter, they need to be converted to L_{night}., in accordance with the END definition. The most important assumption is the correction from inside levels to outside levels. An average level difference of 21 has been chosen, as this takes into account that even in well-insulated houses windows may be open a better part of the year. Therefore:

$L_{night} = L_{night,inside}$ + Y dB[3]

Y is the year average insulation value of the (bedroom) facade. In the EU-position paper on night time noise a default insulation value of 21 dB was used.

To convert to other insulation values, the following method should be used. As the L_{night} is expressed as a year average, also the insulation should be expressed as a year average, to be calculated as:

Δ year = -10 * lg (T_{closed} / 365 * $10^{-\Delta \text{ closed} / 10}$ + T_{open} / 365 * $10^{-\Delta\text{open} / 10}$)

In which

Δ closed = insulation with windows closed

Δ open = insulation with windows open

T_{closed} = number of nights with windows open

T_{open} = number of nights with windows open

Default values for insulation for windows closed depend very much on building practices (single, double or even triple glazing). Open windows usually give attenuation from 5–10 dB, slight open windows 10–15 dB. Surveys indicate that windows may be kept open (or half-open) for more than half of the time, even in colder climates (75 % in the Netherlands).

iii $L_{eq,16\ hrs}$ and L_{dn} to L_{den}

The conversion between L_{dn} and L_{den} is relatively straightforward as the basis is almost the same: the L_{dn} is like L_{den} a year average composed from the day (7.00 to 22.00 hrs) L_{eq} and night (22.00 to 07.00) L_{eq} + 10, in the L_{den} the evening is inserted (default 19.00–23.00 hrs) and the night is 8 hrs instead of 9. Although the maximum deviation can be 6 dB (only evening noise), real world differences are between 0 and 1 dB. A representative sample of urban roads showed consistent differences of $L_{den} - L_{dn}$ of 0.3 dB.

The $L_{eq,16\ hrs}$ is less well defined, but usually is a taken to be a 16 hr L_{eq} from 07.00 to 23.00 or 06.00 to 22.00 for a representative situation. For road traffic this is normally an average work-day traffic load. This may differ slightly from the year average traffic. For other sources the difference can be much larger, eg industry.

The second source of differences is the night period (and penalty) in the L_{den}. This can cause very large differences of course, as $L_{eq,16hrs}$ does not cover the night time.

Again a representative sample of urban (main) roads shows differences of $L_{den} - L_{eq,16\ hrs}$ of ~ 2 dB. See also: Bite, M., & Bite, P. Z. (2004). Zusammenhang zwischen den Straßenverkehrslärmindizes LA_{eq}-(06–22) und LA_{eq}(22–06) sowie L_{den}. Zeitschrift für Lärmbekämpfung, 51, 27–28.

Annex III Exposure-response relations between aircraft noise and annoyance due to aircraft noise, average of post 1996 studies

Studies 1996 and later: % LA, A %, % HA and their 95 % confidence limits for L_{den} values 45–75 dB

	% LA			% A			% HA		
L_{den}	Function	Lower	Upper	Function	Lower	Upper	Function	Lower	Upper
45	46.04	39.09	53.11	24.46	19.24	30.36	9.96	7.19	13.43
46	48.84	41.83	55.88	26.72	21.23	32.84	11.25	8.22	15.00
47	51.65	44.61	58.63	29.08	23.34	35.41	12.65	9.34	16.69
48	54.44	47.42	61.33	31.54	25.56	38.05	14.17	10.58	18.50
49	57.22	50.24	63.98	34.08	27.88	40.74	15.80	11.93	20.43
50	59.96	53.05	66.57	36.70	30.30	43.48	17.56	13.39	22.47
51	62.65	55.85	69.08	39.37	32.81	46.25	19.44	14.98	24.63
52	65.29	58.61	71.51	42.10	35.40	49.05	21.43	16.68	26.89
53	67.85	61.34	73.85	44.87	38.05	51.85	23.54	18.50	29.26
54	70.32	64.00	76.08	47.67	40.76	54.65	25.76	20.44	31.72
55	72.71	66.60	78.21	50.47	43.51	57.42	28.08	22.49	34.27
56	75.00	69.12	80.23	53.28	46.29	60.16	30.50	24.66	36.89
57	77.18	71.55	82.12	56.06	49.09	62.85	33.01	26.93	39.58
58	79.25	73.88	83.91	58.82	51.89	65.48	35.59	29.29	42.31
59	81.21	76.11	85.57	61.53	54.68	68.04	38.25	31.75	45.09
60	83.04	78.23	87.11	64.19	57.44	70.52	40.96	34.29	47.90
61	84.76	80.23	88.54	66.78	60.17	72.91	43.71	36.90	50.72
62	86.36	82.12	89.85	69.30	62.84	75.20	46.50	39.57	53.53
63	87.84	83.89	91.05	71.72	65.46	77.39	49.30	42.29	56.33
64	89.20	85.54	92.14	74.05	68.00	79.46	52.10	45.04	59.10
65	90.45	87.07	93.13	76.28	70.45	81.41	54.90	47.82	61.83
66	91.59	88.48	94.02	78.40	72.82	83.25	57.67	50.60	64.50
67	92.62	89.78	94.82	8040	75.08	84.97	60.40	53.38	67.11
68	93.56	90.97	95.53	82.29	77.24	86.57	63.09	56.14	69.63
69	94.40	92.05	96.16	84.06	79.29	88.04	65.71	58.87	72.07
70	95.15	93.03	96.71	85.70	81.22	89.40	68.26	61.55	74.41
71	95.82	93.92	97.20	87.23	83.04	90.65	70.72	64.18	76.64
72	96.41	94.71	97.63	88.64	84.74	91.78	73.09	66.75	78.76
73	96.93	95.42	98.00	89.94	86.32	92.81	75.36	69.23	80.77
74	97.39	96.05	98.32	91.13	87.78	93.74	77.53	71.63	82.66
75	97.78	96.61	98.60	92.20	89.13	94.57	79.58	73.93	84.42

Annex IV Exposure-response relationship between road traffic noise and ischaemic heart disease

$L_{day,16h}$ (dB)	L_{den} (dB)	OR
55	57	1
55.5	57.5	1
56	58	1
56.5	58.5	1
57	59	1
57.5	59.5	1.002
58	60	1.003
58.5	60.5	1.005
59	61	1.007
59.5	61.5	1.009
60	62	1.012
60.5	62.5	1.015
61	63	1.019
61.5	63.5	1.022
62	64	1.027
62.5	64.5	1.031
63	65	1.036
63.5	65.5	1.042
64	66	1.047
64.5	66.5	1.054
65	67	1.06
65.5	67.5	1.067
66	68	1.074
66.5	68.5	1.082
67	69	1.091
67.5	69.5	1.099
68	70	1.108
68.5	70.5	1.118
69	71	1.128
69.5	71.5	1.138
70	72	1.149
70.5	72.5	1.161
71	73	1.173
71.5	73.5	1.185
72	74	1.198
72.5	74.5	1.211
73	75	1.225
73.5	75.5	1.239
74	76	1.254
74.5	76.5	1.269
75	77	1.285
75.5	77.5	1.302
76	78	1.318
76.5	78.5	1.336
77	79	1.354
77.5	79.5	1.372
78	80	1.391
78.5	80.5	1.411
79	81	1.431
79.5	81.5	1.452
80	82	1.473

Annex V Indicators for noise

Under normal circumstances, sound varies over time. Even if we take the integration time of the ear as a lower threshold (around 0.1 s), we end up with 36.000 values per hour of sound if we measure only the intensity, and many more values if we assess also the intensity per frequency band. The need to reduce this number to a single value or a few values is fairly obvious.

There are countless ways to reduce measured values to a manageable quantity, so which is the best? Theoretically, the best way is to state a criteria and to assess which combination rule succeeds in ranking situations according to the established criteria. This was first done for hearing damage and the result was that a simple energy content evaluation sufficed to predict the amount of hearing damage one would suffer after many years of exposure. This resulted in the (A-weighted) equivalent noise level being used.

ISO then recommended this indicator in R-1996 in 1971 for use in environmental noise studies, and also set levels for day, evening and night. The EPA levels document in 1974 united these in the Day-night Level (DNL) which to this day is the indicator for aircraft noise in the USA.

Meanwhile a host of different indicators were developed and used, especially in the aircraft noise area. A number of penalties or correction factors may be added for tonality, impulse character, low frequency content and emergence, sometimes in combination.

The effectiveness of the resulting indicator has rarely been put to the test described above.

From recent research the following facts can be derived:

- Indicators are closer correlated than the correlation between indicators and effect. This makes it unlikely that the choice of indicator can be based on the performance alone.
- Theoretical considerations from measuring theory indicate that a noise indicator should be based on the sound power summation of elements. This would rule out all indicators based on time above, number above or percentiles, which by the way can be demonstrated by common sense.
- Night is a special period which merits its own indicator.

On the basis of this the EU overviewed the most important representatives of possible indicators, and in the end chose the L_{den} and the L_{night}. The criteria for the choices were:

- validity: relationship with effects.

What effects have be to taken into consideration is largely a political question. In most European countries noise regulations are mainly aimed at avoiding considerable annoyance, complaints and disturbance, as well as health effects. A large number of possible effects can be derived from the scientific literature. However, a quantitative relationship has been established for just a few of these: i.e. speech interference, annoyance, sleep disturbance (to some extent: for sleep related annoyance a relationship could be established, but the relationship with physical factors, like waking up, is still open to debate), and the risk of an increase in cardiac disease (weak). In recent times it has been frequently suggested in the scientific community that Lnight should be accompanied by an additional indicator that accounts for the maximum noise level and/or the number of events. This is an ongoing field of research.

- practical applicability:

ease of calculation from available data, or measurement using available equipment. Most importantly, it must offer the authorities a reliable basis on which to make decisions about noise reduction measures.

- transparency:

easy to explain, intuitive, as simple as possible, relationship with physical units, small number of indicators — preferably one.

- enforceability:

Use of the indicator in assessing changes or when the limit values in force are exceeded. One example is the use of a long term average. If the indicator is based on an annual average, a different approach is needed to demonstrate that a set limit has been exceeded than if an instant maximum level is used, which may never be exceeded.

- consistency:

as little change as possible from current practice. In view of the widespread use of indicators, it should be recommended only to change to indicators which belong to a totally different class, if they can be demonstrated to make significant improvements compared to the existing ones.

In conclusion no single indicator(-s) satisfy all of these requirements, as some may be incompatible with each other. If the most valid indicator is a complex, new, this does not satisfy the transparency and consistency requirements, for instance.

As it happens the choice for L_{den} satisfies most of these criteria.

The Frankfurt Airport study from 2006 is one of the few studies where a number of indicators are scrutinized. Figure V.1 in this study shows the correlation coefficients for annoyance for 20 indicators.

Cumulative indicators

The default method to sum noise levels is the simple energetic addition rule:

$L_c = 10 \lg (10 \exp((L_a / 10)) + 10 \exp(\wedge (L_b / 10)))$

For the 2 sound levels L_a and L_b resulting in level L_c.

For effects where no obvious source-dependent information is available, this method should be used to estimate effect from exposure to multiple sources. For the time being this is also the best approach for night time noise.

For a given situation which is exposed to more than one noise source from which the dose-effect relations are known, a method[xxxiii] is available to estimate the combined annoyance by adding the Lden values proportional to their exposure-effect relation.

The approach is relatively simple to carry out.

Road: $L_{r,m} = L_{den,road}$

Rail: $L_{r,r} = (2.10 * L_{den,rail} – 3.1) / 2.22$

Air: $L_{r,a} = (2.17 * L_{den,aircraft} + 15.6) / 2.22$
(Based on the END relationship)

Alternatively, $L_{r,a} = (2{,}05 * L_{den,aircraft} + 61) / 2{,}22$
(Based on the preliminary EU data)

Industry: $L_{r,i} = L_{den,industry} + 3$

Shunting yard: $L_{r,y} = (2.49 * L_{den,shunt} + 21.2) / 2.22$

Windturbine: $L_{r,wt} = (1.65 * L_{den,wt} + 41) / 2.22$

2. Calculate total $L_{den,r} = 10 * \lg (10 \exp^{(0.1 * (Lr,m + Lr,r + Lr,a + + Lr,i...)})$

3. To calculate percentages of annoyed or highly annoyed

If so desired percentages of annoyed(% A) or highly annoyed(% HA) can be calculated from the relationships for road traffic noise:

$\% A = 1{,}795 * 10^{-4} * (L_{den,r} – 37)^3 + 2{,}110 * 10^{-2} * (L_{den,r} – 37)^2 + 0{,}5353 * (L_{den,r} – 37)$

$\% HA = 9{,}868 * 10_{-4} * (L_{den,r} – 42)^3 – 1{,}436 * 10^{-2} * (L_{den,r} – 42)^2 + 0{,}5118 * (L_{den,r} – 42)$

Population indicators

Miedem[axxxi] demonstrated from measuring theory that any noise indicator should be based on a series of successive power sums:

General formula:

$y = [\sum_i (b_i x_i)^{ai}]^{1/a}$

For the application in the noise area the factor a = 1, so the L_{den} can be derived by the following steps in which only the weights bi are applied in each step:

1. Frequency bands: the A-weighting – > L_A: b_i = A-weighting for frequency band *i*

2. Events: contributions of a noise event – > L_{Ax}: b = 1

3. Events values into day, evening and night values : b = 1

4. Day, evening, night values into year average values b = 0 / 3.16 / 10. The factor 3.16 is the result from 10 exp $(^{(5/10)})$), in which 5 is the evening penalty of 5 dB

5. Year average day, evening and night values into L_{den} : b = 1

To this a further integration step can be added to obtain a population average:

6. L_{den} values into $L_{den,pop}$, b = 1

As all former steps follow the (weighted) power sum rule, this becomes:

$L_{den,pop}$ = 10 lg ($\sum$n.p.10 exp ($^{(Lden,i/10)}$))

Where n = number of dwellings and p = number of inhabitants per dwelling.

If the population indicator is to be calculated over different sources, it is advisable to substitute the Lden per source with the $L_{den,r}$ described under cumulative indicators.

The calculation of the DALY, can be seen as a special case of the population indicator. This not further worked out here.

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