# NSW Road Noise Policy Review – Initial Scoping Study

**A LITERATURE UPDATE FOR REVIEW OF THE NSW ROAD NOISE POLICY AGAINST CONTEMPORARY SCIENTIFIC STANDARDS AND SETTINGS**

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## Contents

# *LIST OF FIGURES*

# LIST OF TABLES

# INTRODUCTION

The brief for this work was that it should provide a scoping document of literature that would be able to be used by the NSW EPA, and others, to inform any future review of the 2011 NSW *Road Noise Policy* Department of Environment Climate Change and Water NSW (2011). It was to achieve this through identifying new scientific evidence that had not been available at the time of the original Road Noise Policy preparation. That policy had been constructed on the literature on road traffic noise, its effects, modelling, and management, that was available at the time of its preparation – circa 2010/2011. To this end, the current study is a desktop review of findings and practice that have since become available. This scoping review is based on literature published post 2010 for matters such as prediction, modelling, and management, or post 2014 for evidence of the effects of noise on people – though this protocol has not always been strictly adhered to as some relevant material from outside these periods has occasionally been included. The 2014 date was based on WHO publishing the *Environmental Noise Guidelines for the European Region* in 2018 (World Health Organization, 2018) based on systematic reviews of studies published between 2000 and 2014 – hence the date of 2014 to begin searches for new literature on noise effects.

This document reports the literature found in the above search periods, reviewing national and international material with respect to road traffic noise, its effects, and management. It examines a wide range of health effects associated with road traffic noise, not only annoyance, and looks to the evidence-base in these for noise metrics and thresholds for assessment of exposure to road traffic noise and warrants for noise mitigation. The review also canvasses, but only to the limited extent that the evidence allows, the generalizability of these noise indicators and criterion levels across different traffic conditions and across different contexts in which traffic noise is generated and experienced.

It also examines developments in road traffic noise prediction models and mapping during the review period, and current practice regarding road traffic noise management – the latter primarily as published in the peer-reviewed scientific and engineering literature and, in some cases, in conference papers and in government and agency reports. Some best-practice management, however, may only be able to be accessed through approaches to leading environment, transport, and planning authorities in other countries but, in general, making such approaches through direct contact was not within the scope of the present study.

No recommendations are made in this document. Its purpose is solely to provide a survey of the evidence-base of new scientific information and analysis that may need to be considered. It is thus a collection of prominent studies, results, quotations[1], illustrations, and observations from a very broad literature. It generally avoids making conclusions or summaries within each section in order to not pre-empt a reader's own judgements regarding the salience of particular information. Its presentation is not as an academic paper, being designed for a target audience of acoustic, transport, environmental, health and planning professionals, and decision-makers. Also in mind, regarding the material selected and presented, was identification of scientific information that might be useful in interactions between the above professionals and the wider community.

# METHODOLOGY

The current study required extensive searches of relevant literatures over the review periods. Given the emphasis of this work on application of recent, but established, international evidence and experience to policy development, the search methodology focussed heavily on review papers rather than on the large set of original research papers

[1] Throughout this document, nearly all use of text from other sources is enclosed in quotation marks and attributed. However, there are circumstances where some paraphrasing of the original was appropriate, particularly in figure and table captions such as where the original included multiple sources of transport noise from which this author extracted the road traffic noise information. As these are no longer direct quotations, they have not been enclosed in quotation marks. However, this author acknowledges that most of the figure and table captions in this report, and related conclusions, are text copied or paraphrased from the attributed original, in whole or in part, with modifications generally identified by 'adapted from …' or 'extracted from …". Selective verbatim use of the original material from the attributed sources, as far as possible within both the confines of this report and minimizing complexity for the reader, was considered important to best convey the evidence-base.

on which reviews are based. Review papers utilized tended to be those in which authors had applied a comprehensive and systematic search of the relevant data bases and who had usually made explicit the search strategy they had adopted. Reviews sometimes included meta-analyses of the results of the original studies. Some more recent studies not yet included in the systematic reviews have additionally been reported in this document where inclusion has been considered useful.

For many of the topics mentioned in the introduction, an existing systematic review conducted within the review period was able to be identified. For example, for health effects and annoyance, the systematic reviews prepared for the revision of the WHO environmental noise guidelines (World Health Organization, 2018) were often the starting points. Some of these reviews had subsequently been revisited, with reanalysis often by the same, or at least overlapping, sets of authors. Reanalyses often incorporated the original studies and added additional studies reported since the initial reviews were completed. Starting with these systematic reviews, this author then searched forward, locating research articles and commentaries that referenced the review paper. In cases where this process located a subsequent review, the search extended to papers that had referenced it in turn. Given the significant number of effects of noise and other topics required to be examined, this was an effective strategy - providing access to not only the best quality syntheses of research, but also to subsequent commentaries on these by both specialists and practitioners. The major foci for this scoping study were peer-refereed papers in international journals rather than conference papers - the latter can be valuable in terms of observations on practice and commentaries on other work, but they do not represent a large proportion of the documents used in preparing this narrative. Government and agency reports were also accessed where appropriate.  Studies that had not been published in English generally were not included within this scoping review. While it followed the methodology described above, this scoping document should not be referred to as a 'systematic review'.

This document does not report the results of the literatures searches *per se*. Instead, key papers identified by the searches were read by this author and used to prepare this narrative review. Not all topics covered in this report were subject to the same depth of literature searching – there were simply too many given available resources. The more comprehensive searches were reserved for the topics identified in the brief for this study. For some other matters (electric vehicles and noise, for example) the search was regarded as sufficient when a recent informative review of the topic had been located - and summarized, or at least recorded, here.

All the papers identified through the searches are included in the document reference list and most, some 80%, have specifically been referred to within the text. Where the material appeared potentially apposite to future review of the Road Noise Policy, quotations, researchers' observations and conclusions, and illustrations, from the original papers have been included where feasible.  The narrative has been constructed primarily around the series of topics/issues specified in the original brief for this work, with a few others being identified in the course of this study.

# OVERVIEWS: ROAD TRAFFIC NOISE POLICY & MANAGEMENT

In most countries, road traffic noise policies tend to be managed at the national, or sub-national levels of government, though there are some driving supranational policies – mostly in Europe.  The first section of this document draws from existing overviews of road traffic noise policies from Australia, North America and Europe. The most recent update internationally is contained in the review of Perna et al. (2021) below.

## OVERVIEW: Australia

There have been two useful overviews of Australian road traffic noise policy since 2010.

Burgess and Macpherson (2016) noted the division of responsibilities for noise from road traffic was across three levels of government. They summarised the similarities, and the many differences, in metrics[2], limit levels and

[2] The terms *metric* and *indicator*, also *scale* and *noise descriptor*, appear to be used somewhat interchangeably in the environmental noise literature to describe a dimension/measurement of the acoustic environment for use in noise management. This review tends towards *metric*, but *indicator* is used extensively in European Commission papers (European

application, in the States and Territories. The metrics used in these jurisdictions tended to be either $L_{A10,18h}$ or $L_{Aeq,T}$ – the latter for each of the day period and the night period. All are levels at one metre in front of the façade a of noise-sensitive premises (thus requiring a façade adjustment of +2.5 dB to be made to the noise level assessed in a free field to account for façade reflections). These policies are intended primarily for the design guidance of major roads and to control the planning of new noise-sensitive areas adjacent to major road corridors. The authors noted: (a) a general absence of enunciation of policies by local governments for road traffic noise controls, despite a significant proportion of noise-affected dwellings in Australia lying adjacent to roads controlled by local governments, and (b) that most States, with the possible exception of NSW and WA, did not explain the relationship between their chosen criteria and scientific evidence available on the non-auditory effects of road traffic noise. Burgess and Macpherson (2016) argued that future policy reviews would need to ensure alignment of criteria with the current state of knowledge on the effects of noise, and that more knowledge of cost-effectiveness of noise reduction methods will likely have to figure in future noise mitigation policy.

An earlier discussion paper (Vic Roads, 2015), *Traffic Noise Reduction Policy Review*, had a purpose similar to that of the current review for the NSW EPA. The Victorian government's *Traffic Noise Reduction Policy* had been written in 1989 with only minor changes in 1997 and 2005 and needed to be brought up to date – given increased traffic flows on Victorian roadways, increased noise at night, increased truck flows, and the accumulating international evidence regarding significant health effects of traffic noise exposure. Policy and technical issues that exercised the authors of the VicRoads discussion paper included:

a) Noise from individual noisy vehicles
b) Planning and building controls – including reverse sensitivity (new development built within areas of high noise near existing infrastructure)
c) Different criteria for new as against existing roadways (they reported expressions of concern by the public that it seemed absurd to have residents along different sections of the same highway potentially subject to different criteria – given the only difference was in the build date of those sections of roadway). On this topic, also see the section '*Different Responses/Limits for Different Categories of Roadway*' re useful wording in the WA Planning Policy
d) New roads in quiet areas and criteria based on increased, not on limit, levels
e) Aspirational versus mandatory limits, and the concepts of 'principle-based' policy: viz. do as well as one can in noise reduction; better than criterion if possible; but if limit values are exceeded then manage by other mechanisms
f) Relative costs of noise mitigation and various (health, property value, perceived) costs of traffic noise exposure.

While all these issues were canvassed in the VicRoads Discussion Paper, they remained largely unresolved. Several are identical to matters raised in the brief for this study for NSW EPA, specifically (a) and (d), and these are examined later in this document.

## OVERVIEW: North America

Overall, there has been limited research and policy action in the environmental noise field in the USA, including road traffic noise, since 1981 (Beranek & Lang, 2003; Finegold & Finegold, 2003). It was in 1981 that President Reagan removed all funding from the Office of Noise Abatement and Control of the Federal Environmental Protection Agency (Shapiro, 1992) – even though the Noise Control Act of 1972 remained in force with zero funding support. It is primarily in the field of aircraft noise that there does appear to have been some renewed activity over recent years in research, management, and policy development. There appears also to be little innovative work in road traffic policy and abatement at the State level in the USA. INCE-USA has recently announced a new website *A Quieter America* (INCE-USA, 2022), but, while clearly an effort to bolster noise management in the USA, the web site, at this stage, makes only limited mention of noise from transport sources.

There are a few publications which do show some minimal continuing interest in road traffic noise research and management in the USA. Because this body of work is quite limited, the opportunity is taken to summarise it here

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Commission, 2000) and elsewhere. Different terms, particularly *metric* and *indicator*, are used interchangeably in this review in order to mirror those terms used by cited authors.

within this *Overview* section - except for Rochat et al. (2022), a National Cooperative Highway Research Program (NCHRP) document. which is reported separately below in the sub-section ‘*Barriers & Vegetation’* under ‘*Literature on Innovative Traffic Noise Mitigation Practice/Studies’*.

Hammer et al. (2014) rehearse the failure of the USA to address its known noise problems, and provide commentary on the scientific and policy aspects of noise exposure, including road traffic noise. They call for, *inter alia,* updating of 1981 U.S. EPA national-level estimates of individual noise exposure, and action at each of federal (noise at source), state (altering the built environment) and local (buildings) governments, to reduce levels. Other relevant NCHRP work included National Academies of Sciences (2014) that provided supplemental guidance on the application of FHWA’s Traffic Noise Model (TNM). A paper from Canadian authors, Sun et al. (2014) compared the implementation of the TNM model to predictions of the exposures of the first row of buildings on road projects. They found both softwares offered convenient interfaces, and that SoundPLAN predictions more closely corresponded to TNM results than did Cadna/A. The Federal Transit Administration (Hanson et al., 2006) updated an earlier guidance manual, including road traffic noise, that covered assesing noise and vibration impacts, but nearly all of the reported material related to investigations undertaken several decades earlier. There was also work by Casey et al. (2017) who used a geospatial land-use-regression type model of sound levels (including base measurements from 492 urban and rural sites) across the contiguous USA to determine that there was evidence of inequality in exposures (to all sources of noise, including road traffic) at the U.S. census block level. There was higher noise exposures in groups characterized by lower SES and higher proportions of American Indian, Asian, black, and Hispanic residents. They suggested that such inequalities in noise exposure may have implications for more fully understanding drivers of environmental health disparities in the United States.

While not restricted to road traffic noise alone, the U.S. Department of Transportation Bureau of Transport Statistics has reported the first biennial update of a national multimodal transportation noise mapping initiative using data sources from the Federal Highway Administration (FHWA), the Federal Railroad Administration (FRA), and the Federal Aviation Administration (FAA) to create a comprehensive map of noise levels. The data visualized on the map (see https://maps.dot.gov/BTS/NationalTransportationNoiseMap/) is not intended for assessing impact and its stated purpose is for tracking trends. The site allows comparisons of 2016-2018 years. The user choses whether to show noise from all three modes simultaneously (as in the figure below) or noise from road traffic or from aircraft alone. A recent news release (U.S. Department of Transportation Bureau of Transport Statistics, 2020) makes the (not particularly useful) observation that, nationally in the U.S.: ‘Under a quarter of population is exposed to office-type transportation noise’, and provides an extract from the map as reproduced here as Figure 1. Exactly how such extensive mapping could be utilized in noise management, other than as a general communication tool with the public and other professionals, is unclear.

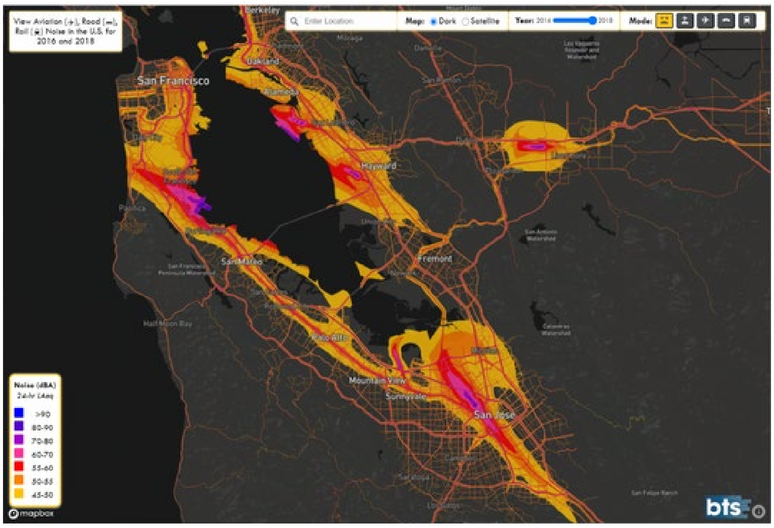


**Figure 1 Reproduction of map of aviation, highway and rail noise in the San Francisco Bay Area, from the interactive web application (U.S. Department of Transportation Bureau of Transport Statistics, 2020).**

## OVERVIEW: Europe

By contrast, in Europe, the Environmental Noise Directive (EU, 2002), generally referred to using its acronym, the END, has been a major and continuing driver for research, policy development, and noise management over two decades This included the UK, until the latter left the European Union in January 2020[3]. The END sets legally binding obligations for the measurement, reporting, reduction, and management of environmental noise. As Murphy and King (2022b) note, in their international book on *Environmental noise pollution: Noise mapping, public health, and policy*:

> *'Focus is placed on the EU precisely because it is the world leader in environmental noise policy and related legislation. Relevant noise mapping studies and associated research in other jurisdictions are also presented* [in this chapter] *to demonstrate the wide adoption of the EU approach to assessment and mitigation of environmental noise across the globe'.*

The END requires (with respect to road traffic noise):

- road traffic noise maps of all roads carrying greater than 3million vehicle passages per year, and for all urban areas of greater than 100,000 population. Mapping was to be updated every 5 years (2007, 2012, 2017, 2022…) with the third round intended to be finalised by the end 2017 – but this has apparently not been achieved in full
- estimation of populations exposed to these levels, in 5dB bands
- adoption of Action Plans to manage noise, based upon the noise mapping results. These were to be finalised by the beginning 2019 – but, as of early 2019, they were far from complete across Europe. Overall implementation of the action planning process appears to have been poor.

The current situation and approach to road traffic noise management in Europe has also recently been summarized by the European Environment Agency (2020). This included: an updated assessment of population exposed to high levels of noise; an assessment of health impact based on the WHO Environmental Noise Guidelines for Europe (there are amendment of Annex III of the END regarding assessment methods for harmful effects of environmental noise (EU, 2020a) that formally implement in the END the recommendations from the WHO Environmental Noise Guidelines (World Health Organization, 2018)); and reviews actions to reduce noise through action plans[4]. Conclusions with respect to the current situation of road traffic noise in Europe include (extracted from the original 100-page report that contains a lot more detail than conveyed in the summary below):

- the number of people exposed to high levels of road traffic noise remains high and is likely to increase in the future
- at least 20 % of the EU population live in areas where traffic noise levels are harmful to health (and exposure is likely to be underestimated)
- an estimated 113 million people are affected by long-term day-evening-night traffic noise levels of at least 55 dB(A)
- managing and reducing noise through land use and urban planning represents a very small percentage of the measures chosen to address noise – most are source or propagation path controls
- cost-benefit estimations for mitigation measures can be more favourable if the positive impacts of addressing both air quality and noise are taken into account. This calls for effective coordination between communities of policymakers and stakeholders working to address noise and air pollution.

This author suggests that it would be reasonable to assume that similar conclusions (with numbers above reduced in proportion to Australia's population) would arise if a comparable detailed accounting were able to be conducted of the Australian road traffic noise situation. Outside of the END, Europe also requires specific source controls related

[3] The UK had transposed Directive 2015/996/EC (the END ) into national legislation in 2018: http://www.legislation.gov.uk/uksi/2018/1089/made/data.html It appears (subject to locating any information to the contrary) that this UK national legislation, and hence the effective requirements of the END, is still in force there.

[4] There is some further consideration of noise management strategies discussed in European Environment Agency (2020) later in this document – see Section '*Interventions & Change Effects*'.

to noise from road traffic vehicles. An examination of the history of test requirements, and limits for individual motor vehicles is not within the scope of this review, for Europe (or for Australia[5]).

The European focus on environmental noise through the END has also led to funding for a significant number of consortia research projects and advanced training activities. European-funded projects are known by their acronyms, and these acronyms appear often in journal articles, conference papers and reports, and some familiarity with them can be useful. The quality of findings and documented output from different projects can be quite variable – given that they all are reported through a set of deliverables that had to be identified at the proposal application stage (prior to the commencement of any work). A brief introduction to some of these projects relevant to this review is provided in a last section of this document, *'Recent & Current International/ National Projects Relevant to this Brief'*.

## COMPARATIVE ANALYSES OF LIMIT VALUES FOR ROAD TRAFFIC NOISE

Numerical values of noise metrics that trigger management of traffic noise are described by various terms. This report uses the generic '*limit value'*. Other terms such as *threshold*, *criterion*, target or *guideline value*, are also used, and where such distinctions are important, they will be noted[6]. The noise metrics used for road traffic noise, and the limit value, differ across jurisdictions including across Australian States. While a detailed examination of limit values used nationally and internationally was not a specific requirement of the current study, a comparative analysis of those used in Australia, and elsewhere, is informative. The comparative analyses here is focussed on the levels of the limits after transforming them to a common noise metric – and is not an examination of the merits of alternative metrics themselves (In fact, most metrics are so highly intercorrelated, there is little reason to look for distinctions between them).

Comparative analysis of limits values is useful for benchmarking across jurisdictions. But it is recognized that it would be preferable if jurisdictions had articulated how they determined limit values based on quantitative relationships between road traffic noise exposure and its effects. Rarely is sufficient background information provided regarding the processes adopted, and justification, for the choice of a particular metric and its limit value[7]. Derivation of limit values from evidence-based exposure-response functions[8] for traffic noise effects is examined in more detail later in this report.

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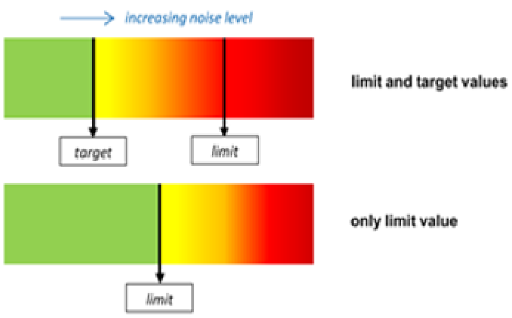
[5] Australia has tended in the past to adopt international testing requirements and limits – but the state of play with regard to these in Australia is not immediately clear to this author, and Australia appears not to have implemented the last rounds of test changes, nor the European tightening of vehicle noise limit requirements.

[6]

Based on a survey across different noise control jurisdictions in Europe, Peeters and Nusselder (2019) provide a useful schema to interpretate sometimes confusing terminologies regarding target and limit values. In the diagrams on the right, increasing intensity of yellow to red in the panels represents increasing intensity of potential management response to exceeding a limit or target. Some jurisdictions may specify only *limit* values, with no action (green) required below that limit. Other jurisdictions may specify both target and limit, with no action required below the target value.




[7] One example of justification is in Department of Environment Climate Change and Water NSW (2011) where it is noted (p31 of that document) that: '*55 dB(A) LAeq corresponds to approximately 10% of residents who were highly annoyed, and 60 dB(A) LAeq corresponds to approximately 18% of residents who were highly annoyed…Based on research findings, environmental objectives for transportation-related noise sources are set approximately at the point at which 10% of residents are highly annoyed by the noise*'. Reference is made to a range of exposure-annoyance relationships from the 1970s and 1980s (Figure 2 of that document) but there is no detailed examination of the derivation of, and scientific uncertainties associated with, those. These matters are explored further in a later section of this report that discusses evidence-based exposure-response relationships.

[8] The term *exposure-response function ERF (or exposure-response relationship ERR)* is used in this report – but *dose(age)-response*, *exposure-effect, and dose(age)-effect* are also used by various authors.

**Comparative across Jurisdictions**: Metrics and levels used in different jurisdictions can be compared to each other. While there are difficulties and limitations in rigorously making these comparisons, several have been attempted across subsets of jurisdictions. Comparisons per se are primarily of administrative rather than scientific value, but their usefulness increases if the limit values in a jurisdiction are able to be compared to some benchmark such as the guideline values recommended for road traffic noise by the World Health Organization (2018) *Environmental Noise Guidelines for Europe*.

Vic Roads (2015) summarized, as did Burgess and Macpherson (2016), the limit values applied in all Australian States and Territories, for new, upgrade and existing roadways both in terms of the metrics of noise and their dB levels. Two further comparative analyses of road traffic noise policies and limit values were also published within the review period, one across Europe (Peeters & Nusselder, 2019) and one internationally (Perna et al., 2021).

Comparisons of limit values across jurisdictions is difficult because of different noise metrics, different time periods of aggregation for the same metric, different ways in which limit values are applied, and different requirements as to where levels are to be assessed or measured. Quasi-rigorous comparisons require first that the contexts of the comparison are identical, then transforming between pairs of metrics of interest, preferably with known levels of error associated with each transformation. Some transformations with known levels of uncertainty were available based on empirical studies of traffic noise data sets from the UK, Victoria, Queensland and Europe - mostly adjacent to limited-access roadways with relatively high traffic volumes (Abbott & Nelson, 2002; Austroads, 2005; Naish et al., 2011; Brink et al., 2018). In some cases, several algorithms for transformation between the same pair of metrics existed. These were used to examine consistency in the outcome levels of the transformed limits, given that the different algorithms had been derived from different data sets. This gives increased confidence in the transformed limit values. The European comparative analyses reported below (Peeters & Nusselder, 2019) regrettably did not attempt any transformations

## WITHIN-AUSTRALIA COMPARISONS (Tables 1 & 2)

Burgess and Macpherson (2016) and Vic Roads (2015) provide information about the application of different limits, and various caveats in their use, across Australian States and Territories. Their tabulations encouraged some interstate comparisons of limit levels (largely as $L_{10,18h}$) but they warn that simplistic comparisons should be used cautiously and require reference back to the exact wording of the policies in each jurisdiction. Different limit values are applicable to new, to existing, or to upgraded roads, increasing the complexity of comparison.

An uncomplicated, but still legitimate, comparison of limit values across Australian States can be achieved by considering the application of limit values **to *new roadways* alone**[9]. Table 1 below shows the day and night *new road* limit values extracted from Burgess and Macpherson (2016) and Vic Roads (2015) (the same values are also reported in the 2011 NSW Road Noise Policy). These are transformed to the equivalent $L_{den}$ and $L_{night}$, as defined in Europe – which allow comparison to the WHO Environmental Noise Guidelines for Europe (World Health Organization, 2018).

The fourth column of Table 1 lists the studies from which the algorithms to transform between each required pair of noise metrics ($L_{1018h}$ to $L_{den}$ and $L_{night}$; $L_{day}$ to $L_{den}$) was sourced. Time periods of both original and output metrics also needed to be identical, but as this was rarely the case, had to be closely matched in a way which did not introduce significant errors. Exact transformations are shaded yellow in the table. Changing the evening/night transformation from 10 pm to 11 pm is likely to have introduced minimal error, and these are shown in blue. One transformation, shaded green, used periods one hour earlier than the European default hours for $L_{den}$, with unknown, but likely small, effect on the error. The 95% confidence interval for each transformed limit value is shown, each sourced from its original study, where available.

9 Adopting different limit values for roadways in different development stages (new, existing or upgraded roadways), or of other different classifications, is a policy/economic choice to which the scientific evidence in this literature review can contribute little or no information. This is discussed further in the section below: '*Different Responses/Limits for Different Categories of Roadways'*

**Table 1 Intercomparisons of Limit Values for Road Traffic Noise for *new roadways* across Australian States/Territories (Vic Roads, 2015; Burgess & Macpherson, 2016), together with WHO guidelines. Algorithms used for the transformations are listed in Table 2 below, with colour codes as to whether transformation is exact, or inexact, in term of measurement periods for the metrics.**

| States Territories | Limit values *new* roadways 1m from façade | | Alternative sources for transformation formulae for noise metrics | Limit values transformed to $L_{den}$ day: 7am-7pm even:7pm-11pm (+5dB) night: 11pm-7am (+10dB) [1] free-field | Limit values transformed to $L_{night}$ night: 11pm-7am [1] free-field |
|---|---|---|---|---|---|
| | Day | Night | | | |
| ACT/NT/ Qld/Vic/ Tas | 63 $L_{A10,18h}$ | No limit specified | Abbott and Nelson (2002) non-motorway | 60 +/-1.6 # | 51 +/- 2.9 ## |
| | | | Abbott and Nelson (2002) motorway | 64 +/- 1.7 * | 57 +/- 2.5 ** |
| | | | Naish et al. (2011) | 62 +/- 3.3 ^ | 53 +/- 9.6 ^^ |
| | | | Austroads (2005) | 63 CI n.a @ | 55 CI n.a @@ |
| NSW/SA WA | 55 $L_{Aeq}$ 7am-10pm (15h) | 50 $L_{Aeq}$ 10pm-7am (9h) | Austroads (2005) | 57 CI n.a & | 48 && |
| | | | Brink et al. (2018) | 55 +/-1.6 ~ | 48 && |
| | (WA) 6am-10pm (16h) | (WA) 10pm-6am (8h) | Brink et al. (2018) | 55 +/-1.6 + | 49 +/- 1.2 ++ |
| **WHO guidelines** | | | | **53** | **45** |

[1] In Europe (EU, 2002), States may alter the periods of day/evening/night by 1 or 2 hours, and alter period start/end times, but default values are day: 7am to 7pm; evening: 7pm to 11pm, and night: 11pm to 7am .

Correct periods used in both source and output
Evening-to-night occurs at 10 pm in source, 11pm in output
*Periods in source metrics one hour earlier than in output*

**Table 2 Transformation algorithms applied to State/Territory Limit Values of Table 1, with columns showing the transformed level, and 95% confidence interval of the transformation.**

| | **Façade correction of -2.5dB has been applied to all transformed levels below** | | 95% CI dB |
|---|---|---|---|
| # | $L_{den} = 0.9241\, L_{A10,18h} + 4.1982$ | 59.9 | +/-1.6 |
| ## | $L_{night\,(11pm\text{-}7am)} = 0.9044\, L_{A10,18h} - 3.7683$ | 50.7 | +/- 2.9 |
| * | $L_{den} = 0.8963\, L_{A10,18h} + 9.6917$ | 63.7 | +/- 1.7 |
| ** | $L_{night\,(11pm\text{-}7am)} = 0.8691\, L_{A10,18h} + 4.239$ | 56.5 | +/- 2.5 |
| ^ | $L_{den} = 0.88\, L_{A10,18h} + 9.3$ | 62.2 | +/- 3.3 |
| ^^ | $L_{night\,(10pm\text{-}6am)} = 0.91\, L_{A10,18h} - 1.5$ | 53.3 | +/- 9.6 |
| @ | $L_{den} = L_{A10,18h} + 2.2$ | 62.7 | not available |
| @@ | $L_{A10,18h} = L_{Aeq,9h} + 5.1$ | 55.4 | not available |
| & | $L_{den} = L_{Aeq,15h} + 4.4$ | 56.9 | not available |
| && | Facade correction only | 47.5 | |
| ~ | $L_{den} = L_{Aeq,15h} + 2.1$ | 54.6 | +/-1.6 |
| + | $L_{den} = L_{Aeq,16h} + 2.0$ | 54.5 | +/-1.6 |
| ++ | $L_{night\,8h} = L_{nightb\,9h} - 1.1$ | 48.6 | +/-1.2 |

Tables 1 and 2 show that the NSW/SA/WA limits are equivalent to an $L_{den}$ some 2 to 4 dB above the WHO $L_{den}$ road traffic noise guideline of 53 dB, and an $L_{night}$ some 3 to 4dB above the WHO $L_{night}$ road traffic noise guideline of 45 dB. For those jurisdictions continuing to use $L_{A10,18h}$ as limit values, current limits are equivalent to an $L_{den}$ some 7-11dB above the WHO $L_{den}$ road traffic noise guideline of 53dB, and an $L_{night}$ some 6-12 dB above the WHO $L_{night}$ road traffic noise guideline of 45dB. The reasonable consistency in the output limit values based on formula from different authors and road traffic noise data sets, together with the generally small errors, provides reasonable confidence in these comparisons

## WITHIN-EUROPE COMPARISON (Figure 2)

Within the review period, Peeters and Nusselder (2019) reported an overview of noise limits for road traffic (and other noise sources) across countries in the European region. Their study was implemented by the EPA Network Interest Group on Noise Abatement and was designed to examine the relationship between limit values/guidance adopted for road traffic noise management, and the guideline values in World Health Organization (2018). Information was gathered by questionnaire, with 27 countries providing: the indicators (noise metrics) used; the limit values; the scope to which they apply; and the consequences of limit exceedance. The authors append fact sheets for each European country – and their published document thus provide a convenient resource regarding noise management practice in each country surveyed.

For road traffic noise, Peeters and Nusselder (2019) note that there is common use of separate limits for $L_{day}$ and $L_{night}$, with the day period sometime also including the evening period, e.g. 06:00 – 22:00 or 07:00 – 23:00. In addition to $L_{day}$ and $L_{night}$, separate $L_{evening}$ limits may exist for the evening (e.g., 18:00 – 22:00 or 19:00 – 23:00) with the day period shortened. The use of an $L_{den}$ limit, or a combination of $L_{den}$, $L_{night}$ and possibly $L_{evening}$, is less common.

Their approach to comparing limit values in the different jurisdictions was limited as they chose to ignore real differences between different metrics. Their road traffic noise results have been extracted to Figure 2 below.

Figure 2 shows that there is a wide range, some 18 to 20 dB, in limit values for both $L_{day}$ and $L_{night}$ road traffic noise across Europe[10]. Less than 10% of countries have limit values for $L_{day}$ that conform to the WHO road traffic noise guideline, less than 30% for $L_{night}$. While this does highlight that many European countries do not (as at the date of the survey) adhere to the WHO guidelines, more detailed examination of these results is not appropriate, given that the authors did not attempt transformations between the different metrics - even though they recognized that empirically derived transformations of acceptable accuracy were available for Europe (Brink et al., 2018). They do speculate that if the $L_{day}$ limit values included in Figure 2 (a) would be converted to equivalent $L_{den}$ limits, this would shift the cumulative curve to the right by 1 - 2 dB.

While in the process of examining European limit values, it should be noted he EU END (EU, 2002) does not specify limit values. These have been left up to the individual countries as described in Peeters and Nusselder (2019). The END does specify ranges of values - *viz.* $L_{den}$ in 5dB intervals above 55 dB, and $L_{night}$ in 5dB intervals above 50 dB – but these are in no way intended to represent *limit* values. They are merely a specification as to how each country's noise exposure is to be reported to the European Commission.

[10] The (Peeters & Nusselder, 2019) questionnaire asked countries the rationale behind their limit values:

- 9 countries referred to the (1999) WHO guidelines and/or the Miedema exposure-response function used by the WHO. Several countries indicated that the value was chosen at a particular annoyance level, between 9% and 15% highly annoyed people.
- 2 countries referred to the END as a basis for their limit values without further specification.
- 7 countries reported that the limits were set on national/local exposure-response relationships/consultations.
- 5 respondents for countries with defined noise limits did not answer,

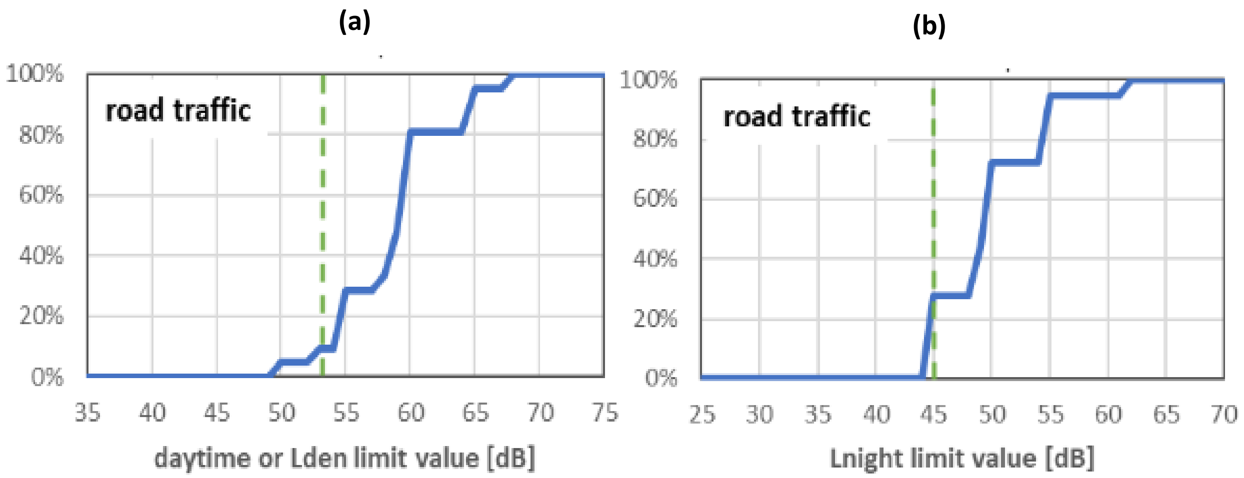



**Figure 2 Comparison of limit values for road traffic noise in different European countries, with cumulative percentage of countries exceeding any limit on the vertical axes. The authors (Peeters & Nusselder, 2019) surveyed countries by written questionnaire. The results have limitations, as they chose to ignore real differences between different metrics. (a) is day-time values: this group includes both $L_{day}$ (day defined with different periods, some including the evening) and $L_{den}$ – without differentiation (b) is night-time values including mainly $L_{night}$ – with some variations in definitions, again with no differentiation between these in the reporting.**

Regarding UK 'limit values' for road traffic, the followings information is from a library research note provided to Members of the British Parliament[11] during the timeframe for this review (2010).

> *There are no relevant formal noise limit values in force in England for environmental noise levels from major roads. However, the Noise Insulation Regulations 1975 (as amended in 1988) under the Land Compensation Act, define a threshold level as part of the eligibility criteria. When a new road is built a calculation is made of future noise levels. The highway authority then offers those eligible help with insulation. It may also install sound barriers…The rules state that…a dwelling within 300 metres of road works would be eligible…if it is calculated that…the traffic noise level at one or more facades will increase by at least 1dB(A) and will be not less than the specified level of 68 dB(A) L10 (18 hour). There are also guideline levels to be found in Planning Policy 7 Guidance 24 that provides guidance on land use with respect to noise from road traffic.*

While nothing in this paragraph is new, it is a useful clarification of the current UK approach to limit values. The same approach should also be recognized as the basis of much of the early road traffic noise policy and limits across many Australian States[12].

## INTERNATIONAL COMPARISONS (Figure 3)

**Australia, North America, and Europe**. Within the review period, Perna et al. (2021) undertook a more extensive and rigorous comparison of road noise policies – extending previous comparisons by including traffic noise policies from both sides of the Atlantic and from Australia. They reviewed administrative responsibilities for both road traffic noise emissions and exposures, described outdoor measurement protocols, and compared noise limit values across the jurisdictions. The focus here is on the outdoor limit values for road traffic noise at residential premises that they reported across the countries, including the USA[13].

[11] https://researchbriefings.files.parliament.uk/documents/SN00347/SN00347.pdf

[12] Viz. the original use of $L_{A10,18h}$ as the preferred metric; an original limit value of 68 dB $L_{10,18h}$; and the utilisation of CORTN for prediction to assess of whether a dwelling's exposure required roadside barriers, or house insulation, to be installed.

[13] FHWA limits (Noise Abatement Criteria) in the USA, (Table 1 to Part 772 FHWA., 2010) were:

A range of metrics and periods of measurement had been used in the limit values across the jurisdictions. To be able to compare the values, Perna et al. (2021) used empirical relationships between different metrics, that had been reported by Brink et al. (2018), to convert each of the country's metrics to a common $L_{den}$ and $L_{night}$ (they used the symbol $L_n$ for $L_{night}$). The empirical relationships had been derived from analysis of diurnal traffic patterns across several European countries, and calculated noise exposures of a stratified sample of the population of buildings in Switzerland. These allowed transformation across a range of noise metrics, including some which differed only in the period of the day utilised. They also provided estimates of the uncertainty arising from each conversion.

Figure 3 extracts from Perna et al. (2021) a plot of the noise limits for road traffic noise outside of residential dwellings, as $L_{den}$ and as $L_{night}$, for all the jurisdictions studied, and compares them to the WHO guidelines of 53dB and 45dB. Australian limit values (New South Wales, South Australia, and Queensland) are not included in this plot because the periods used in these states did not match exactly the periods used in transformation algorithms. The Western Australian limit values did match the periods exactly (highlighted yellow in Table 1) but the authors advise (pers. comm.) that they reported them in their paper as a 'sensitive' rather than a 'residential' zone, and the WA limit values have been added to Figure 3 as blue stars – $L_{den}$ 55dB, $L_{night}$ 49dB. Table 1 shows that, with the evening/night transformation at 10pm rather than 11pm (blue highlight) NSW limit values would be only slightly different to those of WA ($L_{den}$ 55-57dB, $L_{night}$ 48dB).

In summary, this examination of comparative studies of road traffic noise limit values has been able to position the limit values adopted in the Road Traffic Noise Policy of the Department of Environment Climate Change and Water NSW (2011). While both $L_{den}$ and $L_{night}$ levels in the policy are several decibels higher than the WHO guideline values, the detailed comparisons demonstrate that both day and night limit values are closer to the levels recommended by WHO than are those of most jurisdictions in Europe and North America.

This is reported here as a statement of fact, not as a statement suggesting that NSW limits are either too low or too high. In also does not mean that limit values in some overseas jurisdictions are not changing (or perhaps already have changed) since the data was collected.

| Activity category | Activity Leq(h) | Criteria [2] L10(h) | Evaluation location | Activity description |
|---|---|---|---|---|
| A | 57 | 60 | Exterior | Lands on which serenity and quiet are of extraordinary significance and serve an important public need and where the preservation of those qualities is essential if the area is to continue to serve its intended purpose. |
| B [3] | 67 | 70 | Exterior | Residential. |
| C [3] | 67 | 70 | Exterior | Active sport areas, amphitheaters, auditoriums, campgrounds, cemeteries, day care centers, hospitals, libraries, medical facilities, parks, picnic areas, places of worship, playgrounds, public meeting rooms, public or nonprofit institutional structures, radio studios, recording studios, recreation areas, Section 4(f) sites, schools, television studios, trails, and trail crossings. |
| D | 52 | 55 | Interior | Auditoriums, day care centers, hospitals, libraries, medical facilities, places of worship, public meeting rooms, public or nonprofit institutional structures, radio studios, recording studios, schools, and television studios. |
| E [3] | 72 | 75 | Exterior | Hotels, motels, offices, restaurants/bars, and other developed lands, properties or activities not included in A-D or F. |
| F | | | | Agriculture, airports, bus yards, emergency services, industrial, logging, maintenance facilities, manufacturing, mining, rail yards, retail facilities, shipyards, utilities (water resources, water treatment, electrical), and warehousing. |
| G | | | | Undeveloped lands that are not permitted. |

*[1] Either Leq(h) or L10(h) (but not both) may be used on a project.*

*[2] The Leq(h) and L10(h) Activity Criteria values are for impact determination only, and are not design standards for noise abatement measures.*

*[3] Includes undeveloped lands permitted for this activity category.*

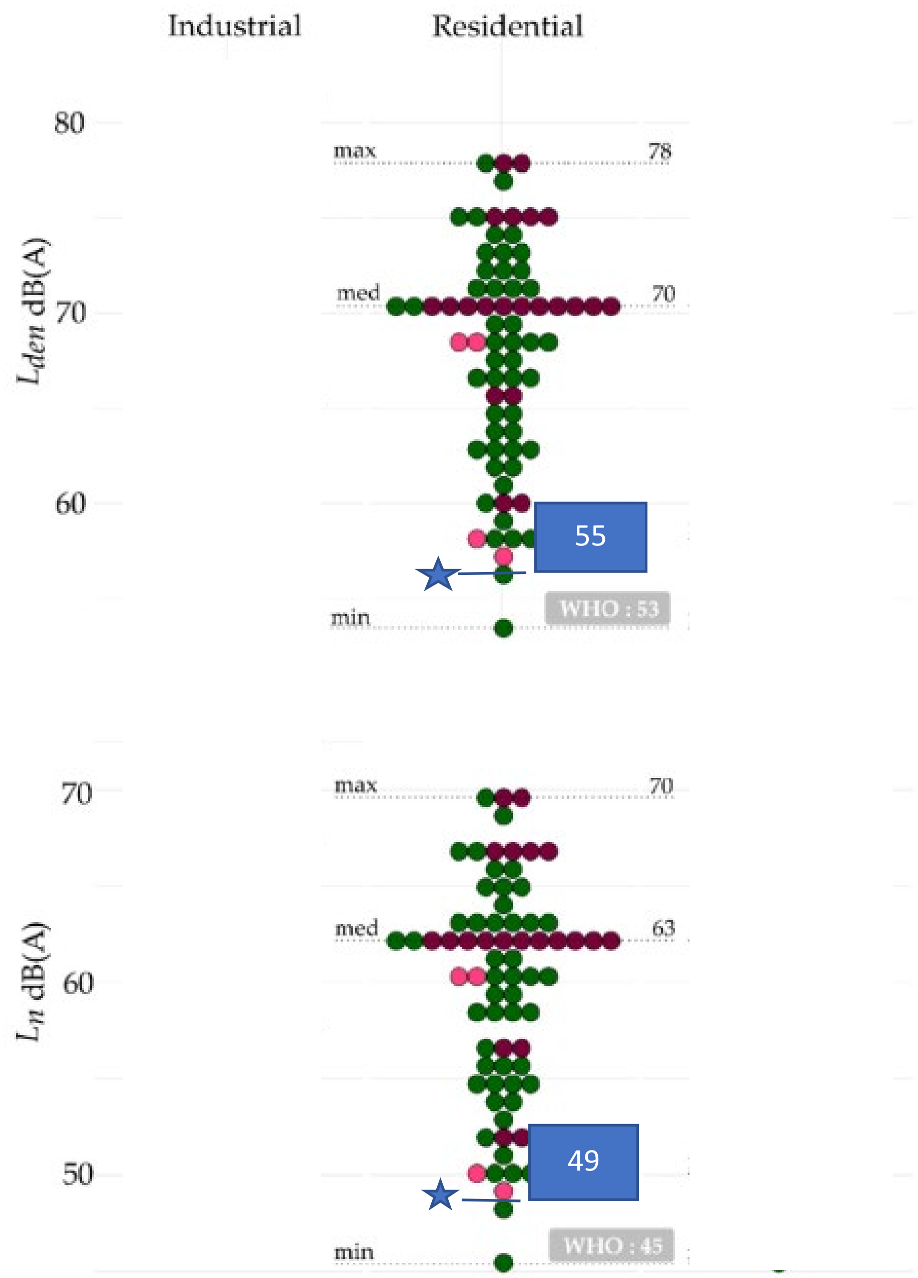



**Figure 3 Outdoor road traffic noise limit values for residential uses from a range of jurisdictions - Canada (pink dots), Europe (green dots), USA (brown dots) - converted to (free field) Lden, upper plot, and Ln (=Lnight) lower plot. Each dot corresponds to a single converted noise limit in one of the jurisdictions in Canada, the USA or Europe. The World Health Organization (2018) guideline levels for road traffic noise are also shown. Extracted and modified from Perna et al. (2021). These limit values are for *upgraded or new roads*. Blue stars are WA limit values – but see Table 1 for relativity of limit values between WA and other states, including NSW.**

## DIFFERENT RESPONSES/LIMITS FOR DIFFERENT CATEGORIES OF ROADWAY

### Roadways Differentiated by Development Stage:

Adopting different limit values for roadways in different development stages (new, upgraded, existing) is a policy/economic choice. In general, the scientific evidence cannot contribute to this choice. This is because limit values are derived from relationships between roadway noise exposure and human response to that exposure where, with few exceptions, the evidence has been obtained from ***existing*** roadways. This is not a statement that there could not be any difference in response to road traffic noise from new or upgraded roadways and from existing roadways, simply that there is no evidence of such difference found in available exposure-response studies. (An important qualification to this observation is that, if a new roadway or the upgrade of a roadway results in adjacent populations experiencing a significant change in noise exposure, that becomes a ***change*** situation – and responses may be different. See section: *'Interventions & Change Effects*' below).

There is, of course, an *economic* argument for using different limit levels on new and existing roadways. Authorities may not be able to afford mitigation to the same standard on all roadways, and prioritising noise mitigation expenditure on new roadways over that of existing roadways, if required, can be achieved by setting less stringent limit values for the latter. If such prioritization is the policy choice, there are options for the way a difference in limit levels between roadway categories might be interpreted for the public.

- The preferable way would be to explain the difference on costs/technical feasibility. A good example is the wording in the Western Australia State Planning Policy 5.4 (Western Australian Planning Commission, 2019): *viz: 'The 5dB difference in the criteria between new and upgrade infrastructure proposals acknowledges the challenges in achieving noise level reduction where existing infrastructure is surrounded by existing noise-sensitive development'.*
- Far less satisfactory is to specify the difference without justification. This opens the default misinterpretation that the policy maker assumes there is some scientific difference in effects of noise adjacent to new roads as compared to adjacent to existing roads. The consequence can be observed in comments recorded from a Vic Roads (2015) public consultation: '*Firstly, you need to make the VicRoads Noise Reduction Policy fair & remove the current unfairness and discrimination that surrounds the dB(A) level'; 'Have a consistent and fair Freeway Noise Policy - two different levels of noise measurement (i.e. 63 dB(A) vs 68 dB(A)), being applied to different sections of freeways & communities is ludicrous'; 'The fact that VicRoads Policy applies a lower standard of noise suppression or attenuation to this 'old' section of freeway is – in this day an age - both absurd and unfair to all of the residential homes and their occupants'.*

Two different criteria for exposure to similar conditions cannot be explained to the community. It is better to clarify that only one (scientifically-based) criterion is in use – but that application of mitigation has also been based on feasibility and economics. While this may seem a trivial distinction, this author suggest it is an important one to emphasize when policy purports to be evidence-based.

### Roadways Differentiated by Jurisdiction:

Similarly for roadways differentiated by jurisdiction – e.g., those under (national) state or local government responsibilities – there is no scientific evidence from existing exposure-response studies for any differentiation between such roadways in terms of human response to noise exposure.

### Intervention/Change Roadways:

Roadways where there has been some form of intervention resulting in a major change to the traffic carried on the roadway – say opening a new roadway, or the closure of an existing one, or some other intervention significantly affecting traffic capacity or traffic composition (lane closures, truck bans) - can be categorized as '*change'* roadways. There is evidence that human response to road traffic noise where conditions change may be different to human response under '*steady*-state' conditions. While the evidence is limited (by the limited number of studies of response to changing exposures) there is sufficient for it to be regarded as of consequence in traffic noise policy. This is because much road traffic noise management involves interventions that results in changed exposure. This topic is discussed in greater depth later in this document in the section: '*Interventions and Change Effects'*.

The provision of a new roadway represents a major change in noise exposure – sometimes elevating levels from quiet rural surrounds to high exposure levels. One example of an administrative response to this *change effect* is (partially – because this author subscribes to the concepts applied, but not necessarily the numerical values specified) described in Vic Roads (2015): '*The use of such a low noise target for new roads in previously very quiet settings is justified by extensive research on human response to changes in traffic noise level. Essentially this research compares the level of annoyance from an existing road with the annoyance from a new road. This research indicates that the level of annoyance from a new road that produces say a 60 dB noise level is greater than the annoyance from a road that has always existed and produces a 60 dB noise level. (Griffiths & Raw, 1990) (Brown & van Kemp, 2009)'.* This has also led VicRoads to adjust its limit values: in areas where pre-existing noise levels are very low, by abandoning its '*new roads'* 63 dB limit value, and instead set criterion as '*no increase of greater than 12dB'*.

### Roadways Differentiated by Functional Classification

This differentiation of roadways by function (e.g., freeway, arterial, distributor, local road) in terms of different roadway types generating different human response to traffic noise at the same exposure levels, is not part of the mainstream of exposure-response literature for road traffic noise[14]. It is mentioned here in this scoping study, primarily as a result of one Danish study that reported differences in noise annoyance (exposure-response functions) between residents near a motorway and near a major urban road (Bendtsen et al., 2014). The same authors have subsequently conducted their own (fairly selective) literature review, reporting it (Bendtsen & Pedersen, 2021) in a paper entitled '*Noise annoyance from motorways is worse than annoyance from urban roads*'. They argue their own finding are supported by the SiRENE study results in Switzerland, and by others (though an Alpine study (Lercher et al., 2008) shows the opposite result, with more annoyance from main roads than motorways). They conclude that the Danish $L_{den}$ limit criterion for roads should be 58 dB, but because motorways are more annoying, there should be a dual Danish limit criterion with $L_{den}$ for motorways set to 52dB.

The above authors suggest both these limits correspond to 10% of the community highly annoyed (as defined by the respective exposure-response functions for 'roadways' and for 'motorways' they adopted from their own studies).

A different (but speculative) functional classification has been identified by Parnell and Peng (2019). They differentiate 'Haul Route Traffic Flows' that carry consistent traffic throughout the night (primarily interstate key freight routes) from 'Metropolitan or Commuter Traffic Flows (primarily suburban and urban routes with low percentages of heavy vehicles)'. The authors also demonstrated that this also corresponds to categorization by an acoustic measure: *viz*. the difference $L_{day}$- $L_{night}$ with haul routes defined by a difference of 4dB or less. As above, there currently is no (non-anecdotal) evidence of difference human response in available exposure-response studies according to such a differentiation of road types. But this author suggests that further investigation may be warranted and speculates that the noise metric difference between the two roadway types[15] is likely to be closely related to different numbers of events heard within the respective night-time traffic streams (See '*Noise Events and Related Literature*' section of this document).

# ROAD TRAFFIC NOISE METRICS

It is not the intent to comprehensively review traffic noise metrics here – only to provide observations on metrics as they appeared, or were referred to, in the literature published in the review period. The simple observation is that it is nearly all about the EU metrics.

## DOMINANCE OF $L_{den}$ AND $L_{night}$ IN THE LITERATURE

The *Overview* sections above noted that most road traffic noise research and policy development, internationally, has been in Europe. It is hardly surprising, then, that the literature on metrics in the review period has almost universally focussed on those prescribed in the END (EU, 2002), viz. the A-weighted $L_{eq}$, over different periods with penalties for the evening and the night over the 24 hour period:

$L_{den}$ the day-evening-night noise level indicator.

$L_{night}$ the night-time noise level indicator.

These two metrics[16] tend (almost exclusively) to be the metrics that are:

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[14] Guski et al. (2017), Supplementary Material S23 (c) note: 'Motorway vs. urban roads: Based on a large European survey including more than 5,000 participants, Miedema [12, p. 33] concluded: 'At higher levels highways cause more annoyance than other road traffic'. The reference (Miedema, 1993) was not able to be accessed for this review.

[15] See also a possible parallel in the observations regarding the Noise Climate ($L_{10}$ – $L_{90}$) on different categories of road – see Figure 70 and related text.

[16] *Exact END definitions* of these $L_{den}$ and $L_{night}$, and related metrics, can be found in Annex 1 of EU (2002), and it is useful to reference these metrics back to that source.

- the outputs of most traffic noise modelling and mapping activity (or there is effort reported to convert other outputs to these metrics, or at least to $L_{eq}$-based metrics),
- used to report traffic noise exposures of communities,
- used when examining the human effects of road traffic noise, and hence form the exposure (or dose) axes of exposure-response functions, and
- while a variety of metrics are used in setting limit values and determining warrants for noise mitigation and action plans in different jurisdictions across Europe, many do use the $L_{den}$ and $L_{night}$.

Outside Europe, one can also observe some tendency for their use, at least in the research literature, in countries where adherence to the Directive is not required – e.g. some studies of exposure in the US (King et al., 2014). What this current author finds notable during the review period is that there appears to be an absence of alternative views in the scientific literature that different metrics to these should be considered for measuring most road traffic noise acoustic environments.

## NO EVIDENCE OF PROGRESS WITH END 'SUPPLEMENTARY NOISE INDICATORS'

It has been recognized for many decades that there may be some special road traffic noise environments where metrics other than $L_{den}$ and $L_{night}$ may be required. In fact, the END flagged this under the term *Supplementary noise indicators*, listing (in addition to $L_{day}$ and $L_{evening}$) candidate metrics and contexts at Clause 3 of Annex 1 (EU, 2002). Within the list, those relevant to road traffic noise include[17]:

- $L_{Amax}$ or SEL (sound exposure level) for night period protection in the case of noise peaks
- extra protection at the day period, or evening period, or weekend, or a specific part of the year
- quiet areas in open country
- where the noise contains strong tonal components or an impulsive character.

While this list of supplementary indicators in the END is now two decades old, and the possible justifications and potential advantage of their use recognised many decades earlier (including attempts at some implementation, as in Environmental Protection Authority, 1999) there has been very little reported in the review period on their research and development - or on the improvement in noise management their use would bring. The consequence is that one of the objectives specified in the brief for this review will not be achieved, viz.:

> *'Examination of whether community response to road traffic noise is adequately evaluated using equal energy indicators drawn from dose response approaches and/or whether maximum event assessment is needed. This is to identify what metrics should be considered to describe road traffic noise and its impact'.*

While there is yet no definitive answers to these questions, the available literature on noise event measures and the related dynamics of the road traffic stream are examined later in this document (see section: '*Noise Events and Related Literature'*).

Low frequency sound is briefly examined in the next section, followed by the limited literature available of rural/urban differences.

## LOW FREQUENCIES AND A-WEIGHTED METRICS

Low frequency sound is reviewed here only in terms of its effects on human response to road traffic noise – it often being mentioned anecdotally as a probable cause of adverse effects, particularly where the noise is heard indoors or where modified by barrier diffraction. Other road traffic noise issues with respect to low frequency noise such as: different spectral characteristics of components of vehicles or of different types of road traffic vehicles, road-tyre interaction emissions, outdoor propagation, transmission through building components, and diffraction around barriers, are not examined.

During the review period, there was limited work published on low frequency sound and human response to road traffic noise – a similar observation to that above regarding limited literature on alternative metrics for road traffic

[17] The END notes that any *Supplementary Descriptor* would require its own *limit value*

noise. There were no systematic reviews on effects of low frequencies in road traffic noise. A (non-exhaustive) search located only two peer-reviewed papers that focussed on this topic – one on annoyance and one on sleep. Both studies were quite restricted in terms of stimuli studied – though both were well-conducted experiments.

Noting that there was, at present, no consensus about the relative importance of low frequency content in urban road traffic noise, Torija and Flindell (2015) conducted subjective listening trials in the laboratory. Their results suggested that differences of at least 30dB between the low frequency and the mid/high frequency content are needed for changes in low frequency content to have as much subjective effect (on annoyance) as equivalent changes in mid and high frequency content. Another part of the same study, *'… aimed at investigating the relationship between loudness and annoyance under conditions where low frequency content (of road traffic noise) is relatively more dominant such as indoors where mid and high frequency content is reduced to a much greater extent than low frequency content because of selective attenuation by the building envelope'*[18]. Torija and Flindell (2015) found that: '*… changes in low frequency content appeared to make smaller contributions to subjective loudness and annoyance than might be inferred from the implied objective or physical dominance of those changes. The results suggested that commonly expressed criticisms of the A-frequency weighting based on an hypothesized excessive downweighting of the low frequency content may be relatively unfounded in this application area'*.

Myllyntausta et al. (2020) reported what they claim is one of the first studies to examine the effect of the road traffic noise spectrum on sleep. They compared, in a well-designed sleep laboratory experiment, the effects of two different road traffic noise spectra of the same overall sound level ($L_{Aeq}$ = 37 dB). Both objective (polysomnography) and subjective sleep quality were measured, but neither revealed differences in sleep quality between the two road traffic noise spectra exposures. However, the retrospective ratings after the whole experiment showed that the high frequency road traffic noise was perceived as a significantly more disturbing sound environment for sleep, and as a more annoying sound environment, than was the low frequency exposure. The spectral differences between the two stimuli they used in their experiment could reasonably be achieved by different building envelope properties alone. They also emphasized that the effect of the spectrum of road traffic noise exposures on sleep has to date had little investigation.

In summary, there is no new evidence that the presence of low frequencies in road traffic noise in common road traffic acoustic environments has an effect on human response not already accounted for by current A-weighted measures.

## RURAL VERSUS URBAN

There is only a little evidence reported, in the review period, regarding the use of/need for different, or adjusted, criteria in rural areas – as against criteria that have been developed using studies in urban areas. Theoretically, differences could be appropriate because any, or some, of the following might apply:

1. the relationship between noise exposure and response might be different between urban and rural populations (however, analogous to the argument above with respect to different categories of roadways, relationships between roadway noise exposure and human response have tended to be obtained predominantly from *urban* roadways – hence no available evidence for an urban/rural difference)
2. the nature of the traffic noise exposure may be different in rural areas. For example:
    a. quiet backgrounds against which passing vehicles are heard – particularly at night. An acoustic effect
    b. different noise emissions from vehicles (such as engine braking as highway trucks slow down at rural/urban transitions) – another acoustic effect
    c. the different context of rural areas and human expectations for those influencing the assessment of the acoustic environment – a 'non-acoustic modifier' in exposure-response terms or an 'expectation effect' in soundscape terms.

[18] See later section '*Window Attenuation for Road Traffic Noise'* for results of a Swiss measurement program (Locher et al., 2018) of indoor road traffic noise spectra resulting from open and closed windows (Figure 81).

There are two data sets reported in the review period that provide observations on different road traffic responses (perceived prevalence) in rural versus urban contexts. Both were national surveys that sought opinions of sampled national populations. The most recent is that reported by Michaud et al. (2022) who examined annoyance towards transportation (and construction) noise in rural, suburban, and urban regions across Canada. The other was the third in the series of National Noise Attitude Surveys, 2012 (NNAS 2012) in the UK (Department for Environment Food and Rural Affairs, 2014a). Neither survey measured nor predicted noise exposure of respondents – hence the only data available here is prevalence of annoyance with road traffic noise. The utility of both these studies for the current review is that they each used appropriate samples of the respective national populations – designed to be representative of adults in Canada and the UK respectively.

The Canadian Perspectives on Environmental Noise Survey (CPENS) study was designed, *inter alia*, to examine differences between Indigenous and non-Indigenous Canadians, and also differences between rural/remote, suburban and urban regions across Canada. Some 6000+ people were surveyed to provide insight into which personal and situational variables influence community noise annoyance, with the objective to investigate expectations, attitudes and responses toward environmental noise. Current Canadian guidance is that, in remote/rural areas, where an impacted community is assumed to have a greater expectation for and value placed upon peace and quiet, a so-called "quiet-rural area" adjustment may be applied in noise assessments[19] (Health Canada, 2017). The quiet rural area adjustment is the only community-based adjustment applied to date, and CPENS was designed to "test" it. An extract from Michaud et al. (2022) reads:

> *"With the objective to investigate expectations, attitudes, and responses toward environmental noise, CPENS showed an apparent increase in the overall prevalence of high annoyance toward road traffic noise (i.e., 8.5%) when compared to a prevalence of ~5% in surveys conducted 20 years earlier (Michaud et al., 2005, 2008b). Across the wide range of evaluated noise sources, the prevalence of overall annoyance was highest toward road traffic, then construction, aircraft/helicopters and rail noise, with no evidence in the data to suggest differences between Indigenous and non-Indigenous Canadians. The overwhelming majority of respondents indicated that they did not hear wind turbines, mining activity, industry unrelated to mining, and marine activity…*
>
> *Data from CPENS provided support for the aforementioned quiet rural area adjustment in rural/remote areas, affirming the implicit assumption that there is a greater expectation for, and value placed, upon peace and quiet in rural/remote areas."*

**Table 3 Weighted prevalence (% and 95% confidence interval) of Canadian population characteristics by Indigenous status and geographic location (highlighted in green). Extracted from Table 1 of Michaud et al. (2022).**

Weighted prevalence (%) and (95% confidence interval) of population characteristics by Indigenous status and geographic location.

| | Frequency (n) | Overall | Indigenous[g] | Non-Indigenous | Rural/remote | Suburban | Urban |
|---|---|---|---|---|---|---|---|
| Do you live in an area where you have a high expectation for tranquility, peace and quiet? | | | | | | | |
| Yes, definitely | 2380 | 35.8 (34.7–37) | 28.2 (23.6–33.4) | 36.2 (35–37.4)[h] | 58.2 (55.3–61) | 37.4 (35.8–39)[i] | 21.8 (20.1–23.6)[i] |
| Yes, somewhat | 3263 | 49.1 (47.9–50.3) | 51.4 (45.9–56.8) | 49 (47.8–50.2) | 36 (33.3–38.8) | 52.5 (50.8–54.2)[i] | 50.8 (48.7–52.9)[i] |
| No | 1004 | 15.1 (14.3–16) | 20.4 (16.3–25.2) | 14.8 (14–15.7)[h] | 5.8 (4.6–7.3) | 10.2 (9.2–11.2)[i] | 27.4 (25.6–29.3)[i] |
| With respect to outdoor noise levels, how often is the area where you live very quiet, calm and relaxing? | | | | | | | |
| Always/Often | 4059 | 61.1 (59.9–62.2) | 52.8 (47.3–58.2) | 61.5 (60.3–62.7)[h] | 76.8 (74.2–79.1) | 64 (62.4–65.6)[i] | 48.4 (46.3–50.5)[i] |
| Sometimes/Never | 2588 | 38.9 (37.8–40.1) | 47.2 (41.8–52.7) | 38.5 (37.3–39.7)[h] | 23.2 (20.9–25.8) | 36 (34.4–37.6)[i] | 51.6 (49.5–53.7)[i] |
| How often do you notice road traffic noise when you are at home? | | | | | | | |
| Always/Often | 1870 | 28.1 (27.1–29.2) | 35.3 (30.2–40.7) | 27.8 (26.7–28.9)[h] | 22.3 (20–24.8) | 26.3 (24.8–27.8)[i] | 34 (32–36)[i] |
| Sometimes/Never | 4777 | 71.9 (70.8–72.9) | 64.7 (59.3–69.8) | 72.2 (71.1–73.3)[h] | 77.7 (75.2–80) | 73.7 (72.2–75.2)[i] | 66 (64–68)[i] |

[19] It is important to note that this assessment in this Health Canada document is not of road traffic noise – it is designed for assessment of construction and operational noise – presumably of resource extraction projects - as part of Environmental Impact Assessment work.

That there is a ‘higher expectation for tranquillity, peace and quiet’; of ‘quiet, calm and relaxing outdoor noise levels’; and lower frequency of ‘noticing road traffic noise when at home’, is evident, in Table 3 above, for ‘rural/remote’ communities and ‘suburban’ communities than for ‘urban’ communities (see original publication for definitions). This, unsurprisingly, confirms that the nature of traffic exposures in rural areas aligns with points 2(a) and 2(c) above - the only added value being that it is measured data rather than anecdotal evidence. However, it is important to again emphasize that this is not evidence of a different response to intruding traffic noise in a rural area (*i.e.,* not evidence of a different rural/urban exposure-response relationship), only a difference prevalence regarding expectations of quiet/tranquillity.

The equivalent data from the 2012 UK NNAS was more ambiguous. Of the 2747 respondents surveyed in the random sample of the UK population, 25% responded “*moderately/very/extremely*’ to the question “*When you are at home, to what extent are you personally bothered, annoyed or disturbed by noise from road traffic?*”; 75% responded “not at all/a little”. The Odds Ratios for different geographical locations are shown in Table 4 (which are the odds for having a negative response to noise for each geographical category – compared to a reference group). The reference group (country village or small town) is indicated in the tables by an odds ratio of 1.00. It can be seen that the odds of being similarly annoyed in the countryside is half that of a small town, and that in the suburbs/outskirts of a large city 1.06 times that of a small town. The odds in the centre of a large city does not fit this pattern. The same qualifications apply to these UK observations as it did for the Canadian results – *viz.,* this is evidence of a difference in prevalence of traffic noise annoyance, not of a different rural/urban exposure-response relationship.

**Table 4 Odds Ratios for reporting being Bothered, Annoyed or Disturbed by Noise from Road Traffic for the UK population, according to geographic location (extracted from Department for Environment Food and Rural Affairs, 2014c).**

**A9 NNAS2012 When you are at home, to what extent are you personally bothered, annoyed or disturbed by noise from…ROAD TRAFFIC (n=2747)**

| | N | Odds ratio | 95% CI |
|---|---|---|---|
| **Is the dwelling located in … ?** | | | |
| The centre of a large city | 73 | 0.74 | 0.39, 1.40 |
| Suburbs/outskirts of a large city | 846 | 1.06 | 0.83, 1.35 |
| A large town or small city | 519 | 1.13 | 0.86, 1.48 |
| In a country village or small town | 1147 | 1.00 | |
| In the countryside | 159 | 0.50** | 0.30, 0.83 |

*p≤0.05, **p≤0.01, ***p≤0.001

While examining the 2012 UK NNAS results, the following Table 5 was also extracted – though it has no relevance to the rural/urban differences examined in this section.

It is reported here because it shows some interesting broad (population-based) results pertaining to the specific road traffic noise sources of those who hear road traffic noise at home. Somewhat counterintuitively (though see section: ‘*Roadways Differentiated by Functional Classification’* above), it can be seen in the UK population that it is single carriageway main roads and residential country roads that are the source of the greatest prevalence of high annoyance (compared to motorways or dual carriageway roadways), and that private cars generate the same prevalence of high annoyance as do heavy lorries.

**Table 5 The prevalence of being Bothered, Annoyed or Disturbed by Types of Roadway and Types of Vehicles for the UK population (those who hear road traffic noise) (Department for Environment Food and Rural Affairs, 2014b).**

| RTN1 NNAS2012 When you are at home, to what extent are you personally bothered, annoyed or disturbed by noise from…? (Those who hear road noise) 'Not at all – A little – Moderately – Very – Extremely' | | | | |
|---|---|---|---|---|
| **Noise Category (n=2270)** | **Hear % (n)** | **Bothered, annoyed or disturbed** | | |
| | | **To some extent** | **Moderately, Very, or Extremely** | **Very or Extremely** |
| **Types of Road** | | | | |
| Motorways | 35.0 (794) | 4.8 (110) | 1.4 (31) | 0.5 (11) |
| Other dual carriageway roads | 38.8 (881) | 7.8 (177) | 3.0 (67) | 1.1 (26) |
| Single carriageway main roads | 72.6 (1649) | 33.8 (768) | 14.8 (335) | 4.9 (112) |
| Residential / estate roads / country lanes | 84.8 (1925) | 34.8 (790) | 11.9 (270) | 3.3 (76) |
| Any other kind of road | 42.8 (971) | 1.3 (30) | 0.6 (14) | 0.2 (4) |
| **Types of Vehicles** | | | | |
| Heavy lorries | 76.4 (1735) | 34.7 (788) | 17.2 (390) | 7.0 (160) |
| Smaller lorries | 80.7 (1832) | 30.8 (699) | 13.5 (307) | 4.3 (98) |
| Delivery vans | 87.1 (1977) | 26.7 (605) | 9.7 (221) | 2.2 (51) |
| Buses / coaches | 72.2 (1638) | 20.7 (469) | 10.0 (228) | 3.0 (69) |
| Private cars / taxis | 92.6 (2101) | 38.3 (870) | 17.1 (389) | 5.7 (129) |
| Motor bikes / scooters | 88.5 (2009) | 40.7 (923) | 18.3 (416) | 7.0 (158) |
| Refuse collection | 91.8 (2084) | 24.5 (557) | 8.6 (196) | 2.2 (51) |
| Any other kinds of vehicles | 67.4 (1529) | 6.8 (154) | 3.6 (81) | 1.6 (37) |
| **Different types of Road Traffic Noise** | | | | |
| Vehicles starting | 85.6 (1942) | 18.5 (421) | 5.5 (125) | 1.6 (36) |
| Engine revving | 81.8 (1856) | 30.9 (702) | 11.3 (257) | 4.5 (103) |

Considerable care should be adopted in applying this population-based data from Canada and the UK to Australia because of different population and traffic densities, roadway networks, and built forms – but it is suggested that the broad observations around urban/rural differences in prevalence in hearing road traffic noise, and in being bothered by it, are likely to apply in the Australian context.

# BACKGROUND BOX 1

**Author's commentary on evening/night traffic noise period metrics, and possible perceptions by the public**

Notes in this Box are not sourced from the literature in the review period, but from the original '*Position paper on EU noise indicators'* (European Commission, 2000) and from the author's experience. They are included here as they do have potential policy relevance.

In considering measures such as $L_{Aeq}$ (or L10) over a 24h period, or some part of it including the night period, there are several narratives (anecdotes) that are relevant to the interface between the science and the public's (even some specialist's) perceptions.

One relates to concern that some metrics do not include the night period – such as was the case with the $L_{10,18h}$ metric - *viz:* '*How can this metric be appropriate if it does not measure the noise while I am sleeping?'*.

Another relates to metrics which include the night period and apply weights to the periods of night or evening (e.g. $L_{dn}$, and $L_{den}$). The narrative here tends to be along the lines: '*Such metrics are too complex to explain to the public*'.

Two observations can be made which may potentially assist negate both narratives.

Firstly, the evidence is that it makes little difference which of ***almost any*** of the alternative metrics is adopted (e.g. $L_{eq24\,h}$, $L_{day}$, $L_{evening}$, $L_{night}$, $L_{dn}$, $L_{den}$, $L_{10,18h,}$ etc), as they tend to be highly intercorrelated.

> *'From the validity viewpoint, there is not much difference between using evening and night-time corrections of 5 and 10 dB, no corrections at all or only night-time corrections. The reason is probably that in practice these values are closely correlated, so it is difficult to provide evidence for a preference for one over another'.* (European Commission, 2000 - p.33)

Secondly, while recognizing these high intercorrelations, the EU adopted the weighted metrics simply because they (without any evidence of scientific merit) might be more readily accepted by the public, accounting for their perception that these were significant periods of exposure that affected them at home. Again, from (European Commission, 2000 - p.33):

> *'It is surprising that there are relatively few studies demonstrating the benefits of these correction factors when it comes to predicting annoyance. It is almost certain that the origin of these factors is derived from the knowledge that night levels are (or were) usually about 10 dB lower than daytime levels, and evening levels were in between'* and '*From the public's perspective, it would appear to be odd to drop the correction factors, once it has been established that evening and night are sensitive periods. But as there is no compelling reason to deviate from current practice, the continued use of 5 dB for evening and 10 dB for night is recommended.*

In other words, it really does not make much difference adding 5 or 10dB weights, or when these are added, and there is no empirical support for them. They were adopted in the EU metrics primarily with the intent to maintain the impression, in the minds of the public, that the diurnal metric for road traffic noise management was both 'cognitive' of, and 'responsive' to, the importance of the evening and the night.

**Figure 4 BACKGROUND BOX 1: Commentary on evening/night traffic noise period metrics.**

# CHANGING UNDERSTANDING OF EFFECTS OF ROAD TRAFFIC NOISE

By far the largest shift within the scientific literature on road traffic noise in the past two decades has been the significant expansion of material focussed on the non-auditory health outcomes of exposure to environmental noise. These outcomes include cardiovascular and metabolic effects; effects on sleep; cognitive impairment; and mental health and wellbeing. The literature articulates, to various degrees for different outcomes, plausible pathways from noise exposure to effect; estimates exposure-response relationships for many outcomes; attempts to identify exposure levels that could limit adverse outcomes; estimates the number of disability adjusted life years lost that can be attributed to each outcome; and estimates the human health benefits arising from interventions that reduce traffic noise exposure.

Earlier literature had already identified these health outcomes - they have all been reported and even referenced in guidelines (e.g. Berglund et al., 1999; World Health Organization, 2010). The 2011 NSW Road Noise Policy itself carried sections on sleep disturbance from road traffic noise, other health effects[20] and effects on children's learning. The notable transformation in the review period and the previous decade has been that exposure to environmental noise is no longer seen as just a *quality-of-life* issue, with annoyance or dissatisfaction as the outcome of concern. Instead, the informed view is now through a *public health* lens with noise exposure leading to identifiable and significant costs to human health. This transformation is a fundamental shift that appears to have changed, and will continue to change, the dynamics and the politics of road traffic noise management. In fact, the brief for this review had noted *'…there has been some pivotal work in the area of noise impacts on communities including a paradigm shift that focuses on broader health impacts, and not just annoyance from noise'.*

A recent succinct statement of the focus of research and reviews on the effects of noise on humans has been provided by Faulkner & Murphy (2022) in an Applied Acoustics paper [original citation format modified, and citations edited]:

> *…approximately half the population of the European Union is estimated to be exposed to levels of road traffic noise considered to incur negative impacts on health and well-being, with road traffic noise considered the second most prevalent environmental risk factor, after fine particle pollution, to human health in Europe (Hanninen et al., 2014). An increasing number of studies have examined the impact of transportation noise and its association with annoyance (Dratva et al., 2010), sleep disturbance (Halonen et al., 2012), ischaemic heart disease and hypertension (Petri et al., 2021). More recent research has extended the investigation to include associations with respiratory conditions (Recio et al., 2016), diabetes (Clark et al., 2017), obesity (Oftedal et al., 2015), immune system dysfunction (Kim et al., 2017), cognitive impairment and psychological stress (Tzivian et al., 2017), foetal and childhood development (Christensen et al., 2017), with emerging literature proposing a potential link between environmental noise and cancer (Hansen, 2017).*

The finding that the public health risk of noise from road traffic is the second most prevalent environmental risk factor to human health in Europe, after fine particle pollution and equivalent to that from second-hand smoke, is from an Environmental Burden of Disease Working Group (Hanninen et al., 2014) who reported that road traffic noise is considered the second most prevalent environmental risk factor, after fine particle pollution, to human health in Europe (Figure 5). To generate effective policy measures and focus research efforts, it is important to prioritize environmental risk factors based on their health impact. Environmental burden of disease (EBD) measures are one way to do this by expressing different health effects in one unit, disability-adjusted life years (DALYs).[21]. Hanninen et al. (2014) report the EBD that was estimated for nine environmental risk factors (benzene, dioxins, formaldehyde, SHS, lead, traffic noise, $PM_{2.5}$, ozone, and radon) in six countries. The highest overall public health

[20] It can be noted that evidence regarding health effects was, quite reasonably for the time, regarded somewhat tentatively in the Road Noise Policy, e.g. '*Longer-term effects on health are more difficult to quantify, although tentative links have been drawn between noise exposure and heart rate, immune response, hypertension, blood pressure, occurrence of ischaemic heart disease, cardiovascular disease and myocardial infarction' (Department of Environment Climate Change and Water NSW, 2011)*

[21] DALYs give an indication of the equivalent number of healthy life-years lost in a population due to premature mortality and morbidity. See section: '*Burden of Disease and Health Impact Assessment'* later in this report.

impact was estimated for ambient fine particles ($PM_{2.5}$; annually 4,500–10,000 nondiscounted DALYs/million in the six participating countries) followed by SHS (600–1,200), traffic noise (400–1,500), and radon (450–1,100).

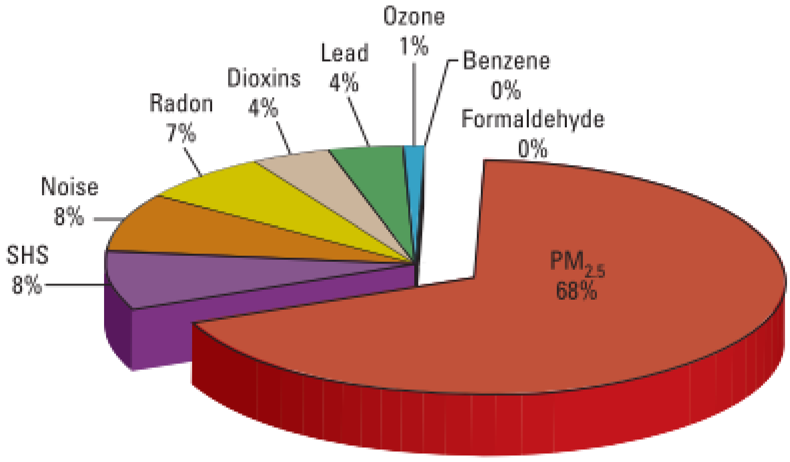



**Figure 5 Relative contributions of the nine targeted environmental risk factors to the estimated burden of disease attributed to these risk factors, averaged over the six participating countries (Belgium, Finland, France, Germany, Italy, and The Netherlands) – from the Environmental Burden of Disease in European countries (EBoDE) project. The figure is reproduced from Hanninen et al. (2014). SHS = second-hand smoke. While the noise is from all transport sources, the burden would primarily have been from road traffic noise.**

## Updating the Definition of Noise: 'Unwanted Sound' to 'Unwanted and/or Harmful Sound'

As part of the changing understanding of the effects of noise, it has been suggested that the definition of noise, in an environmental context, as *unwanted* sound, may no longer be adequate. Fink (2019) documents current definitions of noise and traces the origins and status of his proposal. The need for the change is that the phrase 'unwanted sound' implies that noise is merely a nuisance, ignoring what is now known about the harmful effects of noise. He suggests this outdated, subjective definition continues to adversely influence both public policy and our understanding of noise as a health and public health hazard. Given the evidence of health effects on humans, he suggests updating the definition to more faithfully recognize these effects, with a more apposite definition being '*noise is unwanted and/or harmful sound'.*

In 2021, a meeting of the International Commission on the Biological Effects of Noise (ICBEN) considered if they should update the noise definition, but the matter remains unresolved as yet. However, any future review of the NSW Road Noise Policy could usefully consider the status of this suggestion, internationally and nationally, at the time.[22]

[22] Whilst in the process of the literature search on the definition of noise, this author came across a recent paper '*Evolution in the law of transport noise in England*' (Simona et al., 2021). While of no direct utility to this review, it is an erudite examination of transport noise, including road traffic noise, and the regulatory framework that was developed around it over the centuries. It is recorded here only for the interest of professionals involved in noise management.

# PREAMBLE: EXPOSURE-RESPONSE FUNCTIONS - ROAD TRAFFIC NOISE

The brief for the current study required that it: '*Consider the current assessment criteria / recommended health protection levels and descriptors, and comment on their relevance and adequacy considering contemporary scientific research and literature including but not necessarily limited to WHO 2018 and enHealth 2018'* and, *'as relevant, discuss contemporary threshold levels and noise descriptors designed to afford communities with adequate levels of protection in terms of both adverse health outcomes and unreasonable levels of community annoyance…'*.

This requires, *inter* alia, examination of Exposure Response Functions (ERF) over the review period, for annoyance and for other effects of noise. Basner (2022) notes: '*One of the main goals of noise effects research is to derive exposure-response functions that can then be used for health impact assessments and ultimately to inform political decision making'.* This important role for evidenced-based ERFs is already accepted in Australia, at least for annoyance responses, as illustrated by: '*Regulators rely on scientific studies that provide insights into the effects of road traffic noise exposure on community response to inform decisions and develop strategies for protecting human health from exposure to excessive environmental noise*' (Parnell & Peng, 2019).

There has been continuing evolution, over more than four decades, of international material pertaining to exposure-response functions for different effects of road traffic noise - the first synthesis of results of previous exposure-response studies (only for annoyance) for transport noise was by Schultz (1978). This evolution has consisted of new ERF studies, new syntheses/meta-analyses of various sets of earlier ERFs, or as different interpretations and commentaries on the validity, applicability, and use of these various ERFs.

In considering how best to present an exposition of the ERF literature to assist NSW EPA examine current policy, this author took the view that, in addition to reporting literature over the review period, it would be useful (though somewhat unusual in a literature review) to provide some fundamentals on how ERFs are constructed, synthesised, and compared. These can help position a reader to make judgements on the validity and rigour of at least some of the various discourses on ERFs. These tutorial-like background commentaries are relevant for self-reported effects (such as the annoyance response) are in Background Boxes 2 and 3 below. Later sections provide information on exposure-response functions for health outcomes.

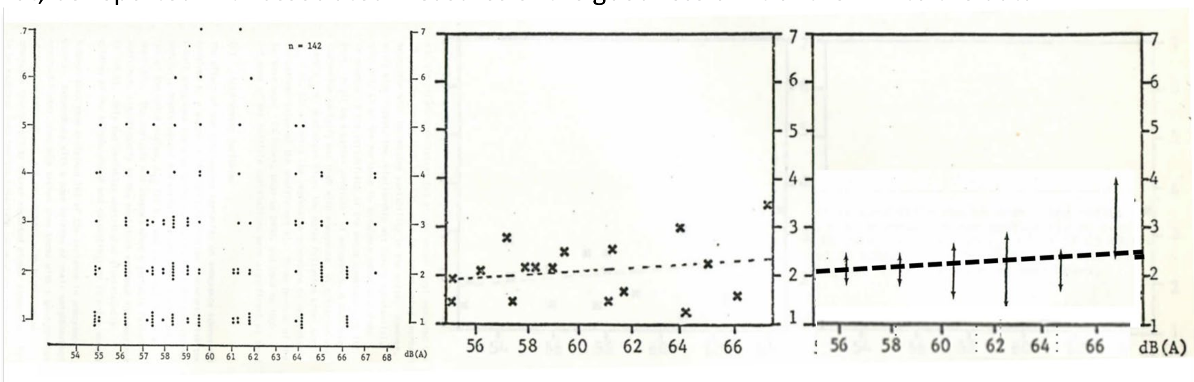
# BACKGROUND BOX 2

**A Single Study ERF: Individual ER Scores, Group ER Scores, and Confidence Intervals**

All exposure-response studies that rely on self-reported effects, such as annoyance or self-reported sleep disturbance, ***measures the responses of individuals***. In the figures below, the vertical axes are seven-point annoyance scales, and the horizontal axes the noise exposure (in this case, $L_{10,18h}$). The data in all three figures is the same: from an ER study of 142 respondents near a freeway, grouped around 17 noise exposure measurement sites (Brown, 1978). The example, metrics used, and results, are not relevant for this discussion paper. The sole purpose is to graphically illustrate the underpinnings of a study to determine an ERF for annoyance.

Diagram (a) shows the exposure and response for each of the individual 142 respondents in their homes. There are many issues with respect to measurement of response and exposure, but these are well covered elsewhere (Clark et al., 2021; Horonjeff, 2021; ISO, 2021). Figures such as this that show ***scores of individuals*** in ER studies are rarely reported by study authors, but the scatter should be kept in mind when looking at the types of results that are reported by authors – such as in Diagrams (b) and (c). ER studies group individuals subject with the same exposure – around measurements sites as in this example, or in exposure bins - then calculate some statistic of group annoyance score.

Diagram (b) shows the ***ER scores of 'groups' (or 'clusters)'***. In this example it is the *median* annoyance score of the group, but in most recent studies it would more likely be 'Percentage Highly Annoyed'. A line of best fit is also shown in (b), in this case fitted by simple linear regression – but again the actual approach to curve fitting is not relevant for present purposes (there are different approaches used). What tends to be reported from such studies is ***the ERF example as shown in (c) -*** *as the broken line*. ERFs reported from individual studies such as this may, or may not, be reported with associated measures of the goodness of fit of the ERF to the data[23].




(a) (b) (c)

For example, in (c), 95% confidence intervals have been calculated for the median annoyance score of subgroups of respondents within each 2.5dB exposure bin. While it is clear from diagrams (a), (b) and (c) that individual responses in exposure cannot be 'predicted' from the ERF or its confidence limits, the confidence intervals do represent the range of values within which the group response measure (in this case the median), is contained, with the probability of 1-α (e.g. with α =.05) While this is quite basic statistical material on confidence intervals, it will be seen in BACKGROUND BOX 3 that there is an important difference in the interpretation of such intervals for an ERF fitted to the data of an individual study as is this, and one that is synthesized from a set of different ERF studies.

Figure 6 BACKGROUND BOX 2: Individual and Group ER Scores & CIs for a best fit ERF.

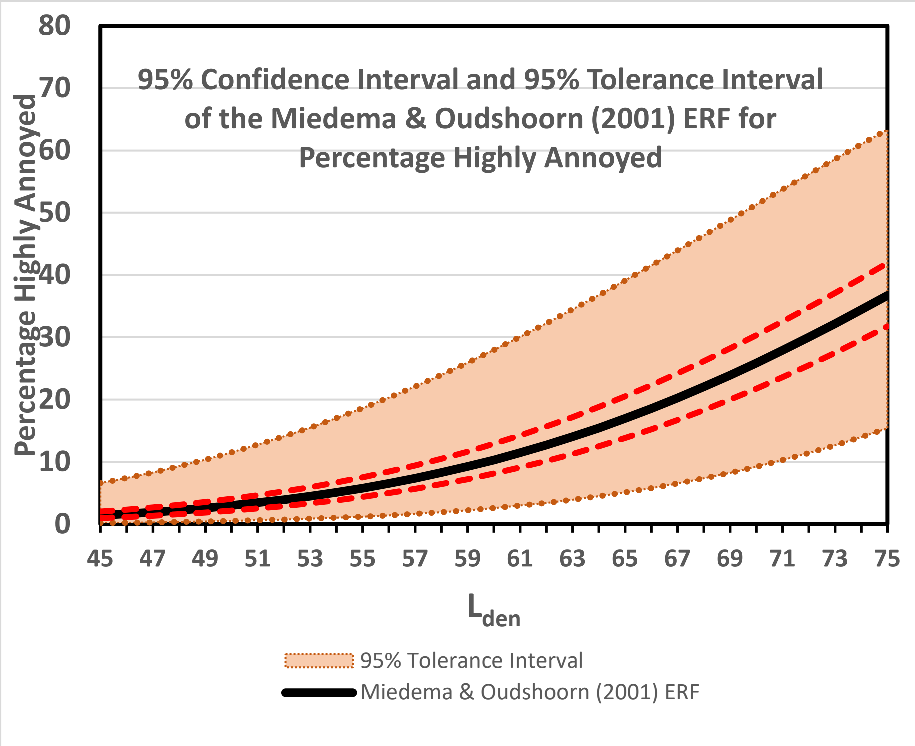
[23] Liepert et al. (2017) contribute to ERF formulation by considering the uncertainty associated with exposure measurement/prediction. They suggest, based on aircraft noise data, that the consequence of considering exposure uncertainty may not change the exposure-response relationship, but it will noticeably increase the confidence intervals around the ERF regression line.

# BACKGROUND BOX 3

**ERF Synthesised from a Range of ER Studies: Confidence Intervals and Tolerance Intervals**

Schultz (1978) synthesized the results of various social surveys of noise annoyance (synthesis of 18 different surveys). Others that followed included the US Federal Interagency Committee on Noise (1992) (aircraft noise only), Miedema and Oudshoorn (2001) (road, rail and aircraft noise separately), and, most recently, Guski et al. (2017). These syntheses or, more recently, meta-analyses (meta-analysis refers to the statistical analysis of the data from independent primary studies focused on the same question, which aims to generate a quantitative estimate of the studied phenomenon: Gopalakrishnan & Ganeshkumar, 2013) establish an ERF for some effect of noise, based on all the original studies, and can (but unfortunately do not do so in all cases) provide statistical intervals which indicate the association of the synthesised ERF with the original data.

For example, the Miedema and Oudshoorn (2001) synthesis for % Highly Annoyed from largely European road traffic noise studies, is shown below (solid black line). The authors reported the 95% *Confidence Interval*: upper and lower bounds are shown as red broken lines. Subsequently, 95% *Tolerance Limits* were made available (Sabine Janssen, TNO, personal communication 17 October 2013) and are shown, for 95% of the population, as red dot lines bounding the orange-shaded area. The original data points of the clustered data (from Miedema & Oudshoorn, 2001), and the Prediction Interval, are shown in Figure F.2 of ISO (2016).




Based on this Figure (synthesis of 26 individual exposure-response curves) one could:

(a) be 95% confident that for any additional ER studies the ***mean value* of ERFs from additional studies** would lie within the red broken line CIs shown, and,

(b) be 95% confident that 95% of the ***values*** that constitute additional exposure-response curves would lie within the orange-shaded Tolerance interval shown (Groothuis-Oudshoorn & Miedema, 2006). Expressing the latter another way: if an ERF from some new study lay within these tolerance limits (orange-shaded area) then it could be said, with 95% probability, that the new ERF is from the same population of studies on which the Miedema and Oudshoorn (2001) ERF was based.

The material in Box 2 and Box 3 is essential background when it comes to examining various commentaries on different exposure-response functions. To the author's knowledge, the (Miedema & Oudshoorn, 2001) ERF is the only one that has available **both** confidence and tolerance limits.

Figure 7  BACKGROUND BOX 3: Confidence Intervals and Tolerance Intervals for a synthesis of ERF studies

# THE WHO ENVIRONMENTAL NOISE GUIDELINES FOR THE EUROPEAN REGION

Systematic reviews of critical health outcomes were undertaken as part of the 2018 WHO Environmental Noise Guidelines (ENG) development. These were based on literature published in the period 2000 to 2015. Primary findings of each of the systematic reviews are outlined below. Different health outcome, and associated ERFs, are considered in turn.

Jarosinska et al. (2018) describe the development process for the ENG: [quote is edited]

> *'The WHO Environmental Noise Guidelines for the European Region will provide evidence-based policy guidance…on protecting human health from noise originating from transportation (road traffic, railway and aircraft)…Compared to previous WHO guidelines on noise, the most significant developments include: consideration of new evidence associating environmental noise exposure with health outcomes, such as annoyance, cardiovascular effects, obesity and metabolic effects (such as diabetes), cognitive impairment, sleep disturbance…impairment and tinnitus, adverse birth outcomes, quality of life, mental health, and wellbeing;…and the use of a standardized framework (grading of recommendations, assessment, development, and evaluations: GRADE)[24] to assess evidence and develop recommendations. The recommendations in the guidelines are underpinned by systematic reviews…as well as evidence on interventions to reduce…health outcomes'.*

The WHO ENG work was based on seven systematic reviews of the relationship between environmental noise and **(1) annoyance; (2) cardiovascular and metabolic effects; (3) cognitive impairment; (4) effects on sleep;** (5) hearing impairment and tinnitus; (6) adverse birth outcomes; and (7) quality of life, mental health, and wellbeing**.** An eighth systematic review assessed the effectiveness of environmental noise interventions in reducing impacts on health. Authors of each of the reviews are listed in Figure 8. The ENG developed separate guidelines for each transport mode, wind turbines and recreational noise – this current document focuses only on road traffic noise. **The outcomes in bold above were selected as the critical health outcomes for road traffic noise.**

The **key question asked in the systematic reviews** was:

- **What is the exposure–response relationship between exposure to environmental noise and the proportion of people with a validated measure of the critical health outcome, when adjusted for confounders?**

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[24] The Grading of Recommendations Assessment, Development and Evaluation (GRADE) has been adopted by national and international organisations as a systematic and transparent framework for evidence-based guideline development. While developed largely for the evaluation of clinical work (GRADE Working Group, 2013), it is being applied to public health, though there are recognized issues with its application (Rehfuess & Akl, 2013). The Table below interprets the four levels of evidence used in the GRADE profile. The GRADE methodology does not rate individual studies but is used to rate the overall quality of evidence available for specific noise source and health outcome.

| Grade | Definition |
|---|---|
| High | We are very confident that the true effect lies close to that of the estimate of the effect. |
| Moderate | We are moderately confident in the effect estimate: the true effect is likely to be close to the estimate of the effect, but there is a possibility that it is substantially different |
| Low | Our confidence in the effect estimate is limited: the true effect may be substantially different from the estimate of the effect. |
| Very Low | We have very little confidence in the effect estimate: the true effect is likely to be substantially different from the estimate of effect |

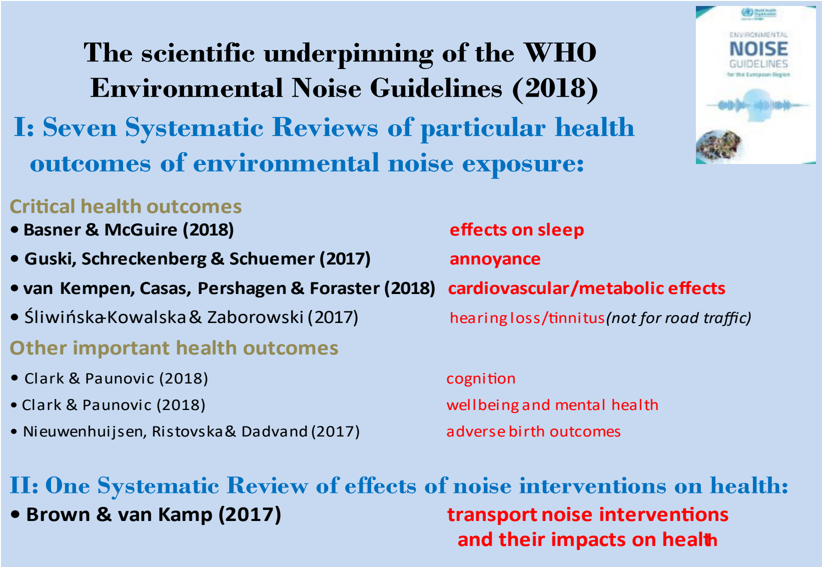



**Figure 8 Systematic reviews underpinning the recommendations in the World Health Organization (2018) ENGs for the European Region.**

The **key question asked in the interventions review** was:

- **Are interventions effective in reducing health outcomes from environmental noise?**

The reviews and exposure-response relationships for annoyance, sleep, cardiovascular and metabolic effects are examined in some detail below. The interventions review is considered in a later section of this document.

By and large, the systematic reviews did not test the efficacy of different noise metrics, instead focusing on $L_{den}$ and $L_{night}$ as these were the noise indicators of the EU's END. Despite adoption, in the title of the ENGs, of the '*for the European Region'* qualifier, WHO notes that the ENG '*…still have global relevance. Indeed, a large body of the evidence underpinning the recommendations was derived not only from noise effect studies in Europe but also from research in other parts of the world – mainly in Asia, Australia, and the United States of America'.*

However, before discussing each of the reviews, starting with '*Annoyance*', the next section reports on the new studies and materials that have become available subsequent to the WHO reviews (that is, post 2015)..

# NEW EVIDENCE AVAILABLE POST THE WHO REVIEWS

The systematic literature reviews undertaken for the WHO ENG update covered literature published in the period 2000 to 2015. Since then, many new publications have emerged describing results of existing and new studies that were not included in the WHO reviews. For example, the Department for Environment, Food and Rural Affairs (DEFRA) of the UK Government, on behalf of the Interdepartmental Group on Costs and Benefits Noise Subject Group (IGCB(N)), wished to investigate whether the WHO literature reviews would potentially benefit from an update – i.e. how many studies of good quality have been published since. As a result, van Kamp et al. (2020) (also see van Kamp, 2019; van Kamp et al., 2019) describes the results of a scoping synthesis of the literature of the effects of environmental noise on health, published between January 2014 and December 2019. The new literature was on sleep disturbance, annoyance, cardiovascular and metabolic effects, and the search protocols for the new papers followed those used in the WHO reviews. The quality of the new studies was indirectly assessed for cardiovascular and metabolic effects by including only studies with a case control or cohort design. For studies on annoyance and sleep disturbance, the risk of bias was expressed in exposure misclassification, selective participation, and confounding. The update yielded 87 papers, pertaining to 108 new studies of which 40 new studies were on annoyance, 42 on sleep disturbance and 26 cardiovascular and metabolic effects. See Table 6 below for how these were distributed across different noise sources (van Kamp, 2019) The number, size and quality of the new studies (particularly for road traffic noise) suggested new meta-analyses should be undertaken for the sources, and effects, included in the WHO reviews.

**Table 6 Number of new studies (January 2014 to June 2019) per source and outcome that could be used to update the WHO systematic reviews. A recommendation for an update of the meta-analyses is signalled by an asterisk (van Kamp, 2019).**

| | Air | Road | Rail | Wind |
|---|---|---|---|---|
| Annoyance | 13* | 10* | 8* | 9* |
| Sleep | 12* | 10* | 6* | 11* |
| Hypertension incidence | 2* | 8* | 2* | 1 |
| IHD incidence | 3* | 15* | 2 | 2 |
| IHD mortality | 2 | 5* | 2 | 0 |
| Stroke incidence | 3* | 8* | 2 | 1 |
| Diabetes | 3* | 3† | 2 | 1 |
| Change in body mass index | 1 | 3* | 3* | 0 |
| Change in waist circumference | 0 | 2* | 2 | 0 |

In fact, as will be seen in the following sections, updated reviews have now already been undertaken for the sleep and cardio-vascular meta-analyses and ERFs, and these are incorporated into the summaries provided below. An updated road traffic annoyance review and reanalysis has also recently been reported (August 2022).

The sections on *annoyance, sleep disturbance, cardiovascular and metabolic effects of noise* below examine the reporting of the 2018 WHO syntheses, as well as any updates/reanalyses of the information in the systematic reviews, together with any subsequent papers which are commentaries on any of these. Literature prior to the WHO ENG work is included only where essential.

# ANNOYANCE

## THE WHO 2018 ENG SYNTHESIS OF ANNOYANCE STUDIES

The systematic review and meta-analyses of effects of road traffic noise on annoyance, undertaken for the WHO (2018) Environmental Noise Guidelines revision, was reported by Guski et al. (2017). The quantitative meta-analysis of road traffic noise was based on nine publications describing 26 studies of response to road traffic noise reported in the review period (2000-2014). Rigorous criteria were applied to the selection of studies but, as will be seen below, the outcomes of the selection proved challenging.

The objectives of the Guski et al. (2017) systematic review were:

1. to assess the strength of association between road traffic noise exposure and annoyance
2. to quantify the increase of annoyance with an incremental increase in noise exposure, and
3. to present an exposure-response relationship.

**Results re objectives 1 and 2**: Meta-analysis on subsets of the studies (depending on data availability) showed **moderate** quality of evidence (see GRADE definitions in footnote above) for statistically significant correlations between noise levels and (group) annoyance scores ($r = 0.325$; $p < 0.001$). It also showed an Odds Ratio (OR) for the increase of observed percentage of people highly annoyed (%HA) per 10 dB level increase greater than 1 that was statistically highly significant (OR = 2.738, 95% CI = 1.880–3.987; $p < 0.001$). Similarly, meta-analysis of slope parameters of the ERFs for individual studies obtained by logistic regressions also showed increase in %HA for 10 dB increases in level (OR = 3.033; 95% CI = 2.592–3.549; $p < 0.001$). The quality of evidence has been judged **moderate** (GRADE) in the case of original %HA data and **high** (GRADE) in the case of the logistic regression model data [25].

[25] The nature of the analysis described in this paragraph, and the way it is expressed, may be somewhat unfamiliar to environmental noise managers. It is a demonstration of the rigorous approach WHO now requires in all areas of health to justify claims that there is a measurable change in human health outcomes with a change in exposure to some agent. Put simply, the information in the paragraph is evidence that there is an increase of annoyance with an incremental increase in noise exposure. See the Jarosinska et al. (2018) reference to an evidence requirement, and for use of the standardized GRADE framework to assess evidence and develop recommendations.

**Results re objectives 3**: There was extreme variation of average %HA across the full level range (See Figure 10). The quality of evidence assigned to the ERF for % HA was ***low*** (GRADE). In an attempt to resolve whether study selection influenced this outcome, the authors reported two ERFs, one for the 'full WHO data set' and the other 'excluding Alpine and Asian studies' (Figure 9). Both ERFs were regarded as 'tentative'. Quadratic equations have been provided for both the ERFs. At the critical lower 40-60dB levels, the 'excluding Alpine and Asian' ERF was closer to the Miedema and Oudshoorn (2001) ERF than was that for the full WHO data set.

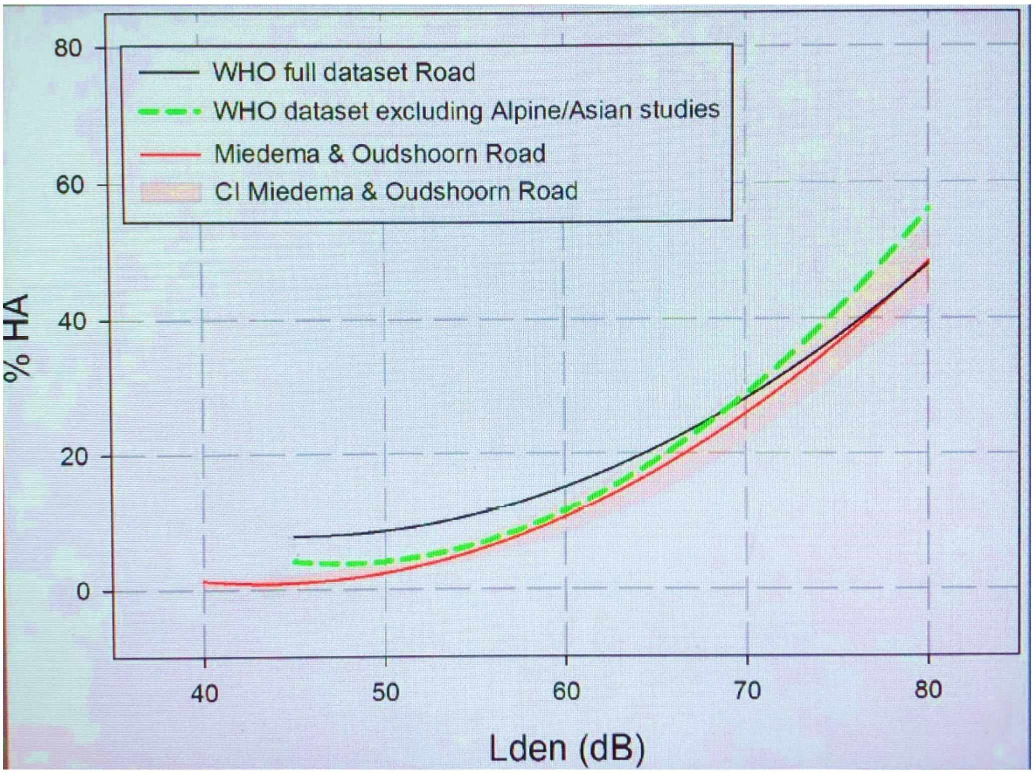



**Figure 9 Quadratic regressions of the relation between $L_{den}$ and the calculated %HA for 25 road traffic noise studies, ('full WHO data set', black) vs. 10 studies (dashed green, same data set excluding the Alpine and Asian studies). For comparison, the exposure-response function by Miedema and Oudshoorn (2001) (road) is shown (red), together with its pink-shaded confidence intervals (CI).**

The %HA at 5dB exposure intervals[26] for each of the studies is plotted in Figure 10a. There is clearly a major study effect in the results, with the ERFs of some studies consistently being higher or lower than the overall ERF results. Notably, the valley (Alpine) studies have very high %HA across the exposure range, and three of the Asian studies have predominantly low responses (and a highly restricted range of exposures 65-75 dB within the study). The large (10,077 respondents) Hong Kong[27] study was also classified as an Asian study, though the %HA was more closely aligned with the Miedema and Oudshoorn (2001) ERF.

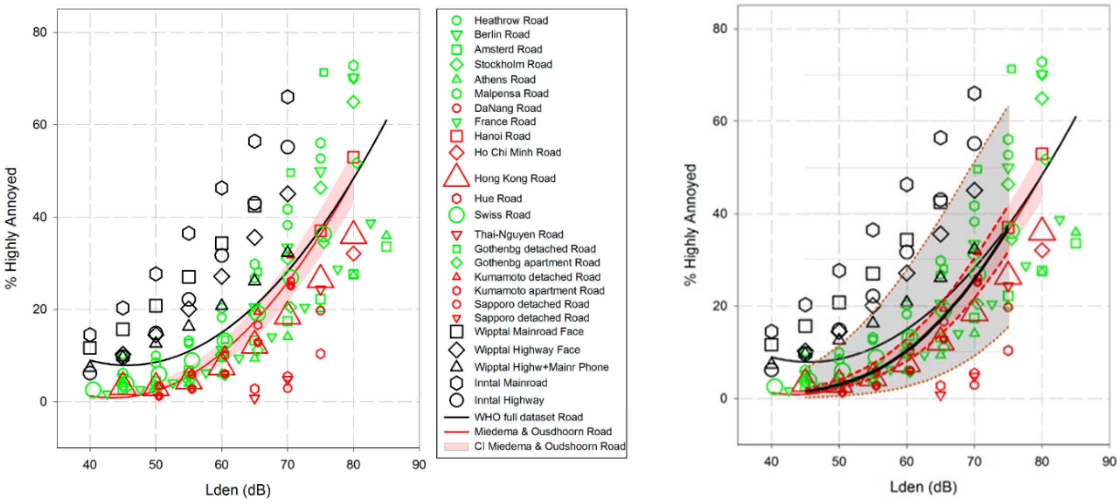
The appropriate way to examine how the ERF from each of the studies compares to the Miedema and Oudshoorn (2001) ERF synthesis is illustrated in Figure 10 (b), where the Tolerance Limits of the synthesis have been plotted (See Box 3 and explanations there). Figure 10(b) shows that most of the ERFs from the set of studies examined by Guski et al. (2017) fall within these tolerance limits (shaded area) and thus, with 95% probability, can be said to come from the same population of studies as that on which the Miedema and Oudshoorn (2001) ERF was based. It is only the four Alpine valley studies, and the three Asian studies with the restricted 65 to 75 dB range, that are from

[26] The %HA are not the aggregate data points from the original studies but are the *calculated* %HA based on the ERF reported in each of the original studies.

[27] The current author was a chief investigator of the Hong Kong study (see Brown et al., 2015)

different populations. That the Alpine studies were associated with ERFs that were higher than various ERF syntheses for annoyance has been known for two decades (Lercher, 1998; Lercher et al., 2008).

In summary, while there is no definitive explanation as to why the results from these seven studies are different (Guski et al. (2017) note that the comparability[28] of the Alpine studies with studies from more or less flat landscapes, as well as the comparability of studies with and without air-conditioned homes (Asia?) may be questioned. The very restricted 65-75 dB range in some studies, and the fact that the five Alpine studies are the only studies left that provide %HA at 40 dB, are other differences). Each of the **other** new studies reported from the 2000-2014 period was not particularly different from the set of studies that Miedema and Oudshoorn (2001) used in their synthesis.




(a) (b)

**Figure 10 (a) ER 'curves' (scatterplots) of all road traffic noise studies (Guski et al., 2017) and overall quadratic regression ERF (black line), together with the Miedema and Oudshoorn (2001) ERF (red line). Black symbols refer to 'Alpine' valley studies, red symbols refer to 'Asian' studies, and green symbols refer to 'European non-valley studies'. Panel (b) shows the same as (a) but has been overlaid with the Miedema and Oudshoorn (2001) ERF tolerance limits (see Box 3). The percentage of people highly annoyed by road traffic noise, and the uncertainty. increases with exposure level.**

## COMMENTS/CRITIQUES OF THE Guski et al. (2017) ANNOYANCE SYNTHESIS

### Dialogue with Gjestland (various)

In a series of papers, Gjestland is critical of the WHO recommendations for limiting annoyance – with most of his focus on aircraft noise as well as on road traffic noise. Gjestland (2019a) argues that WHO's new guidelines are based on a non-representative selection of existing studies whose findings cannot be generalized to the residential population at large. He notes that, for road traffic noise, the new WHO guidelines represent a lowering of the 'safe' exposure level by 3 to 5 dB compared with WHO's former 2000 recommendation. He is critical of:

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[28] Guski et al. (2017) raise the question of the 'comparability' of certain studies. This is the term they use, but they do not define it nor provide specific data to support their concern. The inference, though, is that, while external exposures of dwellings in terms of $L_{den}$ may be the same across the included studies, there could be something else that might be different about the exposures in the different studies such that they should not be bundled together in an ERF synthesis. For example, in the situation of the Alpine studies, *inter* alia, vehicles passing along the valley will have very much longer pass-by durations when heard by elevated respondents living on the valley sides, and there could be very different ground and other propagation effects from sources traversing the valley floor to elevated respondents than would occur for respondents in what they term 'more or less flat landscapes' (which presumably is most sites from all other studies included in the synthesis). Their comment on the Asian studies is that air conditioning in tropical Asian homes could result in quite different internal exposures (in this author's opinion, double glazing as used extensively in European dwellings could, equally, be a source of non-comparability). However, all such observations are *ex post* facto and should have been addressed in the selection process for studies before the synthesis was undertaken. Notwithstanding such *post*-hoc commentary, as Schreckenberg and Hong (2021) and others note, study selection for all systematic reviews was done according to a strict protocol set by the WHO Steering Group and the Guideline Development Group.

- the J shape of the ERF (increasing %HA at the lowest exposures) suggesting the J tail was an artifact of the quadratic ERF fit
- the Alpine studies using a different definition of HA with consequence of elevated ERF (large black symbols in Figure 10 above).
- the HYENA studies (six of the small green symbols in Fig 9 above) being used as they were restricted to >45 years of age, and also used a 'non-standard' annoyance question (separate annoyance questions for day and night, rather than overall annoyance.

Guski et al. (2019) responded (the response authorship was the original three authors of the annoyance systematic review – Guski, Schrekenberg and Scheumer, together with two members of the WHO Guideline Development Group – Brink and Stansfeld), as again did Gjestland (2019c); (Gjestland, 2020b). But the debate was to some extent primarily with respect to the aircraft noise synthesis, though largely similar arguments flowed through to road traffic noise results. A significant part of the debate centred around the use of the Community Tolerance Level (CTL) methodology. This CTL approach, along with regression analysis, is included in ISO1996-1 (ISO, 2016 Appendices E, F and H)[29], specifies relationships to estimate the percentage of a population highly annoyed with transport noise.

Gjestland (2020a) also published a paper (part published previously as a conference paper (Gjestland, 2019b)) in which he applied the CTL methodology to 61 road traffic annoyance surveys conducted between 1969 and 2015. He plots the CTL for each of these against the year of the study, suggesting that this is an examination of how response to road traffic noise has varied over time. This is reproduced in Figure 11 below. He uses this to argue that there has been virtually no change in the average annoyance response over the past 45 years – in contrast to the results of Guski et al. (2019) who claimed annoyance had increased in their WHO synthesis when compared to earlier results. Gjestland (2020a) suggests this is further evidence that the new lower recommendation from WHO regarding road traffic noise is not supported by existing evidence. Brink (2020), in a letter to JASA, suggests that Gjestland's work has serious shortcomings itself, including lack of relevant information, data sources and data extraction, and that no efforts are made to substantiate the selection of studies underpinning the analysis.

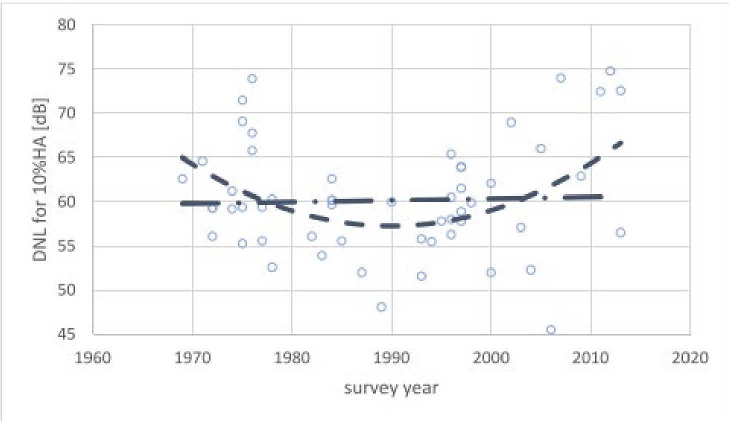


**Figure 11 CTL values for 61 surveys on road traffic noise conducted between 1969 and 2015. The two lines show a linear regression function (dash–dotted) and a second order polynomial regression function (dashed). Figure from Gjestland (2020), who used it to argue that response to road traffic noise has not varied over time (or response increased slightly to about 1990, then decreased again to the present. See, also, Figure 12.**

[29] Despite its inclusion in the international standard ISO1996-1 (ISO, 2016), the current author does have concerns regarding the methods of the CTL vis-à-vis regression analysis to estimate ERFs. To date it is a relatively limited number of authors who promote its use. Brink (2020) provides a sample of these concerns as part of a dialogue with Gjestland: '*Data extraction procedures and the statistical modelling approach are not detailed at all, just vaguely referenced. No information is provided about exposure-response pairs having been available to the author on the level of individuals or in aggregated form, or if they have been manually extracted from published tables, or even reconstructed from formalized exposure-response relationships.*'

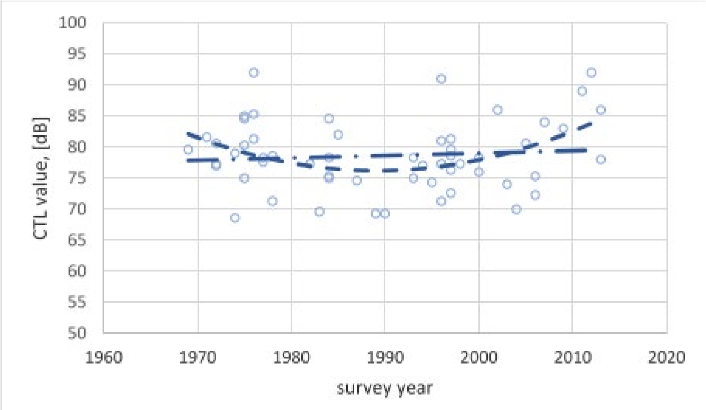


**Figure 12 This is the same data as the previous Figure but shows the noise exposure level at which 10% of respondents are highly annoyed. Calculated for 61 surveys on road traffic noise conducted between 1969 and 2015 using the CTL method.**

## Schreckenberg and Hong (2021)

Schreckenberg and Hong (2021), in their ICBEN review of research on community response to noise 2017-2021, rehearse the issues in the above debate on the WHO annoyance synthesis. These included: the selection of studies for the synthesis; the methods used to model %HA against $L_{den}$ in the WHO review; that other selections would lead to the conclusion of no change in the exposure-response relationship for aircraft noise annoyance over the decades; and lower %HA per $L_{den}$ than presented in the WHO review. A similar concern was expressed for road traffic noise. They recognized that different input data, and different methods, will lead to different results. They also noted that, because of this, the annoyance synthesis had worked within the strict study protocol for the systematic reviews provided by the WHO Steering Group and the Guideline Development Group (GDG).

They also note the debate is ongoing. Suggesting that all discussants agree that there is a large variance in the level of annoyance at a given sound level. So general exposure-response functions will not be able to reflect the local situation. For local noise management…exposure-response information as tailored as possible to the situation in local communities would be more helpful for noise interventions that aim to minimise noise health effects, including annoyance. This view, to use, if they are available, local ERF, is mentioned by several other authors.

van Kamp (2019) reported the availability of new studies subsequent to the completion of the WHO systematic reviews (2014). She noted, in view of the issues raised by Guski et al. (2017) and the current debate (Gjestland, 2018; Guski et al., 2019; Gjestland 2019) regarding the effect of selection of studies in the WHO meta-analyses on the Guideline values, that closer examination of new studies would be worthwhile from a scientific, as well as a policy point of view. That was not intended as a criticism of the systematic review conducted by Guski et al. (2017); more a comment on the nature of the studies/data that Guski was required to review under the selection criterion provided in the WHO brief for the noise effect systematic reviews.

## The Parnell and Peng (2019) critique of the WHO annoyance synthesis

The critique of the WHO synthesis by Parnell and Peng (2019) was based on comparison of various ERF studies reported in the NSW Road Noise Policy with the ERF synthesis by Guski et al. (2017) and, earlier, the synthesis by Miedema and Oudshoorn (2001).

The 'original' set of ERF curves used in formulation of the Road Noise Policy is reproduced below in Figure 13, panel (a). This shows the aggregate annoyance scores at different levels of exposure for four different exposure-response studies, together with the Schultz (1978) synthesis. The exposure metric used is the $L_{day}$ (including façade correction) as used in the NSW Road Noise Policy. Parnell and Peng (2019) suggest that some of the scatterplot data in panel (a) may be plotted incorrectly– '*The discrepancy observed in the original EPA synthesis plot…is likely to be attributed to* *the conversion error from $L_{dn}$ to $L_{eq(day)}$)'.* They provide an 'updated' plot (Figure 13 panel (b)) in which the Nemecek and Hall ERFs are plotted as smoothed curves (with $L_{den}$ as the exposure metric[30] without façade correction).

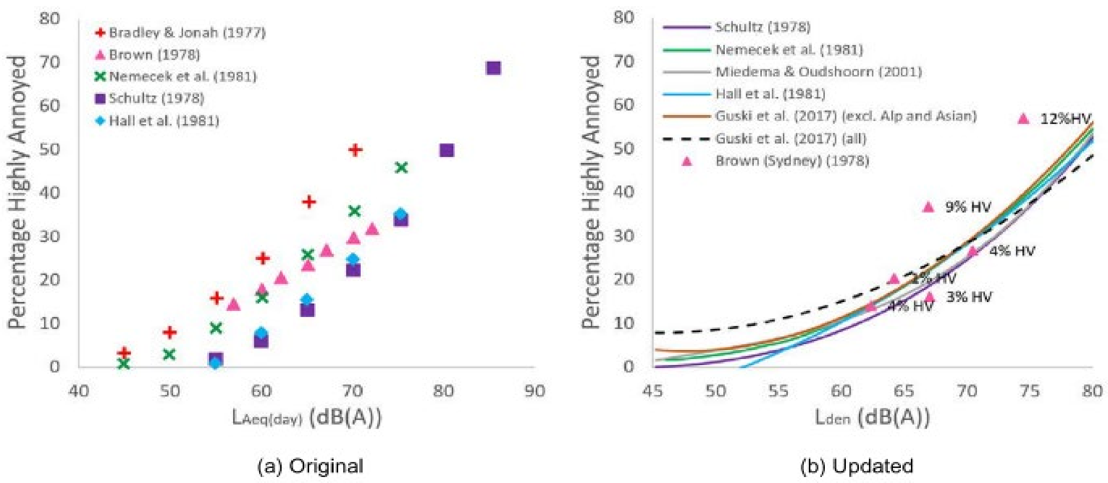


**Figure 13 Reproduction of Figure 6 (from Parnell & Peng, 2019)'**

From this Parnell and Peng (2019) perspective, this facilitated comparison with the ERFs synthesized by Miedema and Oudshoorn (2001) and Guski et al. (2017) - the latter including ERFs of both the 'WHO full data set' and the ERF 'excluding Alpine and Asian studies'. Parnell and Peng (2019) also included in panel (b) the %HA from six individual roadways in Sydney reported in the BSM study by Brown (1978) (purple triangles), and the Schultz (1978) synthesis. The way Fig 12 (b) has been plotted clearly encourages the reader to see that most of the curves follow a similar trajectory across the whole exposure range – ***except*** for the Guski et al. (2017) ERF (all) which suggests greater prevalence of high annoyance by perhaps 3 to 6 percentage points for exposures up to 65-70 dB. In particular, the Guski ERF suggests higher prevalence of high annoyance at low levels – and this appears to be the major point of contention with various commentators. The Guski ERF that excludes Alpine and Asian studies is more closely aligned to the other syntheses. These observations that the Guski (WHO full data set) ERF is different, is identical to the observations already made above with respect to the plots in Figures 9 and 10.

Some of the comparisons encouraged by Figure 13(b) (Parnell & Peng, 2019) are not valid. Of the six different plots, the (four) syntheses of Guski, Miedema & Oudshoorn, and Schutz can be compared. But it is not appropriate to compare the ERFs of the individual studies of Hall or Nemcek[31] directly with the ERFs of the synthesized studies, and the same for the six single-road data points from Brown. The reason can be seen by referring to the Background Boxes 2 and 3: the six data points are the 'x's in Box 2 panel (b); the individual study ERF is as in panel (c); and the ERF synthesis of a set of studies is as in Background Box 3.

The purpose of Figure 14 is to demonstrate that, while the plotted ERFs from individual studies (Nemeck, Bradley & Jonah, Brown, Hall et al.) show that the %HA at most levels of exposure will cover a large range (e.g. 8% to near 26% at 60dB – also see the Prediction Interval at this level in Figure 15, and accompanying text, below) nearly all of the data points of the ERF studies fall within the tolerance limits calculated for the Miedema and Oudshoorn (2001) ERF synthesis. Thus, with 95% probability, each of these four ERFs can be said to have been drawn from the same population of ERF studies as that on which the Miedema and Oudshoorn (2001) ERF was based (See Box 3). This point has been laboured here somewhat to emphasize that where an ERF has been synthesized from a set of original studies, that set will likely consist of ERFs with quite different relationships between exposure and response.

[30] It is not reported, nor possible to observe, how the errors in the original were corrected (scatterplot in the original, smooth line in the updated), nor how the Brown %HA data points were 'de-constructed'. This makes it difficult for a reader in attempting to follow the argument being made by the authors, and in replicating the observations.

[31] This consisted of four Swiss studies for which the results were pooled.

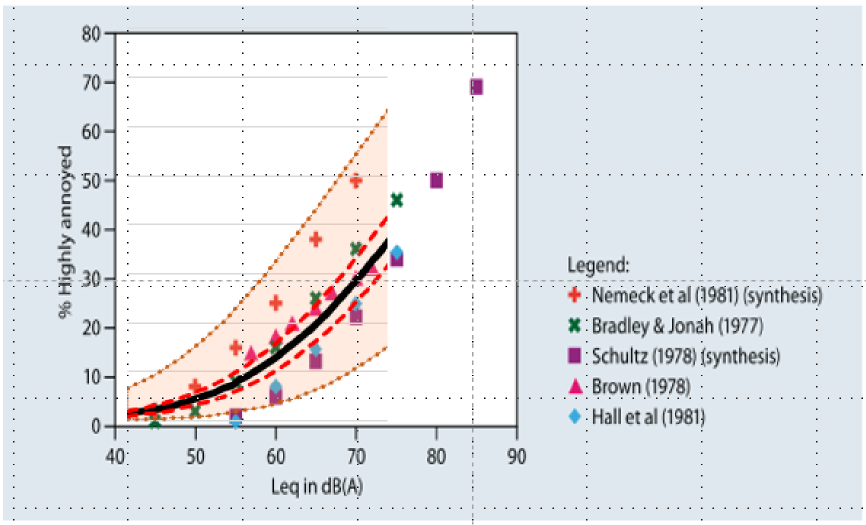


**Figure 14 This figure (originally from p.31 of the NSW Road Noise Policy) is the same as Figure 13 (a), viz., a selection of ERF studies/syntheses with an exposure axis of $L_{eq}$ for the day period ($L_{day}$). The scatterplot has been overlaid with the Miedema and Oudshoorn (2001) ERF (black line), its 95% confidence limits (broken red line) and 95% tolerance limits (orange shaded area) - see Box 3. Figure 14 has been constructed for the current document by graphical overlay rather than by replotting originals, and thus should be regarded as approximate only.**

What should be clear by now from the observations in this report, is that there is always a very large scatter in %HA at any exposure level. And while utilization of available ERFs remains the appropriate way in which to consider the setting of limit criteria, there has, in the past, been too much precision assumed with respect to the responses. That is, there has been a tendency to associate a precise %HA with each particular exposure level. It is important that policy makers appreciate that they should be referring to a wide band of response, and central tendencies of responses, rather than exact percentages of response. Any documentation should take the public along with them in this.

An indication of the 'wide bands of response' is available in Appendix F[32] of the International Standard ISO 1996 (ISO, 2016) '*Description, measurement and assessment of environmental noise*'. Figure 15 below is copied from that

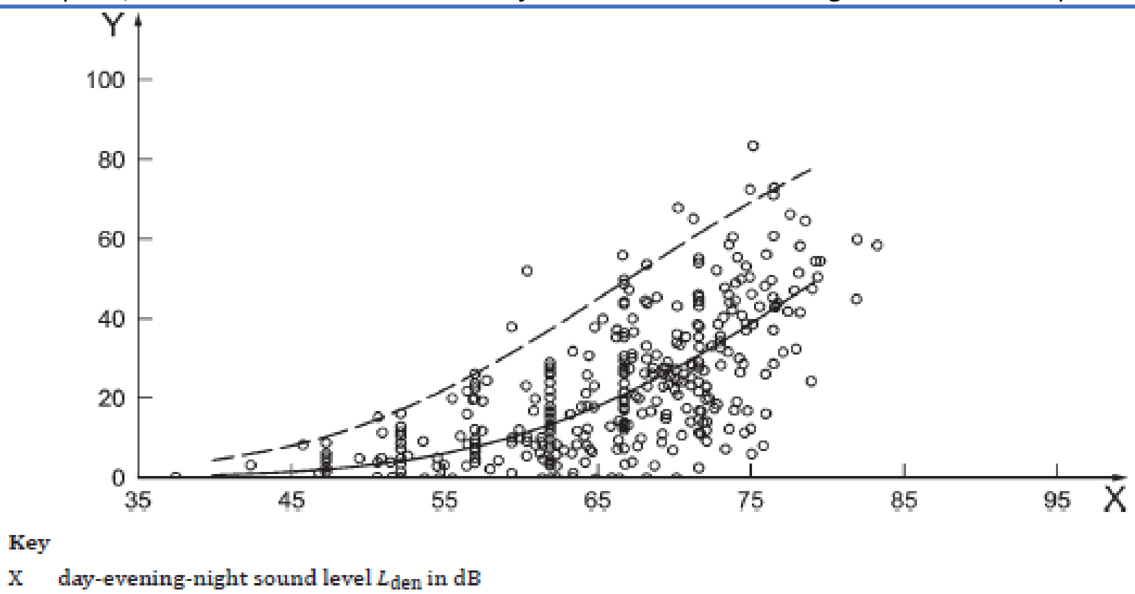


**Figure 15 Prevalence of high annoyance to road-traffic noise (solid line) versus $L_{den}$, based on (Miedema & Oudshoorn, 2001) and the corresponding 95 % prediction interval - upper limit as dashed line (figure from ISO, 2016).**

[32] This author finds Appendix F of ISO 1996 to be very confusing. This is a reflection of the way two different "methods" for estimating prevalence of highly annoyed with road traffic in a population (%HA) have been included – first, an assessment based on "regression" analyses (such as Miedema & Oudshoorn, 2001) and, second, an assessment based on the Community

Appendix. It shows the prevalence of high annoyance (%HA) with road traffic noise at each exposure level ($L_{den}$). It also shows the upper limit of the 95% prediction interval (the lower limit is zero across the range of exposures). For example, at an $L_{den}$ of 60 dB, where the figure shows the mean %HA to be about 10%, the 95 % Prediction Interval covers a range from 0% HA to some 32% HA[33].

### Other comments on the WHO recommendation: Fenech and Rodgers (2019)

In their ICA conference paper, Fenech and Rodgers (2019) note that, following the publication of the Environmental Noise Guidelines for the European Region (ENG), a lot of debate has focused on the Guideline Recommendations, in particular the specific guideline levels and the strength of the recommendations. The authors, from Public Health England, also noted that a sizeable proportion of the UK population was currently exposed to noise levels above the WHO recommendations. This paper is noted here only because it illustrates the difficulties for policy and decision makers when they are faced with a new set of scientific reviews, exposure-response functions, and limit values which do not conform to those already implemented.

The authors compare the WHO ENG work with equivalent UK material on which UK approaches had been based. They look at the studies that informed the exposure response functions currently recommended for the UK. A comparison is then made with the more recent studies that informed the ENG, followed by discussions of which aspects could have led to differences in the functions. They consider all transport sources, not just roadways, and much of their emphasis is on how the WHO work will influence the current UK methods for valuing health impacts. While there is some commentary on ERFs for annoyance (and sleep disturbance and cardiovascular effects), this is largely dealt with elsewhere in this report.

The primary reason to record this paper here is not so much the technical content – much is related specifically to UK approaches – but the nature of the thoughtful comparison of the WHO ENG reviews and analyses with equivalent UK material that the authors undertake. The senior author followed this work with a reanalysis of the annoyance ERF (see next section of this report) using the original WHO data supplemented with data from more recent studies.

## 2022 UPDATE OF THE ANNOYANCE SYSTEMATIC REVIEW AND ERF

Fenech et al. (2022) recently provided an update to the WHO 2018 Environmental Noise Guidelines systematic review of exposure-response relationships for annoyance from road traffic. The 2018 review had been based on studies published between 2000 and 2014. A subsequent scoping review commissioned by the UK's Interdepartmental Group on Costs and Benefits Noise Subject Group identified seven new studies published between 2014 and 2019 (see van Kamp et al., 2020) and a further search, for English language papers only, identified 8 studies published between 2019 and 2022. Half of the new publications reported studies carried out in Europe (n=8), while the remaining publications covered Asia (n=5), Africa (n=1), South America (n=1) and North America (n=1). Fenech et al. (2022) compiled the scatterplot of %HA with road traffic noise at different exposures of $L_{den}$ for the original WHO data (full dataset) and also for both sets of updates (Figure 16). As in the 2018 WHO annoyance synthesis, the data points shown in Figure 16 are not the original aggregate %HA from the studies, but the %HA *calculated* from the reported ERFs derived in each of the studies.

Fenech et al. (2022) report:

> *'For exposures below 55dB Lden, most of the datapoints for studies published post-2014 fall below the WHO aggregated curve. At higher exposure levels, the datapoints pre- and post- 2014 exhibit a similar spread of*

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Tolerance Level (CTL) approach (see comments on this CTL approach earlier in this report: Footnote 29). The authors of the Standard have added to the confusion in that they appear to have shown, in Figure 15, the regression line and its prediction limits from Miedema and Oudshoorn (2001) yet the data points for the figure from an application of CTL (Schomer et al., 2012).

[33] The 95% Prediction Interval means that one can be 95% confident that the %HA of the next group of people who are subject to a particular noise exposure will fall somewhere within the Prediction Interval shown for that exposure level. In a regression analysis, the Prediction Interval is wider than the Tolerance Interval - which is much wider than the Confidence Interval (see Tolerance Intervals/Limits and Confidence Intervals/Limits in Background Box 3, Figure 7 of this report).

*%HA. Therefore, our aggregate regression curve predicts a lower %HA below 55dB Lden, and slightly higher %HA above 65dB Lden, compared to the WHO ERR. This trend was also observed by the authors of the WHO systematic review when they excluded Alpine and Asian studies from their analysis …. The weighted and unweighted regression analysis gave very similar results, with some slight deviations at the highest exposures.'*

The authors reflect on appropriate inclusion criteria when selecting studies for a meta-analysis like this:

*"The publication of the WHO systematic review on annoyance sparked an academic debate…*

*on the specific methodology for conducting the ERR meta-analysis. One of the debated topics was the study inclusion criteria, including annoyance questionnaire wording, and whether respondents had to be representative of the general population or members of the general population. In our systematic review we noticed that very tight inclusion criteria would have resulted in only a handful of studies being screened in. We argue that every socio-acoustic survey will have its unique characteristics, and these differences contribute to the generalisability of an aggregate ERR…".*

In summary, the ERF for annoyance based on the updated data set conforms, by and large, to that reported to WHO (Guski et al., 2017), but is slightly lower over the 45 to 55 dB range. It again suggests that the Alpine studies included the WHO work may have 'distorted' (elevated) the ERF at these lower levels.

Figure 17 overlays these results with the Miedema and Oudshoorn (2001) quadratic ERF, its 95% Confidence Limits and 95% Tolerance Limits, again illustrating the most of the included study ERFs can be said to come from the same population of studies as that on which the Miedema and Oudshoorn (2001) ERF was based.

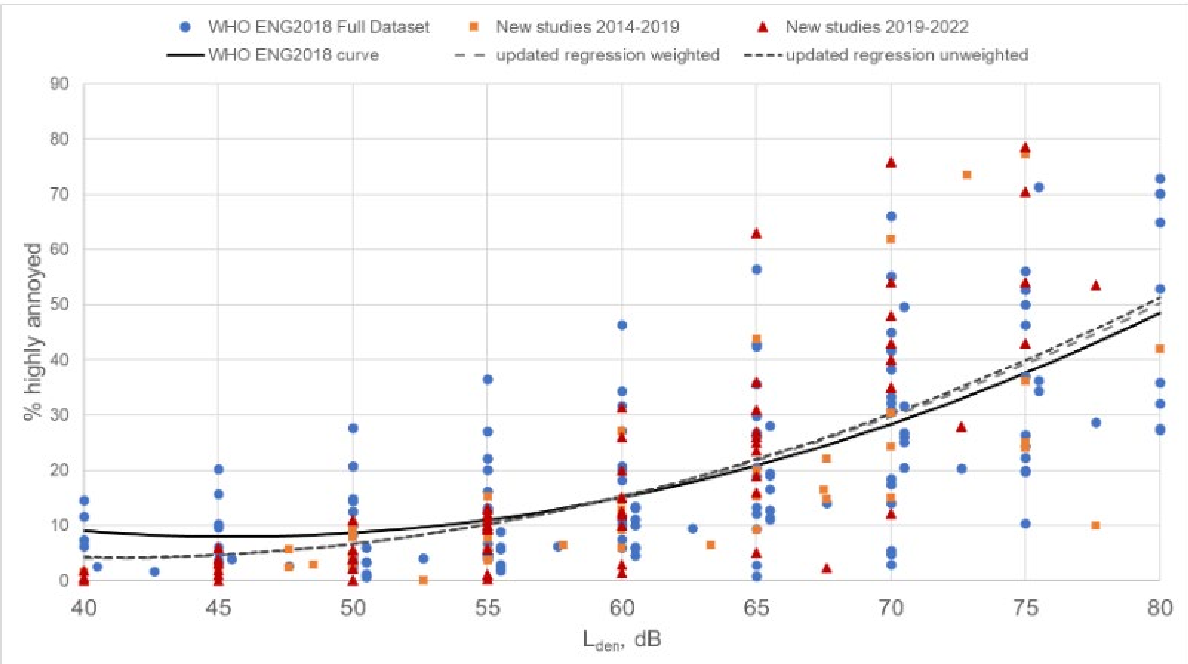


**Figure 16 The 2022 ERF update for annoyance from road traffic noise (Fenech et al., 2022). Scatterplot and quadratic regression of the relation between $L_{den}$ and the calculated %HA for road traffic noise. Blue dots refer to studies identified by the WHO systematic review. Orange squares and red triangles refer to studies identified from the (2014-19) and (2019-22) publication periods, respectively. The dashed lines are from the regression on all studies.**

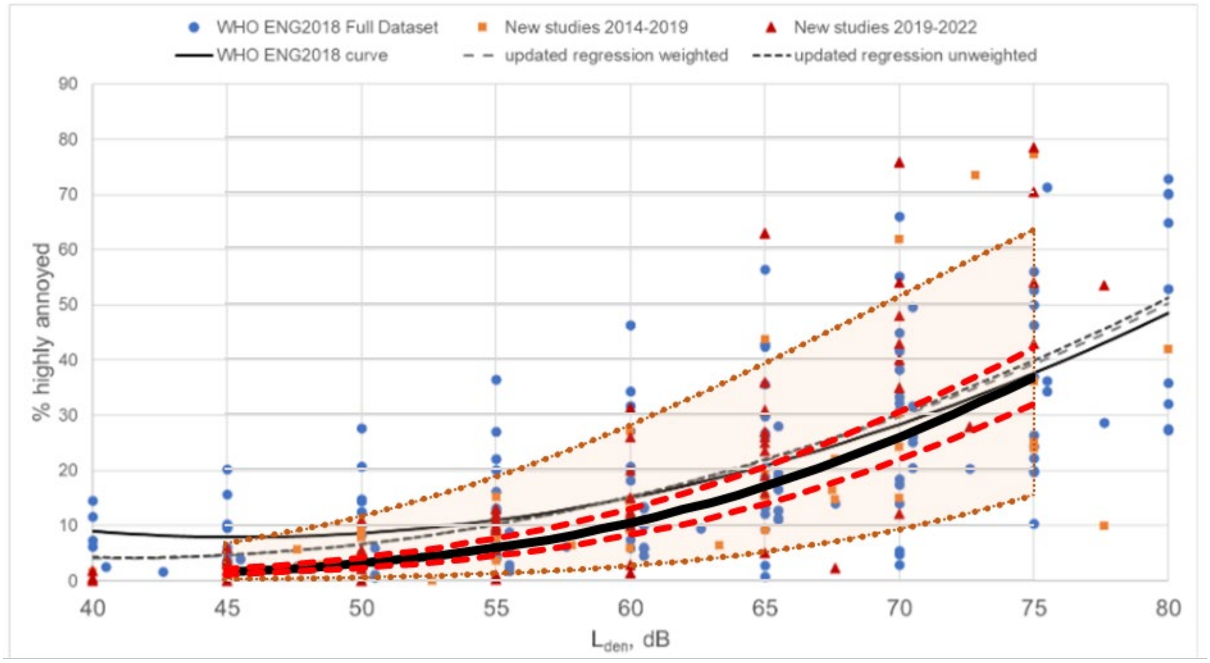


**Figure 17 This is the same plot as Figure 16 (*i.e.*, the data points from the full WHO data set together with data points from studies identified in the 2014-2019 review and the 2019-2022 review) fitted with quadratic regression equation to all data. As in Figure 10 above, Figure 15 has been overlaid with the Miedema and Oudshoorn (2001) quadratic ERF (heavy black line), its 95% Confidence Limits (broken red lines) and 95% Tolerance Limits (dotted orange lines and orange shaded area).**

## SLEEP

Basner and McGuire (2018) reported a systematic review and meta-analysis of the effect of environmental noise on sleep for the preparation of the WHO Environmental Noise Guidelines. That review has since been updated, a surprisingly short period between reviews, incorporating new studies completed in the interim. Eleven new studies (n=109,070 responses) across all transport modes were included in addition to the 25 studies (n=64,090 responses) from the original analysis for the WHO Environmental Noise Guidelines. The new paper is entitled, *Environmental Noise and Effects on Sleep: An Update to the WHO Systematic Review and Meta-Analysis*, and because it is an update of the original work, with overlap in authorship between the two reviews, it is appropriate here to refer primarily to the results of the update study rather than to the original. It is helpful that the authors compare the ERF of the updated analysis directly with the ERF from the original meta-analysis as used in the formulation of the WHO guidelines.

The introduction of the new review provides a succinct summary[34] of the effects of disturbed sleep on health[35]:

[34] References in the quote have been excised.

[35] Basner (2022), who is co-author of both systematic reviews, also provides a separate simplified summary (‘*Noise Effects on Sleep and Health Consequences*’) of the need for sleep, the nature of sleep disturbance by noise, habituation, long-term health effects of nocturnal noise exposure, and night-time noise policy and mitigation. He reports that studies have found evidence for impaired flow-mediated dilation of the brachial artery after a single night of (noise) exposure – and analyses of blood proteins demonstrated noise-induced changes indicative of a pro-thrombotic and pro-inflammatory phenotype and provide a molecular basis and biologic plausibility for the increased cardiovascular disease risks observed in epidemiological studies and other disease endpoints like neurodegenerative disease, obesity, diabetes, and breast and colon cancer (though the evidence on the effects of noise on the latter non-cardiovascular disease endpoints is only emerging).

This paper (Basner, 2022) could prove to be a potential source for future communication with the public and other professionals regarding an up-to-date scientific understanding of noise and sleep.

*Sleep is a vital component of human life that serves many critical roles in physical and mental health and well-being. Sufficient quantity and quality of sleep are requirements for optimal daytime alertness and performance, and high quality of life. Experimental studies suggest that restricted sleep duration causes blood vessel dysfunction, induces changes in glucose metabolism and appetite regulation, and impairs memory consolidation. Accordingly, epidemiological studies have consistently found that chronic short or interrupted sleep is associated with negative health outcomes, including obesity, diabetes, hypertension, cardiovascular disease, all-cause mortality, and poorer cognitive function. Chronic insufficient or disrupted sleep is therefore of public health relevance, and sleep disturbance a major adverse consequence of exposure to environmental noise…Night-time noise can fragment sleep structure by inducing awakenings and shifts to lighter, less restorative sleep. Importantly, these effects do not seem to habituate fully, and arousals and awakenings…can occur even among chronically exposed individuals. Although noise-induced sleep fragmentation and reductions in total sleep time are less severe than in sleep restriction studies, sleep disturbance by chronic noise exposure may lead to the development of disease in the long term. Experimental studies have found adverse effects of nocturnal aircraft noise on parameters of endothelial function, oxidative stress, and inflammation. This points to the importance of noise-induced sleep disturbance for cardiovascular disease risk, and, indeed, this is supported with epidemiological data where night-time noise is more strongly associated with indicators of vascular stiffness and hypertension compared with daytime noise.*

The reviews utilized field studies (no laboratory studies were included) in which the effects of sleep on adults were measured by self-reported questionnaire. Different sleep outcome variables had been measured in the questionnaires across the different studies used in the synthesis and these can be categorized as: 'awakenings from sleep' - defined as events where a participant wakes from sleep, regains consciousness, and recalls the awakening the following day; 'process of falling asleep' - defined as the transition from wakefulness to sleep; and 'sleep disturbance' - defined as the internal or external interference with sleep onset or sleep continuity.  A combined estimate derived from all relevant outcomes is also reported.

Self-reported sleep disturbance outcomes then form the basis of the 'highly sleep disturbed' (%HSD) exposure–response relationships. They are, therefore, critical outcomes from a noise policy perspective. Studies on acute noise-induced awakenings using objective measures, such as actigraphy or polysomnography, were not included in Smith et al. (2022), but were in Basner and McGuire (2018) - though the latter did not inform the final formulation of the WHO Environmental Noise Guidelines. Objectively measured outcomes are discussed separately below in the section on noise events because they still have potential relevance to the consideration of noise event metrics as supplementary noise indicators.

Reported below are the results of the meta-analysis for the road traffic noise studies:

a) the Odds Ratios (OR) for each of the individual studies (Figure 18) – which is the odds of being highly sleep disturbed by road noise per 10-dB increase in $L_{night}$. In both the first and second synthesis, there is a difference in results between studies where the sleep questions asked of respondents specifically included mention of road traffic noise as the source of the sleep disturbance, and where the question did not mention noise specifically as the source of disturbance (i.e., 'how much is your sleep disturbed by noise' as against 'how much is your sleep disturbed'). The different result is illustrated in Figure 18 where there are lower, but generally still significant, ORs, where the sleep disturbance question did not refer specifically to noise. and
b) the ERFs (Figure 19) for high sleep disturbance with $L_{night}$. The ERFs in Figure 19 report only those outcomes where noise was mentioned as the source.

There is considerable consistency between all the individual studies in Figure 18 with Odds Ratios greater than 1, and most with a 95% confidence interval not including 1 (there are two outliers with high ORs >12 and >19…that is, very high HSD). The summary (shown as 'Subtotal') for the studies where noise was mentioned as the source of disturbance has an OR of 2.32 (95% CI 1.90 to 2.84) – see black diamond in Figure 18. This can be interpreted as adults, across all new and original studies, who experience road traffic noise at home, being 2.32 times more likely to report being highly sleep disturbed when they are exposed to $L_{night}$ levels that are 10dB higher.

The coloured dots on the right of the Figure 18 relate to application of a rigorous assessment of the quality of studies included in the meta-analysis – they are examined in the original paper but are not discussed further here. Overall, using the GRADE assessment (see footnote 24), Smith et al. (2022) judged the quality of the evidence for the association between $L_{night}$ and sleep disturbance to be 'moderate'. Overall, the meta-analysis provides clear evidence that traffic noise exposure at home is associated with sleep disturbance. They also examined their results according to study-region and found that non-Europeans were less disturbed by road traffic noise than were Europeans, but the difference was not significant.

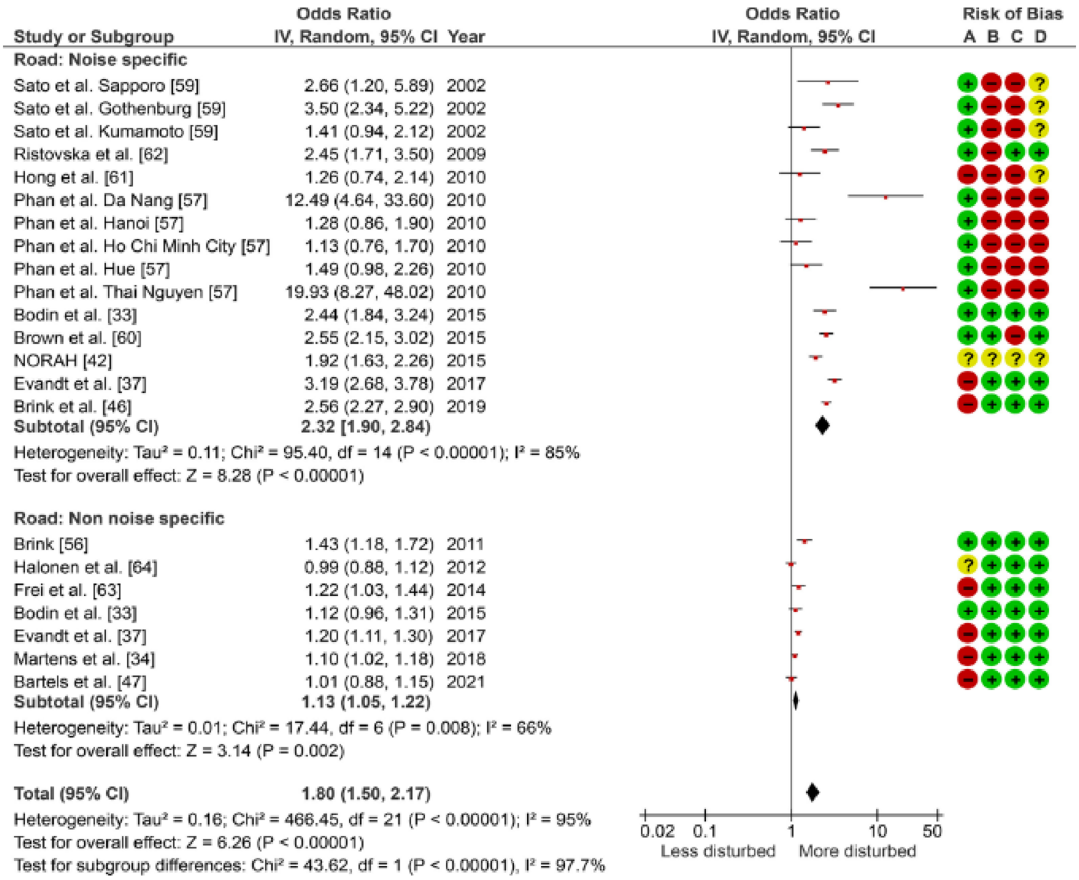


**Figure 18 Forest plot for the odds of being highly sleep disturbed by road noise per 10-dB increase in $L_{night}$ (combined estimate derived from all relevant outcomes within studies). Results are shown in two subgroups: those in which questions mentioned noise as the source of the disturbance, and those that did not specify noise as the source of the disturbance. Risk of bias: A: selection bias; B: exposure assessment; C: confounding; D: reporting bias. Green (+) denotes low risk of bias, red (–) denotes high risk of bias, yellow (?) denotes unclear risk of bias. Plots were generated using an inverse-variance random effects method across the full noise range for each individual study (not restricted to 40–65 dB $L_{night}$. (Extracted from Smith et al., 2022).**

The percentage of people highly sleep disturbed by road traffic noise ($L_{night}$) for each of the four sleep-disturbed outcomes, are shown in Figure 19, together with the 95% confidence Intervals for each ERF. The figures show the results of the original 2018 ERFs together with the updated analysis combining the original studies with the new studies. With increasing night-time noise levels, there were increased probabilities of reporting awakenings, having difficulties falling asleep, or having disrupted or disturbed sleep. When the sleep disturbance outcomes were combined, the resulting exposure–response curve was very similar to that calculated in the WHO review (bottom plot in Figure 19).

Figure 20 below compares the ERF of the newly identified studies only with the updated analysis. The curves are nearly identical below 50 dB $L_{night}$ but with slightly higher %HSD in the newer studies 55 to 65 dB. The similarity in the exposure–response curves between the updated analysis with the larger number of studies and the original analysis improves confidence in the earlier estimated ERF used in the formulation of the WHO guidelines.

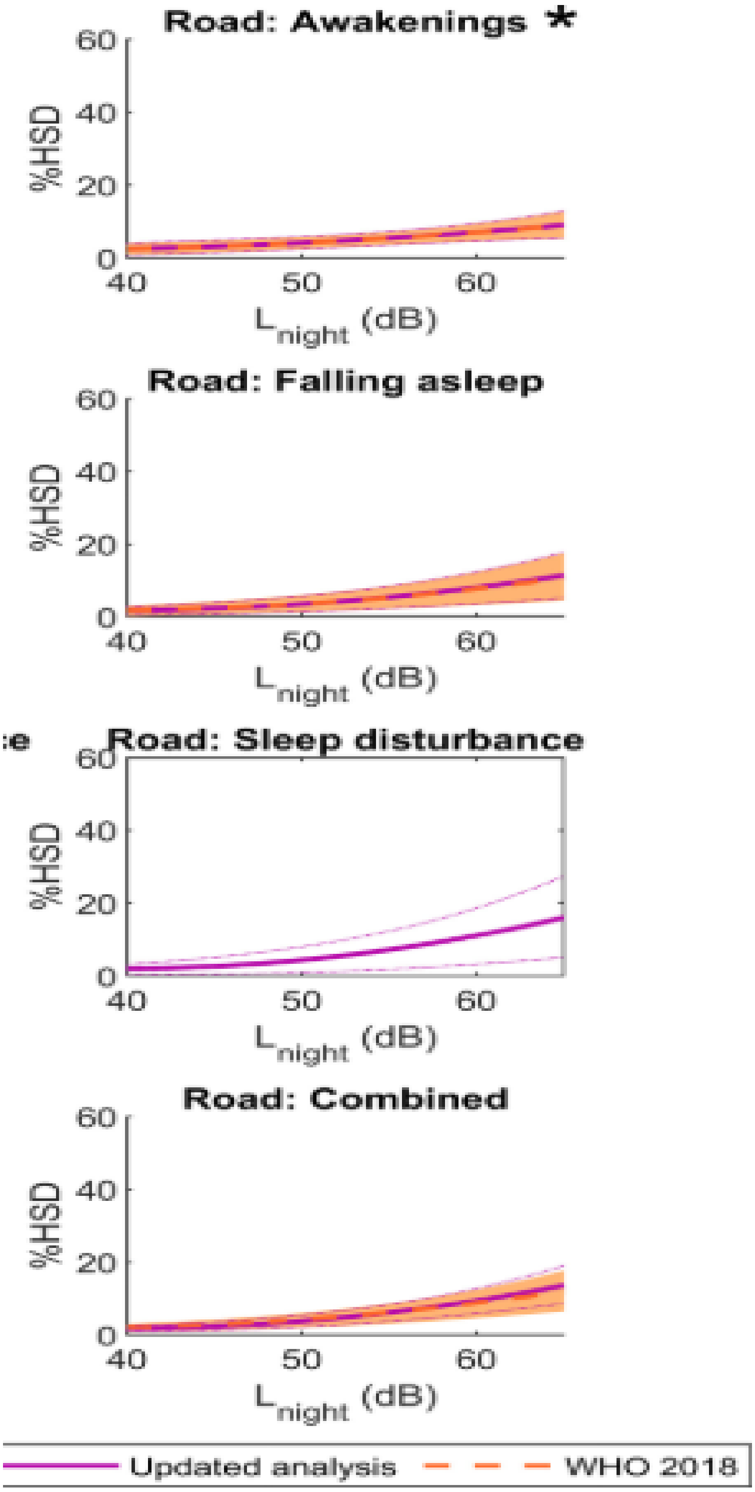


Figure 19 Probability of being highly sleep disturbed (%HSD) by night-time noise, determined via questions that mention noise as the source of disturbance, stratified by disturbance type. ERFs of the updated analysis (Smith et al., 2022) (solid purple lines with dotted 95% Confidence Intervals are compared against results of the 2018 WHO review (Basner & McGuire, 2018) (dashed orange lines with shaded 95% CIs). The ERF for the 'sleep disturbance' questions had not been calculated in 2018. Asterisk (*) indicate sleep outcomes for which no new studies have been published since the 2018 review. Parameter estimates were calculated in logistic regression models with $L_{night}$ included as the only fixed effect and study included as a random effect, restricted to the noise exposure range 40–65 dB $L_{night}$. The combined estimate was calculated using average responses of the awakening, falling asleep, and sleep disturbance questions (Extracted from Smith et al., 2022).

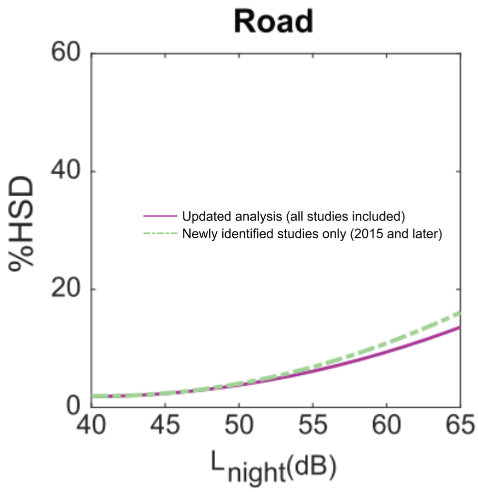


**Figure 20 Exposure-response relationships for the probability of being highly sleep disturbed (%HSD) by night-time noise. Curves are shown for the updated analysis that includes all available data (solid purple lines), and for analysis including only newly identified studies published after the WHO review (dashed green lines). Data are calculated as the combined response using average responses of the awakening, falling asleep, and sleep disturbance questions within studies, determined as the within-study average of disturbance questions that explicitly mentioned noise as the source of sleep disturbance. Parameter estimates were calculated in logistic regression models (Extracted from Smith et al., 2022).**

The authors note: *'A limitation of the meta-analysis was that many studies modelled noise exposure at the most exposed façade of the residence, and thus noise levels specifically at the bedroom façade are unknown. This means there is probably some exposure misclassification, with lower noise levels if the bedroom faces away from the noise source. ... This would, in effect, shift the exposure–response curves to the left, leading to an increased probability of disturbance at lower noise levels, given that noise levels at the bedrooms are, on average, probably lower than assuming they are all positioned at the most exposed façade. This was supported by two studies in the meta-analysis that found that a lower proportion of respondents were highly sleep disturbed by road traffic noise* (Brink, Schaffer, Vienneau, Foraster, et al., 2019) *or reported insomnia symptoms* (Evandt et al., 2017) *when the bedroom faced away from the street. Furthermore, disturbance was lower when the difference in noise level between the bedroom and the most exposed façade was greater* (Brink, Schaffer, Vienneau, Foraster, et al., 2019).

The authors (Smith et al., 2022) also add a cautious note to their meta-analysis of *self-reported* sleep disturbance…'*Our overall findings of self-reported disturbance by noise should be treated with some caution when considering noise-induced effects on sleep. Sleep is, by its nature, an unconscious process, meaning that its subjective evaluation is difficult. Accordingly, there can be substantial differences between self-reported and physiologically derived measures of sleep and noise-induced sleep disturbance. … self-reported outcomes capture habitual sleep quality and disturbance, unlike physiologic measurements that capture only acute effects within single nights. It does, however, remain unclear how long-term self-reported sleep disturbance by noise relates to overall health'.*

## EFFECT OF CONFOUNDERS ON AN ERF FOR SLEEP (SIRENE STUDY)

The meta-analysis described above (Smith et al., 2022) did not consider the effect of confounders on the noise exposure-sleep disturbance relationship as many of the included studies did not report the necessary data. However, one of the included studies, the Swiss SIRENE project (Brink, Schaffer, Vienneau, Pieren, et al., 2019) did examine the role of a range of potentially relevant effect modifiers on the ERF based on %HSD and $L_{night}$[36], the latter measured at the most exposed facade. These included: the "eventfulness" of noise exposure expressed as the Intermittency Ratio

[36] Data from the SiRENE study was included in the meta-analysis reported above (Smith et al., 2022) and contributed to the resulting OR analysis and ERF for %HSD with $L_{night}$. SiRENE is examined separately here only because of its consideration of the effect of several confounders of interest on sleep responses, but it can be noted in passing that the SiRENE ERF was significantly higher in terms of %HSD, particularly at the higher noise exposures, than was the synthesized ERF from the meta-analysis.

(IR) metric[37]; bedroom window opening/position; bedroom orientation towards the closest street; access to a quiet side of the dwelling[38]; degree of urbanization; sleep timing factors (bedtime and sleep duration); sleep medication intake; survey season and night air temperature.

The $L_{night}$-%HSD associations were not affected by bedroom window position, sleep timing factors, survey season, or outside temperature. But the effects of several variables on the %HSD ERF are shown in Figures 21 and 22. Unfortunately, the authors chose to: *'…show all exposure-response graphs, even if the respective effect modifiers were not statistically significant'*, leaving it to the reader to examine the coefficients of the models used for significance - provided in a Supplementary Material table. This author's interpretation of the complex tables is that only the effect modifiers IR and Bedroom Orientation in Figures 21 and 22 were significant (but not Window Position nor the level difference between bedroom window and most exposed façade (D quietside) – even though, when plotted in the figures below, the ERFs for the different values of the effects appear to be different for the different values of the modifier).

The authors had hypothesized that 'eventful' noise exposure situations with distinguishable noise events would more often trigger sleep disturbances than exposure to continuous noise. Figure 21 (left) does show this above about 60 dB(A), but earlier work shows (Brink, Schaffer, Vienneau, Foraster, et al., 2019) that higher IR was, counter-intuitively, associated with lower %HA.

Figure 21 (right) shows exposure-response relationships in the present study separately for those who slept with closed, half-open/tilted, or fully open bedroom windows though the statistics show that window position did not have a statistically significant effect on the ERF. In any case, the figure shows a counter-intuitive and inconsistent effect at the higher exposures – with those sleeping with closed windows having higher %HSD than half-open/tilted windows or open windows, at the higher exposure levels.

For Figure 22 (left), the authors note: *'…those with a considerable level difference at their bedroom window to the loudest façade point of the dwelling unit were, expressed in decibels, around 10 to 15 dB 'less sleep disturbed' than those with no level difference. All paired comparisons between the lowest difference category (< 5 dB) and the other difference categories were highly significant (post hoc), despite the difference category variable itself not being significant* (in the overall model)'.

As could be expected, bedroom orientation exhibited a strong moderating effect (Figure 22, right), with an $L_{eq}$-equivalent of nearly 20 dB if the bedroom faces away from the nearest street.

[37] Brink, Schaffer, Vienneau, Pieren, et al. (2019) note: '*An IR of higher than 50% means that more than half of the total sound energy is caused by "distinct" noise events (which are defined using a dynamic level threshold). Situations where events clearly emerge from background noise (e.g., at a receiver point near a railway track) yield IR values close to 100% whereas constantly flowing traffic situations with a steady noise level, e.g., from a distant motorway, only yield small IR values. In the present study, IR was determined on the $L_{Night}$-corresponding façade point in the time period between 23 and 07 h, henceforth referred to as IRNight*'. The Intermittency Ratio (IR) is considered later under the section on noise events.

[38] The authors estimated a level difference indicator D quietside to parameterize the difference in level (<5dB to>20dB) between outside the bedroom window and the most exposed façade of the building (but see the original paper for the exact definition).

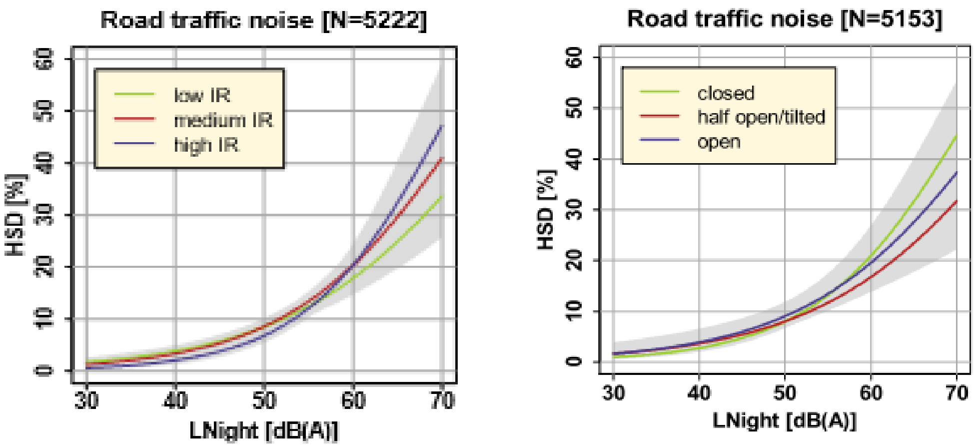


Figure 21 Left: Exposure-response relationships for road for three different IRNight categories, including 95% (overlapping) confidence intervals (shaded areas). Right: for different habitual bedroom window positions (closed, half-open/tilted, open) (extracted from Brink, Schaffer, Vienneau, Pieren, et al., 2019).

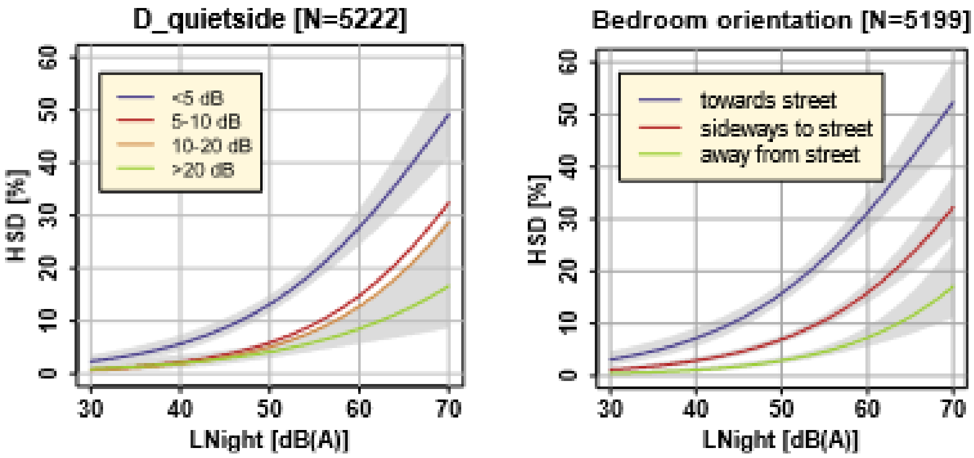


Figure 22 Left: Exposure-response relationships for road traffic noise for four groups of D quietside; 95% confidence intervals as shaded areas shown for the <5 dB and >20 dB category only. Right: Exposure-response relationships for road traffic noise for different orientations of the bedroom window, including 95% confidence intervals as shaded areas. $L_{Night}$ always refers to the maximum exposed façade point of the dwelling unit (extracted from Brink, Schaffer, Vienneau, Pieren, et al., 2019).

# CARDIOVASCULAR AND METABOLIC EFFECTS

Apart from the major systematic review and meta-analysis undertaken for the preparation of the WHO Environmental Noise Guidelines (Van Kempen et al., 2018), there has been a large number of studies of cardiovascular[39] and metabolic effects of noise in the review period. By way of example, Manohare et al. (2022) conducted a bibliometric analysis and review of auditory and non-auditory health impact due to road traffic noise exposure over the period 1992 to 2021 – this is summarized in Table 7 – and documents the extent and range of studies into health effects of road traffic noise.

**Table 7 Examples of different studies of the association between road traffic noise and cardiovascular issues. For references see the original publication (extracted from Manohare et al., 2022).**

| Study type | Reference | Inference |
|---|---|---|
| Long term noise exposure | [77] | Increased risk of diabetes due to excess cortisol and sleep disturbance, with an improved incidence rate range of confirmed diabetes 1,14 (1,06-1,22) to 10 dB in $L_{den}$. |
| | [78] | Per 10 dB increase in $L_{den}$ can cause myocardial infarction. |
| | [79] | A strong association between arrhythmia atrial fibrillation and noise annoyance considering ECG based diagnosis and physician-diagnosed history. |
| | [80] | Noise levels (Leq) during 6-22 hrs. above 65 dBA is correlated with increased risk of CVD. |
| | [81] | Annoyance due to traffic noise serves as a regulator of the relationship between the noise level and hypertension. |
| | [82] | Risk of hypertension among reproductive aged women due to road traffic noise exposure |
| | [83] | Impact of long-term noise exposure on BP and hypertension is not convincingly reported. |
| | [84, 85] | Non-significant risk of cardiovascular issues. |
| | [86] | Rate of risk is low for noise level below 60 dB, but increases for noise levels above 60 dB considerably, complimenting the suggested dose-response relationship. |
| Short term exposure | [87] | The correlation between rate of noise exposure and cardiovascular mortality indicates a combined impact of levels of diurnal and night-time noise. |
| Cross-sectional studies | [88–91] | Positive association observed between RTN and blood pressure change among children and pregnant women. |
| laboratory study | [92] | Increases in blood pressure and hemodynamic factors associated with RTN. |
| | [93] | No significant association with change in blood pressure due to RTN exposure |
| | [94] | Increased heart rate was observed due to noise exposure |
| | [95] | Risk of respiratory illness among children by the effect of emotional stress-induced through the noisy neighbourhood as compared to air pollution |
| Meta-analysis | [96] | Traffic noise and hypertension, and cardiovascular diseases positive association |
| | [97] | A meta-analysis done on literature (hypertension and traffic noise) for the year 2011-17 has reported risk but was on the lower side as compared to a meta-analysis done previously |
| | [98] | Contribute to evidence on traffic noise as a risk factor for cardiovascular disease |
| | [99] | Updated exposure-response relationship for RTN and coronary heart disease (CHD) |
| | [86] | Dose-response relationship developed for traffic noise and myocardial infarction and are widely recognized. |
| | [100] | Dose-response relationship developed for aircraft noise and hypertension |
| | [96] | Dose-response relationship developed for traffic noise and hypertension |
| | [101] | Direct association between RTN, annoyance, and arterial hypertension, with risk of ischemic heart diseases. |

Another indication of the amount of new work is the overview of the state of the science of non-auditory effects of noise over the 2017-2020 presented at the 2021 ICBEN conference (Persson Waye & van Kempen, 2022). They

[39] Below is a lay summary of some of the different cardiovascular terms referred to by different authors:

**Cardiovascular disease (CVD):** The umbrella term for **all** types of diseases that affect the heart or blood vessels.

**CVD deaths** are ALL deaths caused by CVDs.

**Myocardial infarction (MI) deaths (= heart attack)** are often the result of a prolonged and untreated **myocardial ischemia.**

**Ischemia** (or ischaemia) is a restriction in blood supply to any tissue, muscle group, or organ of the body.

**Myocardial ischemia**. is characterized by a decrease in blood supply to the heart tissue which leads to tissue damage, a result of which is chest pain. (**Angina pectoris** refers to chest pain caused by decreased blood supply to the heart muscle, but **without** tissue damage).

**Ischaemic heart disease (IHD)**) is the term for confirmed myocardial ischemia (tissue damage to the heart).

**Coronary artery disease (CAD)** (also **coronary heart disease**, or just **heart disease**) is the commonest cause of myocardial ischemia. Build-up of plaque is clogged arteries or **atherosclerosis** (hence **atherosclerotic cardiovascular disease (ASCVD)**). The plaque reduces the amount of oxygen-rich blood getting to heart, which can cause chest pain (angina). Plaque can also lead to blood clots, which block blood flow and are the **most common cause of a heart attack**. ASCVD in the blood vessels of the brain can decrease blood flow to the brain and result in **ischemic stroke.**

identified some 19 different systematic reviews (see Table 8 below) on the impact of environmental noise on the cardiovascular and metabolic system alone, with road traffic noise as one of the sources in most of the reviews.

**Table 8 Overview of identified systematic reviews on the impact of environmental noise on the cardiovascular and metabolic system 2017-2020 (extracted from Persson Waye & van Kempen, 2022).**

| First author and reference | Studies included | | | | | | | | Meta-analysis included |
|---|---|---|---|---|---|---|---|---|---|
| | No evaluated | No of participants | Time range | Countries* | Population$ | Noise source(s)† | Setting(s)** | End point(s)‡ | |
| Weihofen, 2019[25] | 9 | 780 – 5,523,788 | 1947 – Aug 2017 | 1-10 | GP | A | R | D | Yes |
| Wang, 2020[19] | 8 | 1836 – 380738 | 2009 – Oct 2019 | 3, 5, 8, 15, 20, 34 | GP | A,B, E | R, O | E | Yes |
| Fu, 2017[18] | 32 | 59 – 145,190 | To Dec 2016 | 2, 6, 8, 10, 12-21 | A | A, B, E | R, O | B | Yes |
| Sakhvidi, 2018[27] | 15 | 40 – 381,000 | To Sept 2017 | 1, 3, 5, 6, 8, 12, 15, 20, 22- 24 | GP | A, B, E, F | R, O | E | Yes |
| Peters, 2018[22] | 17 | 60 – 4,4 million | 2013 - 2017 | 1, 2, 4-6, 8-10 | A | A | R | A, B, C, D, E, G | No |
| Hadad, 2019[20] | 9 | 420 - ~4.4 million | 2007 - 2018 | 4-6, 8, 9, 20, 25 | A | A, B, C | R | B, C, D, G | No |
| Dzhambov, 2017[16] | 13 | 115 – 1542 | To July 2016 | 1, 4, 6, 7, 14, 17, 21, 26, 27 | C | B | R,E | A | Yes |
| An, 2018[13] | 11 | 132 – 52456 | To Feb 2018 | 1, 8, 10, 20, 23, 27, 28 | GP | A, B,C, E, G | R, O, E | H, I | Yes |
| Munzel, 2017[29] | ~19 | Up to 8.6 million | NR | 1-10, 20, 24 | A | A, B, C | R | A, C, D, E, G, H, I | No |
| Freiberg, 2019[28] | 2* | 725 – 1238 | 2000 – Sept 2017 | 3, 7 | GP | D | R | B, C, E | No |
| Dzhambov, 2018[17] | 9 | 420 – 4,415,206 | To August 2017 | 3-6, 8, 9, 20, 25, 33 | A | B | R | B | Yes |
| Van Kempen, 2018 [5] | 61 | 85 – 6,027,363 | 2000 – Aug 2015 | 1-8, 10, 11, 13, 14, 16, 18, 20, 22, 25, 26, 33 | A, C | A, B, C, D | R, E | A, B, C, D, E, H, I | Yes |
| Wilding, 2019 [26] | 8 (1) | 54,968 | 1990-Aug 2018 | 20 | C | B, C | R | I, K, N | No |
| Alves, 2020 [12] | 6* | 4 – 717453 | 2016-2019 | 1, 8, 10, 16, 20 | A | B, D, E | R, O | C, D, J | No |
| Van Kamp, 2020 [23] | 18* | 420 – 4,400,000 | 2015-2019 | 2-6, 8-10, 13, 20, 24, 25, 33 | A, C | A, B, C, D | R | A, B, C, D, E, H, I, K | No |
| Khosravipour, 2020 [21] | 13 | 243-854366 | Up to Nov 2019 | 4, 6, 8, 18, 20 | A | B | R | C | Yes |
| Cai, 2021 [14] | 13 | 6,304 – 4,600,000 | 2000-2020 | 2-5, 7, 8, 20, 25 | A | A, B, C | R | C, D | Yes |
| Rugel, 2020 [11] | 29[a] | 513 – 4,284,680 | 2003 – 2019 | 3-10, 16, 20, 23, 25, 33 | A | B | R | B, C, D, E, G, M, O, P | No |
| Dendup 2018 [15] | 4[a] | 513 – 53673 | April 2017 | 5, 8, 20, 23 | A | A, B, C | R | E | No |

*Countries: 1 = USA, 2 =France, 3 = Canada, 4 =United Kingdom, 5 = Switzerland, 6 = Germany, 7 = The Netherlands, 8 = Sweden, 9 = Greece, 10 = Italy, 11 = Croatia, 12 = Brazil, 13 = Japan, 14 = Serbia, 15 = Korea, 16 = Taiwan, 17 = Pakistan, 18 = Lithuania, 19 = Iran, 20 = Denmark, 21 = India, 22 = EU, 23 = Bulgaria, 24 = China, 25 = Spain, 26 = Austria, 27 = Slovakia, 28 = Portugal, 29 = Belgium, 30 = Australia, 31 = New Zealand, 32 = Poland, 33 = Norway, 34 = European Union;

† A = Air traffic, B = Road traffic, C = Rail traffic, D = Wind turbines, E = Occupational, F = Recreational, G = Human, ;

‡ A = Blood pressure, B = Hypertension, C = Coronary heart disease (incl myocardial infarction and angina pectoris), D = Stroke/cerebrovascular disease, E = Diabetes, F = Cardiometabolic, G = Heart failure, H = Change in waist circumference, I = Change in BMI, J = Cardiovascular risk factors, K = Obesity, L = Cholesterol, M = Arterial Fibrillation, N = Overweight, O = cardiovascular disease, P = Cardiocirculatroy pathologies; [a] In this review also the impact on other effects was evaluated. These studies were not taken into account for this review. Only the studies dealing with cardiovascular and metabolic effects were considered;

$ Population under investigation: GP =General population, A = Adults, C = Children; ** Setting(s): R = Residence, O = Occupational, E = Educationa

From just these two examples it is not feasible, within this current scoping study, to report and discuss the large body of work on cardiovascular and other health effects of transport noise that has been reported in the review period. Nor would that necessarily be useful, given that the field of cardiovascular and metabolic effects of noise is recognized as a relatively new and challenging domain of research. For a non-medical, non-epidemiological audience, the number of different health outcomes considered, and the various statistical measures of association applied in epidemiological and medical studies, can be somewhat bewildering.

Fortunately, the availability of one particular review paper[40] (Münzel et al., 2021), entitled '*Transportation noise pollution and cardiovascular disease'*, helps overcome this problem by providing a reasonably succinct update until late 2021. Its point of departure is the findings of the WHO Environmental Noise Guidelines review (Van Kempen et al., 2018)[41], in '*WHO Environmental Noise Guidelines for the European Region: A Systematic Review on Environmental Noise and Cardiovascular and Metabolic Effects: A Summary', which* had reported that, of the 61 studies they identified as useable between January 2000 and August 2015 (also see Figure 23 below as reported in van Kamp (2019)):

> *'A majority of the studies concerned traffic noise and hypertension, but most were cross-sectional and suffering from a high risk of bias. The most comprehensive evidence was available for road traffic noise and Ischeamic Heart Diseases (IHD). Combining the results of 7 longitudinal studies revealed a Relative Risk (RR) of 1.08 (95% CI: 1.01–1.15) per 10 dB (LDEN) for the association between road traffic noise and the incidence of IHD. We rated the quality of this evidence as high. Only a few studies reported on the association between transportation noise and stroke, diabetes, and/or obesity. The quality of evidence for these associations was rated from moderate to very low…'.*

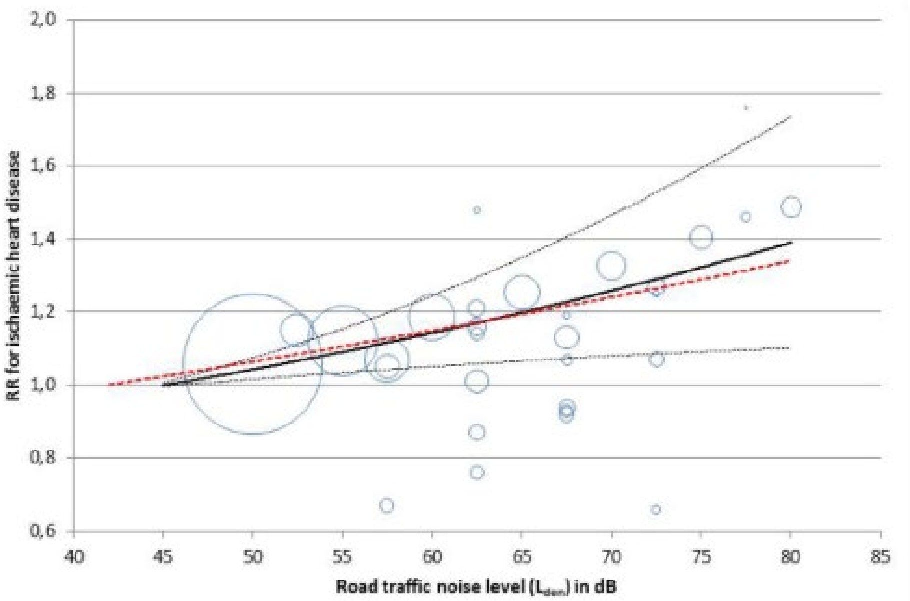


**Figure 23  Pooled exposure-response association between road traffic noise exposure ($L_{den}$) and Relative Risk (RR) established for the WHO evidence review based on seven longitudinal studies. Red dashed line: linear relationship derived (RR = 1.08 per 10 dB) and validated against a restricted cubic spline model (black solid line, with 95% confidence interval represented by black dashed lines). The open circles indicate the relative risk of IHD in individual studies, at different levels of exposure; their size is proportional to the precision (inverse of variance). For further details, refer (Van Kempen et al., 2017).**

The ERF indicates that the association with cardiovascular effects starts in the vicinity of 45-55dB $L_{den}$ at the most exposed façade of a dwelling for road traffic noise.

[40] The high standing of this journal: '*Nature Reviews Cardiology'* should be noted, as evidenced by its Impact Factor of 49.42. It is also fortuitous, for current purposes, that this journal aims to publish articles which are not only authoritative, but also accessible with clearly understandable figures, tables, etc.

[41] The complete review of 300 pages (Van Kempen et al., 2017) is published on the website of RIVM (the Dutch National Institute for Public Health and the Environment) , accessibile via the following link:  https://www.rivm.nl/bibliotheek/rapporten/2017-0078.pdf

The Münzel et al. (2021) review firstly provides a timeline of research on health effects of noise (Figure 24). This helps to understand why there was little focus on health effects before the 1990s – the first cohort study on traffic noise exposure and ischemic heart disease was not until 1988.

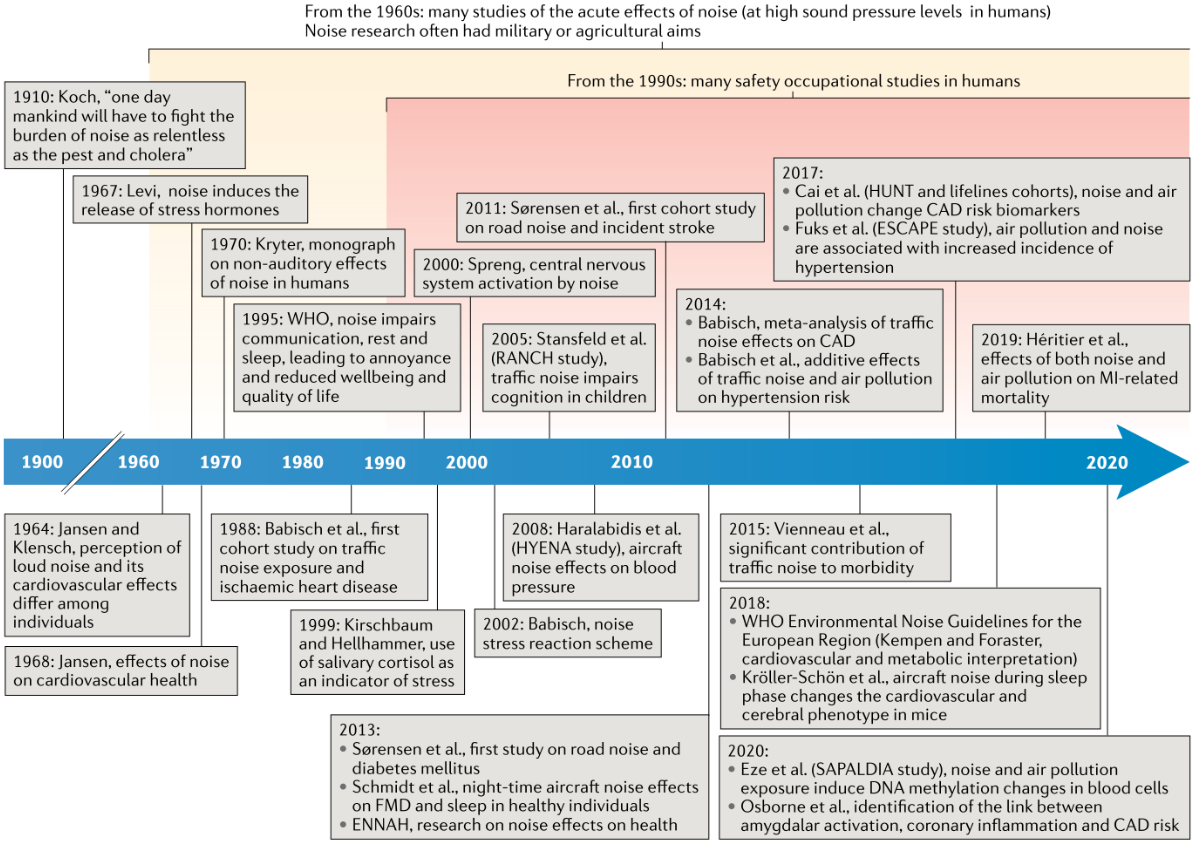


Figure 24 Historical overview of research on the adverse health effects of noise, highlighting important concepts and reports, with focus on transportation noise. The selection of studies and consortia is based on the authors' (Münzel et al., 2021) personal views of the noise research field. (CAD, coronary artery disease; ENNAH, European Network on Noise and Health; FMD, flow-mediated dilatation; MI, myocardial infarction).

Münzel et al. (2021) then summarized important WHO review results (Van Kempen et al., 2018) for road traffic noise - Figure 25. A Relative Risk (RR) of one implies there is no difference in the health effect with increased exposure. If the RR is greater than 1, then the event is more likely to have occurred in association with high noise exposure. For example, the RR of 1.08 for the incidence of coronary artery disease means there is a 1.08 times greater incidence of the disease per 10dB increase in noise exposure, starting at 53dB(A)[42]. If the 95% Confidence Interval for the RR does not include the value 1 (*viz.* incidence of stroke and incidence of coronary artery disease in Figure 25) then the finding is statistically significant. The '*high quality*'[43] level of evidence for the incidence of coronary artery disease should be note, providing evidence of an effect of road traffic noise on coronary artery disease.

[42] The phrase '*starting at 53 dB(A)*' is in the text of Münzel et al. (2021 page 620)

[43] For interpretation of the GRADE '*quality of evidence*' scores on Figure 25, see Footnote 24.

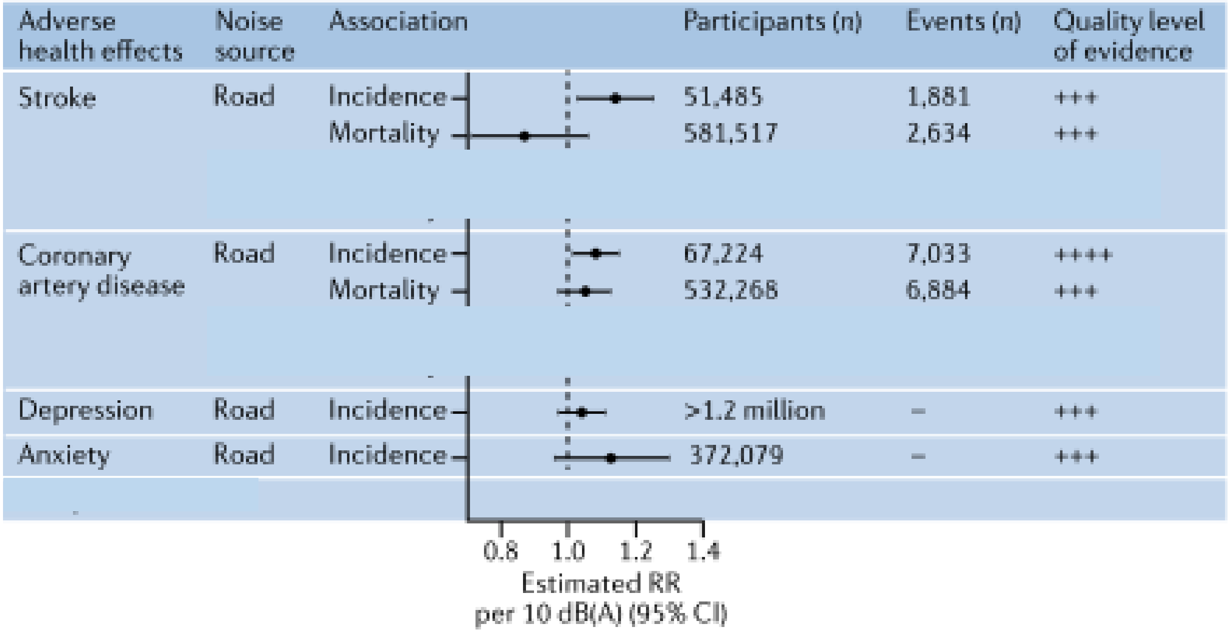


**Figure 25 The incidence of stroke, coronary artery disease, depression and anxiety increases in response to chronic exposure to road traffic noise (expressed as relative risk (RR) estimates for every 10 dB(A) increase in exposure). Extracted from Münzel et al. (2021) who used data summarized in the 2018 WHO Environmental Noise Guidelines (Van Kempen et al., 2018) for stroke and coronary artery disease; by (Dzhambov & Lercher, 2019) for psychological disease. Quality of evidence shown as: +++ moderate ++++ high.**

Münzel et al. (2021) then summarize the large number of cardio-metabolic road traffic noise studies that have become available since the WHO review, many of which have been large and well-designed. Their summary is that more longitudinal studies are still needed to clarify to what extent transportation noise affects the risk of stroke. Heart failure had not been examined in the WHO study, but new studies consistently report an association between road (and other transport modes) traffic noise and heart failure incidence and mortality, with a 2-8% increase of risk per 10 dB(A) increase in exposure.

As well as reporting epidemiological studies of cardiovascular disease (incidence and mortality) Münzel et al. (2021) provided an extensive update of the evidence of the association of road traffic to seven ***cardiovascular risk factors. These*** factors help explain the pathways for noise effects on cardiovascular health: *viz.* sleep disturbance; arterial hypertension; diabetes; obesity; physical activity; smoking and alcohol consumption; and depression. The results are reported in Table 9, for each factor, showing: the WHO result; the new quality studies; and the updated conclusions. The quality of evidence has not increased for the sleep disturbance and hypertension risk factors, but it has for the diabetes and obesity risk factors. There was only low, very low and low-moderate evidence for an association with the risk factors depression, smoking and alcohol consumption and physical activity, respectively.

**Table 9 Epidemiological studies on road traffic noise and cardiovascular risk factors (Source: extracted from table in Münzel et al., 2021). All estimates given per 10 dB(A) increase in noise exposure.**

| Conclusions in the 2018 WHO guidelines[2,21,39,64a] | Prospective case–control and cohort studies not included in the 2018 WHO guidelines[b] | | Updated conclusions |
|---|---|---|---|
| | Design | Results | |
| *Sleep disturbance* | | | |
| Moderate quality of evidence; percentage of highly sleep-disturbed persons (self-reported) for $L_{night}$ OR 2.13 (95% CI 1.82–2.48) for questionnaires referring to noise and OR 1.09 (95% CI 0.94–1.27) for questionnaires not mentioning noise; probability of awakening for indoor $L_{max}$ (polysomnographic studies) OR 1.36 (95% CI 1.19–1.55) | Cohort study, Denmark, *n* = 44,438 | HR for redemption of sleep medication 1.02 (95% CI 0.98–1.07) for 45–50 dB(A), 1.01 (95% CI 0.99–1.03) for 50–55 dB(A) and 1.05 (95% CI 1.00–1.10) for >55 dB(A) per 10 dB(A) increase in 10-year mean $L_{night}$ (reference <45 dB(A))[40] | The quality of evidence has increased only slightly; more studies using objective indicators or standardized subjective indicators of sleep and a longitudinal approach are needed to increase the quality of evidence |
| | Cohort study, Germany, *n* = 9,354 | RR 1.19 (95% CI 1.10–1.28) for self-reported sleep disturbance at follow-up for people reporting to be extremely annoyed by road traffic noise at night at baseline[41] | |
| | Review of cross-sectional studies | For road traffic noise and sleep "no large differences are expected" compared with the evaluation in the WHO guidelines[25] | |
| *Hypertension* | | | |
| Low-quality evidence for RR incidence (0.97, 95% CI 0.90–1.05); only one cohort study; very low-quality evidence for RR prevalence (1.05, 95% CI 1.02–1.08) | Cohort study, Greece, *n* = 71 | OR 1.18 (95% CI 0.92–1.52) per 10 dB(A) $L_{Aeq}$[36] | The quality of evidence has increased, with six new prospective studies, which found either weak or no association with hypertension; however, large between-study variation exists in the definition of hypertension and more studies are needed to examine whether road traffic noise leads to hypertension |
| | Cohort study, Sweden, *n* = 1,386 | HR 0.93 (95% CI 0.86–1.01)[45] | |
| | Case–control study, Germany, *n* = 137,577 | OR 1.00 (95% CI 0.99–1.01)[44] | |
| | Cohort study, UK, *n* = 17,785 | HR 1.01 (95% CI 0.94–1.08) for 55–60 dB(A) and 0.99 (95% CI 0.88–1.05) for >60 dB(A) $L_{night}$ (reference <55 dB(A))[35] | |
| | Cohort study, pooled analyses from 4–6 European cohorts, *n* = 6,207 for self-reported and *n* = 3,549 for 'measured' incident | RR 1.03 (0.99–1.07) for self-reported hypertension, RR 0.99 (0.94–1.04) for measured hypertension[42] | |
| | Cohort study, Denmark, *n* = 21,241 | HR 1.00 (95% CI 0.98–1.02)[43] | |
| *Diabetes mellitus* | | | |
| Moderate-quality evidence<br>RR for incidence (1.08, 95% CI 1.02–1.14); only one cohort study | Cohort study, Canada, *n* = 12,941 | OR 1.11 (95% CI 1.06–1.15)[49] | The quality of evidence has increased, with five new prospective studies, which fairly consistently found that road traffic noise increased the risk of diabetes |
| | Cohort study, Greece, *n* = 30 | OR 1.18 (95% CI 0.85–1.65)[36] | |
| | Cohort study, Switzerland, *n* = 110 | RR 1.38 (95% CI 1.04–1.84)[50] | |
| | Cohort study, Germany, *n* = 330 | RR 1.11 (95% CI 0.97–1.27)[51] | |
| | Cohort study, Denmark, *n* = 1,158 | HR 1.03 (95% CI 0.87–1.22) for 48–58 dB(A) and 1.08 (95% CI 0.89–1.31) for >58 dB(A) (reference <48 dB(A))[52] | |
| | A meta-analysis including five of the six current studies on diabetes incidence | RR of 1.11 (95% CI 1.08–1.15) for road traffic noise[53] | |
| *Obesity* | | | |
| Very low-quality evidence; change in BMI 0.03 kg/m$^2$ (–0.10 to 1.15 kg/m$^2$), change in waist circumference 0.17 cm (–0.06 to 0.40 cm); three cross-sectional studies | Cohort study, Denmark, *n* = 39,720 | Yearly weight gain 15.4 g (2.1–28.7 g), yearly increase in waist circumference 0.22 mm (0.02–0.43 mm), RR for gaining >5 kg during follow-up 1.10 (95% CI 1.04–1.15)[56] | The quality of evidence has increased considerably since the WHO guidelines (which relied only on cross-sectional studies), with four new prospective studies, which fairly consistently found that road traffic noise was associated with obesity, most consistently with an increase in waist circumference (central obesity) |
| | Cohort study, Sweden, *n* = 5,712 | Yearly weight gain 10 g (–9 to 30 g), yearly increase in waist circumference 0.4 mm (0.2–0.6 mm)[54] | |
| | Cohort study, Switzerland, *n* = 3,796 | Change in BMI –0.04 kg/m$^2$ (–0.13 to 0.06 kg/m$^2$); RR for developing obesity 1.25 (95% CI 1.04–1.51)[55] | |
| | Cohort study, Denmark, *n* = 52,661 pregnant women | Postpartum weight retention (18 months after pregnancy) 90 g (2–160 g)[57] | |

## FURTHER EVIDENCE OF CARDIOVASCULAR EFFECTS PUBLISHED IN 2022

There are two recent papers of interest.

Firstly, there is a 15 years follow-up in a nationwide prospective cohort study in Switzerland (Vienneau et al., 2022) with 4.1 million adults. That study reported (paraphrased):

> *Models included all three noise sources plus $PM_{2.5}$, adjusted for individual and spatial covariates…. Absolute excess risk was calculated by multiplying deaths/100,000 person years by the excess risk (hazard ratio-1)[44] within each age/sex group. Results: During a 15-year follow-up, there were 277,506 CVD and 34,200 myocardial infarction (MI) deaths. Associations (hazard ratio; 95%-CIs)* ***for road traffic****, railway and aircraft noise and CVD mortality were* ***1.029 (1.024–1.034)****, 1.013 (1.010–1.017), and 1.003 (0.996–1.010) per 10 dB Lden, respectively. Associations for MI mortality were a respective* ***1.043 (1.029–1.058),*** *1.020 (1.010–1.030) and 1.040 (1.020–1.060)* ***per 10 dB Lden****.* ***Blood pressure-related, ischemic heart disease, and all stroke mortality were significantly associated with road traffic*** *and railway noise, while ischemic stroke mortality was associated with aircraft noise.* ***Associations were mostly linear, often starting below 40 dB $L_{den}$ for road traffic*** *and railway noise… While the absolute number of deaths attributed to noise increased with age, the hazard ratios declined with age. Relative and absolute risk was higher in males compared to females****.*** *Figures 26 and 27 show the ERRs between road traffic noise and and cause-specific cardiovascular mortality relationships in this study.*

Figure 26 shows a largely linear relationship between both All CVD mortality and Myocardial infarction mortality and $L_{den}$, with associations starting as low as 35 dB. This was also similar for BP-related mortality (Figure 27). The association for IHD mortality and the increased risk from road traffic did not begin until noise levels were around the WHO guideline limit of 52 dB $L_{den}$. Similarly, the slightly increased risk for road traffic and Heart Failure (HF) mortality was only in the higher exposure range above 50 dB

The overall conclusion of this study (Vienneau et al., 2022) was that:

> *'Transportation noise exposure was associated with CVD mortality in Switzerland. Independent of air pollution, road traffic and railway noise exposure were associated with the majority of CVD causes of death, often with risk increases starting well below the WHO guideline limits.'*

The above is an important finding because there has been a tendency, over many years, to suggest that road traffic noise is a co-exposure with air pollution and that many of the health effects are due to air pollution, not noise.[45]

Secondly, Thacher et al. (2022) reported '*Exposure to transportation noise and risk for cardiovascular disease in a nationwide cohort study from Denmark'. This aimed* to study whether exposure to road, railway, and aircraft noise increased risk for ischemic heart disease (IHD), IHD subtypes, and heart failure in the entire adult Danish population. They modelled road (and railway, and aircraft) noise at the most and least exposed façades for the period 1995–2017 for all addresses in Denmark (n = 2,769,944 individuals) and calculated 10-year time-weighted running means for 2.5 million individuals age ≥50 years, of whom 122,523 developed IHD and 79,358 developed heart failure during follow up (2005–2017). Analyses were adjusted for individual and area level sociodemographic covariates and air pollution. Results: We found road traffic noise at the most exposed façade ($L_{den}$) to be associated with higher risk of IHD, myocardial infarction (MI), angina pectoris, and heart failure, with hazard ratios (HRs) (95% confidence intervals (CI)) of 1.052 (1.044–1.059), 1.041 (1.032–1.051), 1.095 (1.071–1.119), and 1.039 (1.033–1.045) per 10 dB higher 10-year mean exposure, respectively. These associations followed a near-linear exposure-response relationship and were robust to adjustment for air pollution with $PM_{2.5}$.

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[44] The Hazard Ratio (HR) = risk of outcome in exposed group/risk of outcome in non-exposed group, occurring at a given interval of time. It is interpreted in the same way as an Odds Ratio (*i.e.,* If HR = 1 then there is no association between the variables).

[45] In general, epidemiologists have only recently extended their focus on transport pollutants to include noise – with much previous focus on transport pollutants and health being restricted to air pollution.

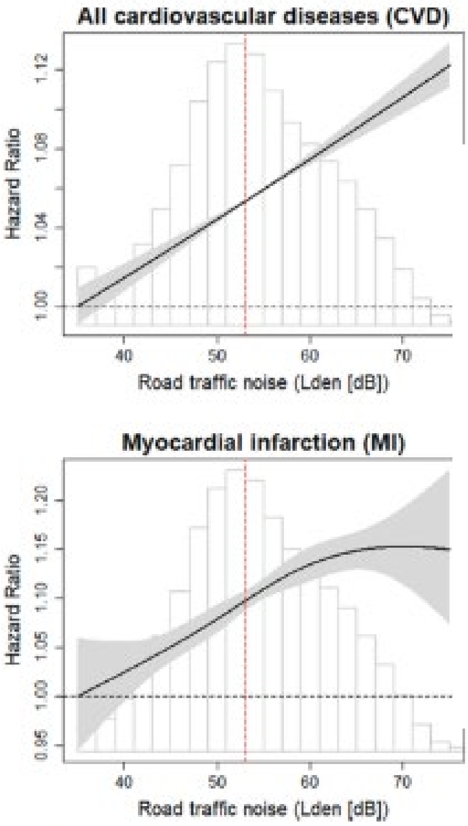



**Figure 26 Natural splines (3 df) for the association between road traffic noise and All CVD and MI mortality. Multipollutant models, adjusting for the other two noise sources (rail and aircraft noise exposures). Model 3: Included strata, sex, and period, and adjusted for mother tongue, nationality, civil status, education, local-SEP, area-SEP and unemployment, and $PM_{2.5}$. Vertical red lines show WHO guideline levels based on $L_{den}$: road traffic = 53 dB. (Extracted from Vienneau et al., 2022).**

**[SEP = socio-economic position, in quartiles]**

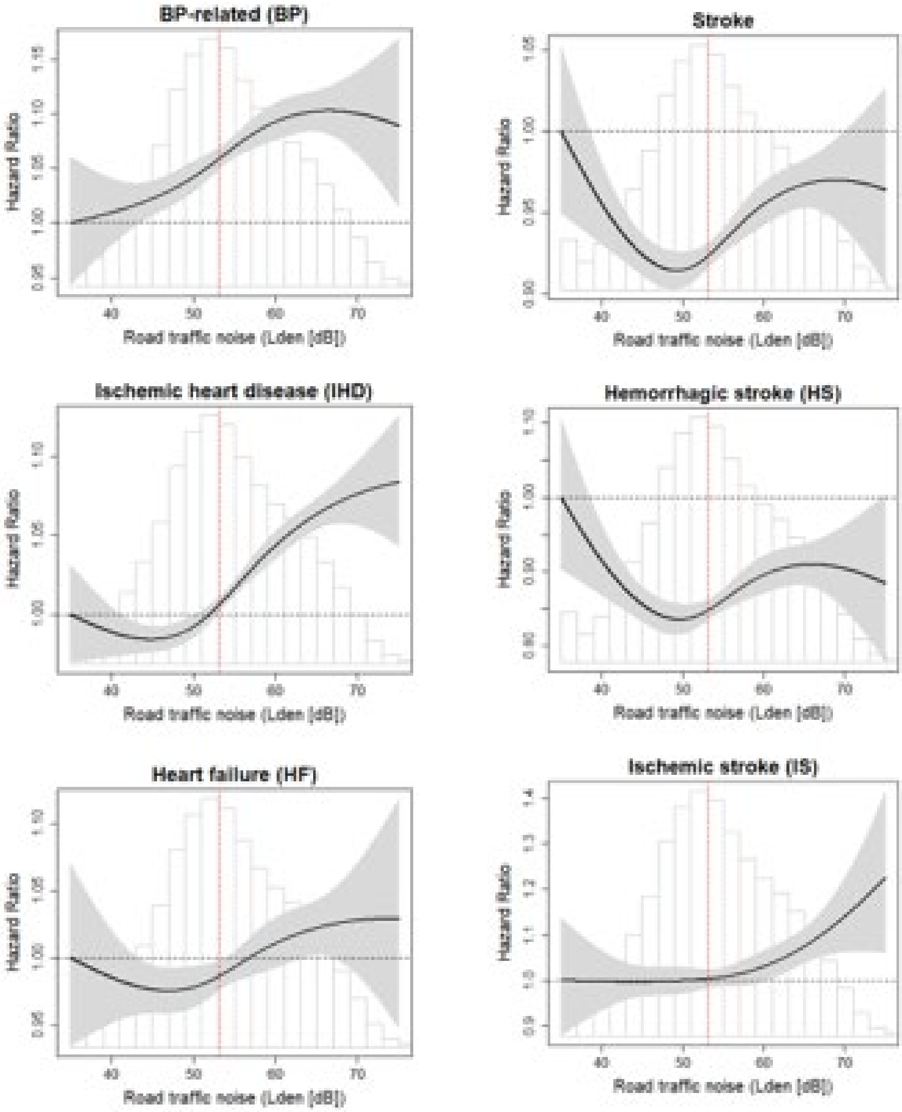



**Figure 27 Natural splines (3 df) for the association between $L_{den}$ road traffic noise and cause-specific cardiovascular mortality. Multipollutant model and adjustments as per Figure 26. Source: Supplementary Material of (Vienneau et al., 2022), Figure S.2.**

The results in Figure 28 again confirm an association between noise levels ($L_{den}$ at the most façade of a dwelling and each of risks of ischemic heart disease, myocardial infarction, angina pectoris, and heart failure - the Confidence Intervals of most of the Hazard Ratios for greater than 50 dB exposure for these health outcomes exceed 1. Similar to the finding in the Swiss study above (Vienneau et al., 2022) that the exposure-response relationship for CVD mortalities were mainly linear, with risks increasing from low noise levels, well below WHO guideline values (CVD and MI), the present study found that the increase in risk for all four outcomes investigated started already around 50 dB for $L_{den}$ at the most exposed façade of dwellings[46], and around 45 dB for $L_{den}$ at the least exposed faced of dwellings.

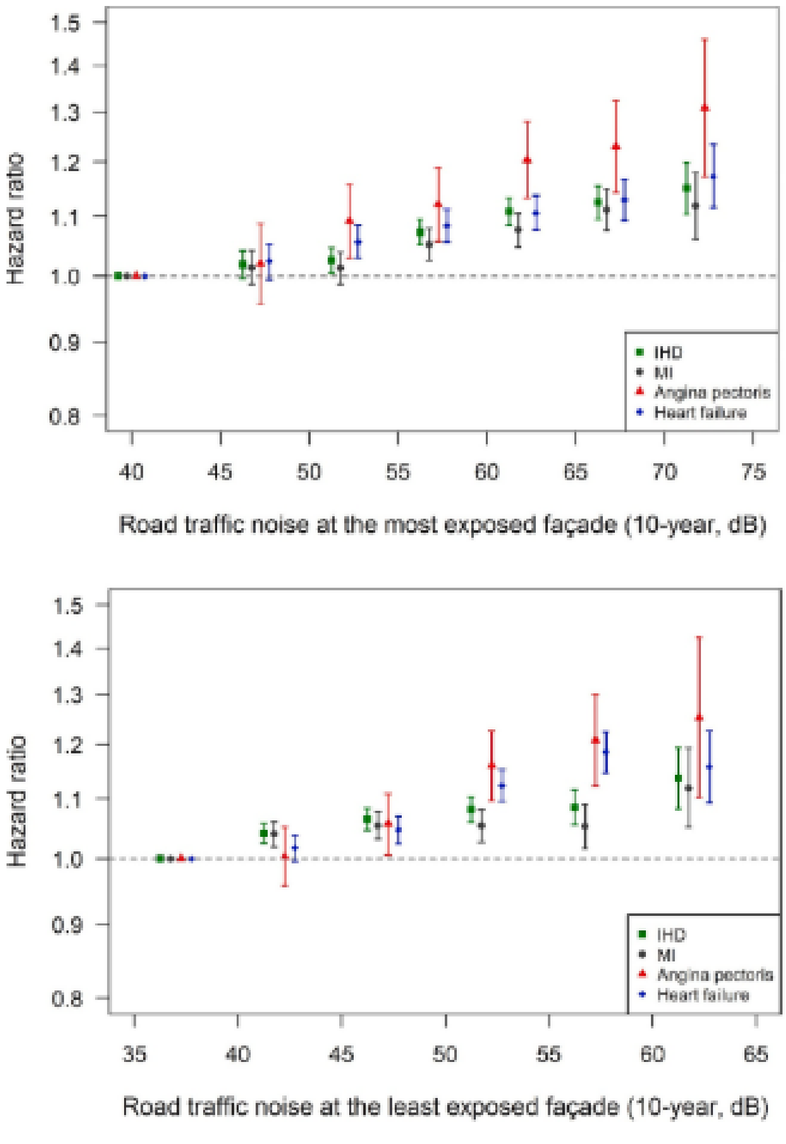



**Figure 28 Association between 10-year mean residential exposure to road-traffic noise at the most and least exposed facades and risk of ischemic heart disease, myocardial infarction, angina pectoris, and heart failure in the fully adjusted Model 2. (from: Thacher et al., 2022).**

Overall, the conclusion in this nationwide cohort study of the Danish population, was for long-term exposure to road traffic noise to increase risk of incident IHD, MI, angina pectoris, and heart failure, starting already at exposures around 50 dB ($L_{den}$) at the most exposed façade. These results add to the evidence of transportation noise as a cardiovascular risk factor and highlights the importance of preventive actions towards road traffic noise.

[46]Dwelling exposures are invariably specified at 'the most exposed façade'. The use of an $L_{den}$ at 'the least exposed façade' at the same time as one at 'the most exposed façade, has been a novel development in these epidemiological studies. It aims to minimize misclassification of the noise exposure of the affected person in the dwelling and, in particular, to better characterize the exposure while the person is sleeping. See, also, the section, '*Uncertainties in Strategic Road Traffic Noise Prediction Modelling*'.

## INDUCTION PERIOD FOR CARDIOVASCULAR EFFECTS

Given the associations, above, between transportation noise and cardiovascular effects – that is, traffic noise as a cardiovascular risk factor – what is known of the ***induction time period*** between exposure to road traffic noise and the initiation of effects? There does not appear a lot written on this topic – presumably because of a lack of studies, with appropriate longitudinal data on traffic noise exposure (historical exposures of individuals over decades is required). Babisch (2000) wrote one of the earliest reviews on traffic noise and cardiovascular diseases in which he noted the limited number of available epidemiological studies as compared to other environmental issues. He also noted: '*Studies should consider mediating factors like room orientation, window opening habits and residence time, in order to reduce exposure misclassification and to* ***account for longer induction periods*** *…'. In a write-up of the third phase of* the Caerphilly and Speedwell Studies, he also reported (Babisch et al., 1999), '*With respect to* ***induction time, we considered years in residence before the subjects entered the study in the analyses either by exclusion (subgroup greater than or equal to 15 y in residence) or*** *by interaction of residence period with noise level in the models'*.

A later paper (Babisch et al., 2005), on a hospital-based case-control study in Berlin, provides some longitudinal results.: '*Associations between noise level and MI incidence were analyzed in the total sample and in a subsample of subjects who had been living at least for 10 years in their present homes. This enabled us to account for chronic noise stress conditions and the long induction period of the disease under study. The cutpoint of 10 years was determined on the basis of the distribution of the residence time on the one hand and on pragmatic grounds of sample size and statistical power on the other. To ensure that effect estimates obtained from the subsample were stable, other criteria were also applied (eg, 15 years)*.'

**Table 10 Association Between Traffic Noise Level (dB(A), 6–22h) and Myocardial Infarction for Total Sample and for Subsample of subjects who had lived at their current address for at least 10 Years (Babisch et al., 2005).**

| | Men | | Women | |
|---|---|---|---|---|
| Traffic Noise Level | No. | OR (95% CI)* | No. | OR (95% CI)* |
| | | *Total sample* | | |
| ≤60[†] | 2231 | 1.0 | 759 | 1.0 |
| 61–65 | 355 | 1.01 (0.77–1.31) | 119 | 1.14 (0.70–1.85) |
| 66–70 | 300 | 1.13 (0.86–1.49) | 131 | 0.93 (0.57–1.52) |
| >70 | 168 | 1.27 (0.88–1.84) | 52 | 0.66 (0.32–1.35) |
| | | *Subsample* | | |
| ≤60[†] | 1547 | 1.0 | 529 | 1.0 |
| 61–65 | 251 | 1.17 (0.81–1.69) | 82 | 1.04 (0.55–1.97) |
| 66–70 | 202 | 1.31 (0.88–1.97) | 95 | 1.11 (0.62–1.98) |
| >70 | 111 | 1.81 (1.02–3.21) | 37 | 0.90 (0.39–2.07) |

*Odds ratios for men and for total sample of women are adjusted for the covariates listed in Table 1 and for indicator variables of work noise, aircraft noise, and railway noise. Odds ratios for subsample of women are adjusted only for diabetes mellitus, hypertension, family history of MI, and smoking as a result of small sample size.

[†]Reference category.

Table 10 gives the distribution of traffic noise levels in the total sample and in the subsample of subjects who had lived at their current address for at least 10 years. This refers to the highest average sound level measured during the daytime at any outside wall of the subjects' houses. Sixteen percent of the subjects' houses were exposed to sound levels of more than 65 dB(A) during the day. Two thirds (69%) of the subjects had lived at their present address for at least 10 years**. In the subsample of subjects who had lived for at least 10 years at their present address, there was a stronger monotonic increase in risk for men across the noise categories. For males in the highest noise category, the odds ratio for MI was 1.8 (1.0 –3.2)**. The result was similar when 15 years of residence was considered. No noise effect was found for women in this study – but the sample size of women was about one third of that of men. Babisch et al. (2005) conclude: '***The finding that the estimated effect is larger with longer residence is plausible and in accordance with the test hypothesis. The disease outcome under study has a long***

***induction time. One would expect many years of chronic noise stress exposure before pathologic changes become manifest**. Residence time has also been found in other studies to be an important effect (exposure) modifier of the relationship between traffic noise and cardiovascular diseases'*.

Recent studies (Thacher et al., 2022; Vienneau et al., 2022) have provided more detailed data on historical exposures.

Paraphrasing Vienneau et al. (2022):… Following a recent update, the Swiss National Cohort (SNC) now provides 15 years of follow-up enhancing the database for ongoing longitudinal research into transport noise and cardiovascular deaths, with address history and high quality noise exposure data available at multiple time points. They note that applying a time-varying approach to account for long-term variation in noise exposure, mortality rates and individual covariates is important in such studies, because **various physiological response mechanisms in cardiovascular effects of transport noise, including direct (auditory) sleep disturbances and indirect (non-auditory) stress pathways[47], may occur over a broad range of induction time periods**. Specific aims of their study included investigating cardiovascular mortality effects of transport noise exposure … taking account of time trends. The full cohort they studied was 4.1 million individuals of the Swiss population (those less than 30 years of age were excluded) for which they had address data, road (and other transport source) traffic noise exposure, and records of cardiovascular mortality, for each member of the cohort. For longitudinal research, this was considered as three 5-year periods/sub cohorts (2001-2005; adults 30+), (2006-2010; adults 35+) and (2011-2015; adults 40+), with noise exposures available at the dwelling of every member at the beginning of each of these three periods. Vienneau et al. (2022) note in their discussion that: … temporal trends were specifically considered in their analysis. As a whole, the population exposure distribution remained steady over the 15-year period, and, for road traffic noise, Hazard Ratios were also consistent across the three virtual sub-cohorts - showing little sign of time trends in mortality risk. The latter is an observation that their assumption of a 5-year induction period for cardiovascular effects of road traffic noise was likely adequate, though they also comment that: '***There is also uncertainty about the relevant induction period and what the most appropriate time-varying exposure would be** (for cardiovascular effects of traffic noise). **Our approach assumes an induction period of acute to 5 year effect, in line with the range studied in previous research … . There is no systematic research to judge this assumption. If the induction period would be longer than 5 years, our approach would result in exposure misclassification, most likely to be modest, given the moderate to high correlation of the various noise exposure metrics over time***'.

There is more evidence on this topic (Thacher, Hvidtfeldt, et al., 2020; Thacher et al., 2022). These describe studies of long-term residential road traffic noise and mortality in Danish cohorts, using a high-resolution assessment of road traffic noise exposure.

In one study (Thacher, Hvidtfeldt, et al., 2020) the researchers used the Danish cause of death register to identify cause-specific mortality for individuals in the cohort – in this study focussing on death from various identified cardiovascular diseases. The study assessed the present and historical residential addresses of 52,758 individuals from Copenhagen and Aarhus from 1987 to 2016, estimating ten-year time-weighted mean traffic noise exposures at the facades of all residences. Road traffic noise exposure was calculated for the years 1995, 2000, 2005, 2010, and 2015. Road traffic noise exposure was modelled as time-weighted averages per interquartile range (IQR) for the preceding 1, 5, and 10-years at a given age, and accounted for all addresses in the respective periods. Table 11 shows that in this study, there is little difference in the strength of the association between $L_{den}$ exposures and cardiovascular mortality irrespective of whether the individual's exposure had been estimated over 1 year, 5 years or 10 years.

[47] See Figure 29 below.

**Table 11 Associations between residential exposure to traffic noise at the most exposed façade per 10.4 dB (IQR) and mortality risk (Thacher, Hvidtfeldt, et al., 2020)**

Associations between residential exposure to traffic noise at the most exposed façade per 10.4 dB (IQR) and mortality risk.

| Exposure to road traffic noise, most exposed façade (per 10.4 dB) | N cases | Model 1[a] HR (95% CI) | Model 2[b] HR (95% CI) | Model 3[c] HR (95% CI) |
|---|---|---|---|---|
| **All-cause** | | | | |
| 1-year mean exposure | 11,596 | 1.17 (1.14–1.20) | 1.09 (1.06–1.12) | 1.08 (1.05–1.11) |
| 5-year mean exposure | 11,596 | 1.18 (1.15–1.22) | 1.10 (1.07–1.13) | 1.08 (1.05–1.11) |
| 10-year mean exposure | 11,596 | 1.19 (1.16–1.23) | 1.10 (1.07–1.13) | 1.08 (1.05–1.11) |
| **CVD** | | | | |
| 1-year mean exposure | 2623 | 1.22 (1.16–1.29) | 1.12 (1.06–1.18) | 1.11 (1.05–1.17) |
| 5-year mean exposure | 2623 | 1.24 (1.17–1.31) | 1.13 (1.07–1.19) | 1.11 (1.05–1.18) |
| 10-year mean exposure | 2623 | 1.26 (1.20–1.34) | 1.14 (1.08–1.21) | 1.13 (1.06–1.19) |
| **Ischemic heart disease** | | | | |
| 1-year mean exposure | 938 | 1.16 (1.06–1.27) | 1.06 (0.96–1.16) | 1.05 (0.96–1.15) |
| 5-year mean exposure | 938 | 1.17 (1.07–1.28) | 1.05 (0.96–1.16) | 1.04 (0.95–1.14) |
| 10-year mean exposure | 938 | 1.18 (1.07–1.29) | 1.05 (0.95–1.15) | 1.03 (0.94–1.14) |
| **Stroke** | | | | |
| 1-year mean exposure | 636 | 1.16 (1.04–1.29) | 1.10 (0.98–1.23) | 1.10 (0.98–1.23) |
| 5-year mean exposure | 636 | 1.18 (1.05–1.32) | 1.11 (0.99–1.25) | 1.11 (0.99–1.24) |
| 10-year mean exposure | 636 | 1.19 (1.06–1.33) | 1.12 (1.00–1.26) | 1.11 (0.99–1.25) |
| **Respiratory** | | | | |
| 1-year mean exposure | 2363 | 1.15 (1.09–1.22) | 1.04 (0.98–1.10) | 1.02 (0.96–1.08) |
| 5-year mean exposure | 2363 | 1.18 (1.12–1.25) | 1.06 (1.00–1.12) | 1.03 (0.97–1.09) |
| 10-year mean exposure | 2363 | 1.20 (1.13–1.27) | 1.06 (0.99–1.12) | 1.02 (0.96–1.09) |
| **Cancer** | | | | |
| 1-year mean exposure | 5465 | 1.09 (1.05–1.13) | 1.05 (1.01–1.09) | 1.03 (0.99–1.07) |
| 5-year mean exposure | 5465 | 1.09 (1.05–1.13) | 1.05 (1.01–1.09) | 1.03 (0.99–1.07) |
| 10-year mean exposure | 5465 | 1.09 (1.05–1.14) | 1.04 (1.00–1.08) | 1.02 (0.98–1.06) |

[a] Age, sex, and calendar year.
[b] As model 1, and further adjusted for level of education, disposable income, cohabitation, area-level proportion of low income, basic education, and unemployment.
[c] As model 2, and further adjusted for, smoking status, smoking duration, smoking intensity, alcohol intake, abstainers, sport during leisure time (y/n), sport (hrs/week), vegetable intake, and fruit intake.

# MECHANISMS: THE NOISE EFFECTS REACTION SCHEME

In addition to the review of the evidence of the effects of transport noise on cardiovascular and metabolic conditions arising from epidemiological studies, discussed above, Münzel et al. (2021) also reviewed mechanistic studies of noise exposure in humans and laboratory studies of the effects of noise in animals. They also flag several potential effects of noise exposure emerging in the literature: *viz.,* alterations to gut biota; to circadian rhythm; and potential epigenetic regulation. These matters will not be discussed further here in this scoping study.

However, it is useful to report their current thinking with respect to the '*noise-stress'* concept and the pathways and mechanisms in the human body by which noise exposure may lead to the range of effects discussed throughout this report. Figure 29 shows this noise reaction model. The labels with pink-shaded backgrounds are the health consequences. In particular, as described in the Figure 29 label, '*noise exposure triggers signalling via the hypothalamic–pituitary–adrenal axis and sympathetic nervous system (SNS). In the hypothalamic–pituitary–adrenal axis, the hypothalamus releases corticotropin-releasing hormone (CRH; also known as corticoliberin) into the pituitary gland, which stimulates the release of adrenocorticotropic hormone (ACTH) into the blood. ACTH induces the production of glucocorticoids by the adrenal cortex, and the activation of the SNS stimulates the production of catecholamines by the adrenal medulla. The release of glucocorticoids and catecholamines in turn leads to the activation of other neurohormonal pathways (such as the renin–angiotensin–aldosterone (RAAS) system) and to increased inflammation and oxidative stress, which can ultimately have adverse effects on cardiovascular function and molecular targets*'.

It is not anticipated that most readers of this scoping review (and similarly for its author) will have sufficient knowledge of human physiology to be able to comprehend these mechanisms. However, it is still regarded as critical that there be an understanding in the acoustics/transport/policy-making community of the potential scientific basis for the links between noise and the wide range of health outcomes that are being investigated. This is particularly the case in those jurisdictions (most) where '*annoyance'* has been the predominant, if not only, human non-auditory effect of road traffic noise that has been considered to date.

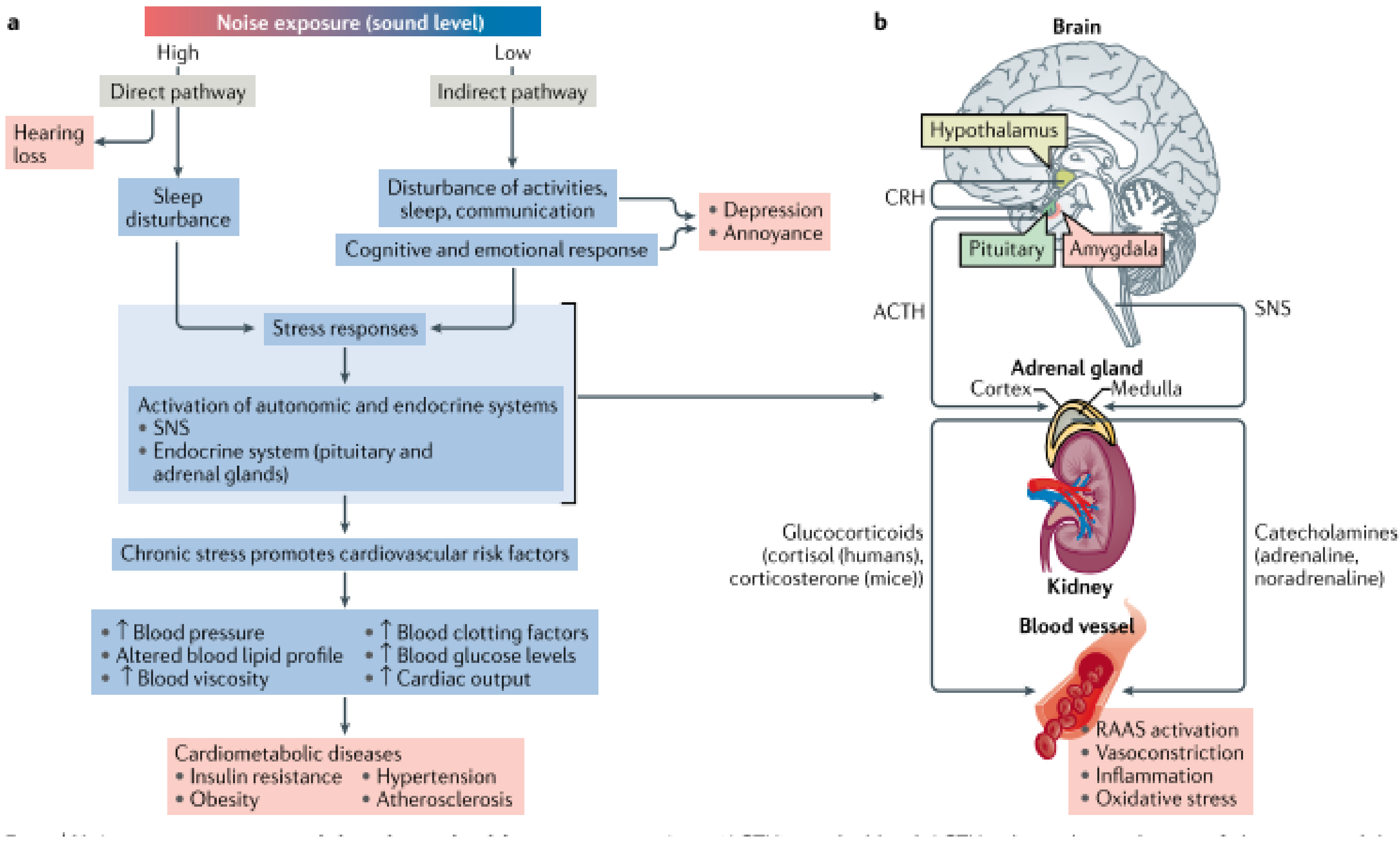



**Figure 29 Noise–stress concept and the adverse health consequences in humans. a | Noise reaction model for the direct (auditory) and indirect (non-auditory) effects of noise exposure. b | Neuronal activation (arousals) induced, for example, by noise exposure triggers signalling via the hypothalamic–pituitary–adrenal axis and sympathetic nervous system (SNS). In the hypothalamic–pituitary–adrenal axis, the hypothalamus releases corticotropin-releasing hormone (CRH; also known as corticoliberin) into the pituitary gland, which stimulates the release of adrenocorticotropic hormone (ACTH) into the blood. ACTH induces the production of glucocorticoids by the adrenal cortex, and the activation of the SNS stimulates the production of catecholamines by the adrenal medulla. The release of glucocorticoids and catecholamines in turn leads to the activation of other neurohormonal pathways (such as the renin–angiotensin–aldosterone (RAAS) system) and to increased inflammation and oxidative stress, which can ultimately have adverse effects on cardiovascular function and molecular targets (Copied from: Münzel et al., 2021).**

## NOISE AND COGNITIVE DEVELOPMENT/LEARNING

The systematic review on environmental noise and cognition for the 2018 update of the WHO guidelines was completed by Clark and Paunovic (2018a)[48]. They noted that the existing WHO guidelines for children's environmental noise exposure (Berglund et al., 1999), were published prior to much evidence being available. Focusing on children's environmental noise exposure, the existing guidelines suggest that school playgrounds outdoors should not exceed 55 dB $L_{Aeq}$ during play to protect from annoyance and that school classrooms should not exceed 35 dB $L_{Aeq}$[49] during class to protect from speech intelligibility and, disturbance of information extraction. The authors note that that 35 dB $L_{Aeq}$ is a very low level of noise exposure, considered unachievable by some stakeholders.

Several plausible pathways and mechanisms for the effects of noise on children's cognition have been put forward: noise may directly affect children's cognitive abilities such as reading comprehension, but effects could also be accounted for by other mechanisms such as teacher and pupil frustration, learned helplessness (low motivation to learn resulting from lack of control over one's environment), and increased arousal which can impact task performance. Experimental studies of acute exposure show negative effects on speech perception and listening comprehension. Noise most likely interferes with the interactions between teachers and pupils: teachers may have to stop teaching whilst noise events occur which may contribute to a reduction of morale and motivation in teachers. Another pathway is impaired attention - it has been suggested that children exposed to environmental noise at school may cope with noise exposure by 'tuning out' the noise: this strategy may then be over-generalized resulting in poorer learning experiences,

Since publication of (Clark & Paunovic, 2018a), there have been at least two updates – including one by Clark et al. (2020). This did not have a detailed analysis of the different cognitive faculties as outcomes nor any meta-analysis and is not discussed here. The other was by Thompson et al. (2022) who identified 16 new studies, reviewing these in tandem with the 32 studies previously reviewed. A summary of the results is provided in Figure 30. Their finding is that there is moderate to low quality evidence of some associations between noise exposure and a range of cognitive outcome measures for children, but more good quality research using standardised methodology is required to corroborate these results and to allow for precise risk estimation by larger meta-analyses.

| | Children | Adults |
|---|---|---|
| Academic ability | ✓ | ✓ |
| Attention | ✓ | ✓ |
| Memory and learning | ✓ | × |
| Executive function | × | ✓ |
| Reading and language | ✓ | ✓ |
| Fluid intelligence and general cognition | ✓ | 50/50 |
| Cognitive impairment | n/a | ✓ |
| Perceptual speed | n/a | ✓ |

Associated in literature: ✓

Not associated in literature: ×

High Quality evidence

Moderate Quality evidence

Low Quality evidence

Very low quality evidence

**Figure 30 Summary of direction (supportive or not supportive) and rating of confidence (high, moderate, low, very low) in evidence for associations between environmental noise and cognitive outcomes (from Thompson et al., 2022). Note that 'environmental noise' includes sources other than road traffic.**

The most compelling evidence regarding children's cognition comes from a carefully conducted new study, Foraster et al. (2022). It followed up a population-based sample of 2,600 children aged 7-10 years from 38 schools in

[48] The same authors also undertook the review (Clark & Paunovic, 2018b) of quality of life, wellbeing and mental health guidelines for community noise (not discussed in this scoping study).

[49] The NSW Road Noise Policy (Department of Environment Climate Change and Water NSW, 2011) criterion for school classrooms is $L_{Aeq,(1\ hour)}$ of 40dB (internal) when in use.

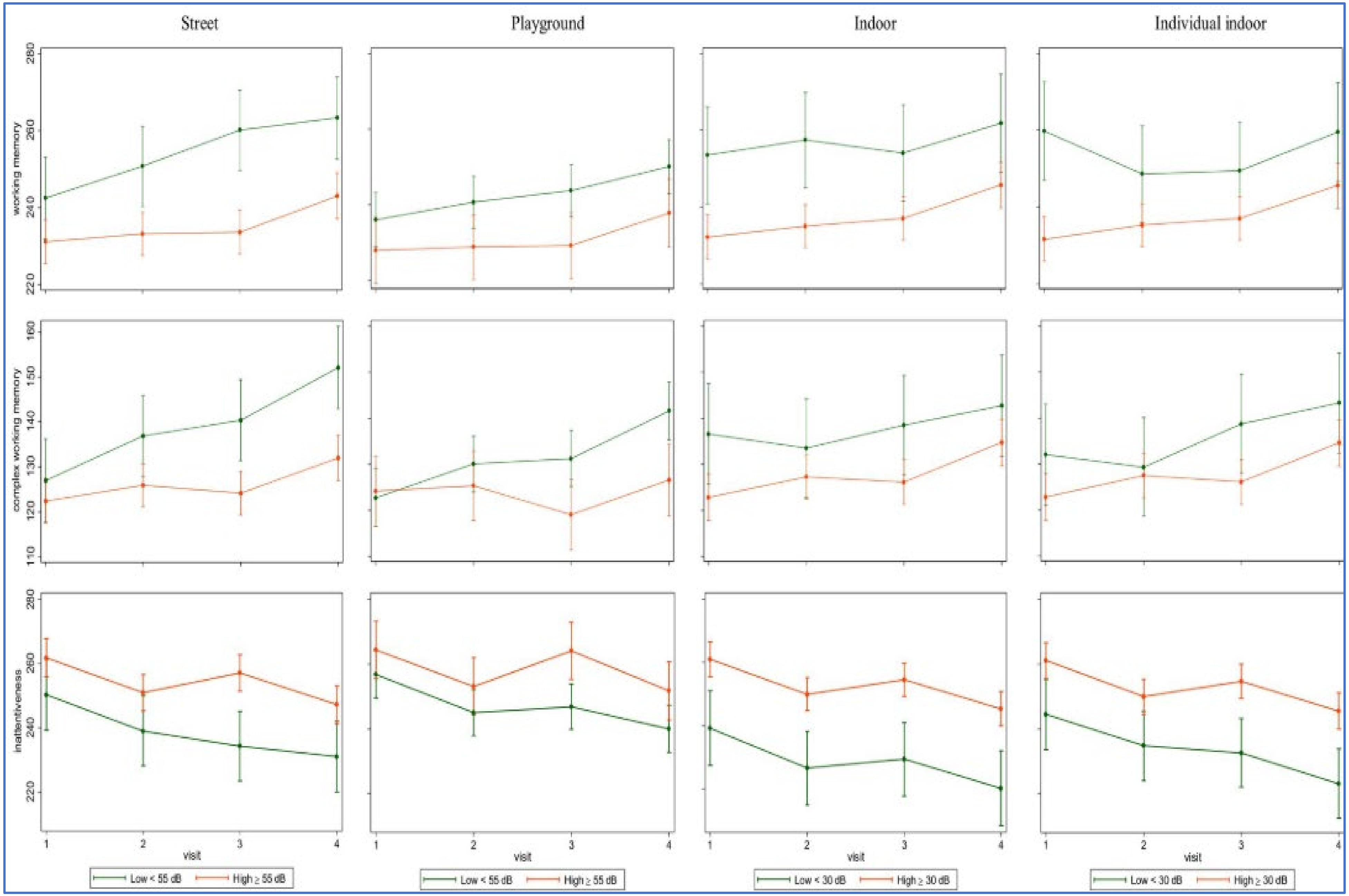
Barcelona with the intent to obtain more information about the effect of road traffic noise on children's cognition. They studied the association between exposure to road traffic noise and the development of working memory and attention in primary school children, considering school-outdoor and school-indoor annual average noise levels and noise fluctuation characteristics, as well as home-outdoor noise exposure. They tested children four times over a year (x axis) as shown in Figure 31, with the outcomes shown at each of the four test times

Trajectories of working memory and inattentiveness at schools (Fig 30) show that children who attended schools with high road traffic noise at street level (>=55 dB) had a slower development of working memory, complex working memory, and a slower improvement of inattentiveness over 12 months than those attending quieter schools - in adjusted models. Similar trends with slightly weaker differences between groups were observed for schools exposed to high noise at the playground. Finally, children who attended schools with high road traffic noise in the classroom (>=30 dB) had a slower improvement of inattentiveness over 12 months than those who attended schools with quieter classrooms.

The overall findings of the study were that exposure to road traffic noise at school, but not at home, was associated with slower development of working memory, complex working memory, and attention in schoolchildren over one year. An additional interesting result was that associations with noise fluctuation indicators (*e.g.*, number of noise events) were more evident than with average noise levels in classrooms.

This result shows only an association between road traffic noise levels in schools and cognition – it is not, *per se*, evidence that the road traffic noise causes the cognitive differences. However, when looking for recommended internal levels of exposure to road traffic noise in classrooms, the Foraster et al. (2022) study provides is a good indication that internal $L_{Aeq}$ levels <30dB are likely to be a better environment for children's cognition than levels >30dB.

**Figure 31 (OVER PAGE) Annual trajectories of working memory, complex working memory, and inattentiveness in children attending schools with low (green: < 55dB outdoors/<30dB indoors) and high (orange: >55dB outdoors/>30dB indoors) average road traffic noise levels ($L_{Aeq}$, dB) outdoors in street and playground or indoors in 1 classroom (indoor level) or in each child's classroom considering change of room between years (individual indoor level). Y axis: point estimate (beta coefficient), error bars (95% confidence intervals). Predictions for working memory (2-back number stimuli, d0), complex working memory (3-back number stimuli, d0), and inattentiveness (HRT-SE, ms) adjusted at the means of age, sex, corresponding road traffic noise indicator, age*road traffic noise indicator, maternal education, socioeconomical vulnerability index at home and outdoor or indoor TRAP at school for models including outdoor or indoor noise levels, respectively. Child and school included as nested random effects. d0: detectability, a higher value indicates better working memory; HRT-SE: hit reaction time standard error, a higher value indicates greater inattentiveness; $L_{Aeq}$: A-weighted equivalent noise levels; TRAP: traffic-related air pollution.**



Street
Playground
Indoor
Individual indoor
working memory
complex working memory
inattentiveness
visit
Low < 55 dB
High ≥ 55 dB
Low < 30 dB
High ≥ 30 dB

# DEMENTIA

A recent large scale study (2.1 million adults aged greater than 60 years) in Denmark (Cantuaria et al., 2021)[50] found evidence of association or road traffic noise (and also, but independently, rail traffic noise) with all-cause dementia and dementia subtypes (Alzheimer's disease, vascular dementia, and Parkinson's disease related dementia) each identified from national hospital and prescription registries. People were excluded from the study if they had a diagnosis of dementia before the baseline year. Previous research on transportation noise and dementia had been scarce and the few existing studies generally indicated no association between transportation noise and dementia.

Cantuaria et al. (2021) predicted noise levels, using Nordic models, at both the most-exposed (max) and at the least-exposed (min) façades for the addresses of every individual in the study. Road traffic $L_{den}$ were modelled for each of the years: 1995, 2000, 2005, 2010, and 2015 for all Danish addresses, and these were interpolated, for each individual's specific address history, to calculate 10-year running means of exposure for all individuals in the study. Some results, expressed as Hazard Ratios[51] for categories of 10-year exposures (after adjustments for a wide range of co-variates) are shown in Table 12, referenced to a <45 $L_{den}$ category for most- exposed facade (referenced to <50 $L_{den}$ category for least-exposed façade). For example, the table shows a 10 year most-exposed facade category of 55-60 $L_{den}$ is associated with 1.17 times (95% CI 1.14 to 1.20) the risk of all-cause dementia as do people in the 10-year exposure category of <45dB. The study also adjusted for $PM_{2.5}$ and $NO_2$ as covariates, with only a small reduction in the Hazard Ratios, demonstrating that it is noise exposure, not transport exposure (noise and air pollution), that is associated with the risk of dementia. These results in Table 12 are also shown in an exposure-response association form in Figure 32 – and similarly for each risk of dementia subtype in Figure 33.

The Cantuaria et al. (2021) work was based on a 'survival analysis' using the **Cox proportional-hazards model** - essentially a regression model commonly used in statistical in medical research for investigating the association between the survival time of patients and one or more predictor variables[52].

Again, it is important to note that this study is evidence of an ***association*** between road traffic noise exposure in the home and dementia (it does not demonstrate causality). However, taken together with evidence discussed on earlier pages of the associations with cardiovascular and metabolic effects, and sleep effects, it does add somewhat to the view that health effects of road traffic noise may start to emerge at $L_{den}$ residential exposures to road traffic noise at the most expose façade above 50 dB

[50] As for the (Münzel et al., 2021) cardiovascular effects paper, this dementia paper was published in a prestigious journal: the BMJ with an Impact Factor of 96.2.
[51] Again, in summary, HR = 1 No effect; HR < 1 Reduction in the hazard; HR > 1 Increase in Hazard.
[52] The Cox Model survival analysis is not a technique familiar to the current author. For example, while it is the same data, it is unclear to this author how the Hazard ratios in Figure 32 are related to those in Table 12.

**Table 12 Associations between categories of 10 year mean residential exposure to road traffic noise at the most ($L_{denmax}$) and least ($L_{denmin}$) exposed residential facades and risk of all-cause dementia (extracted from: Cantuaria et al., 2021).**

| | All cause dementia | |
|---|---|---|
| Noise exposure (10 year) by category | No of cases | Hazard ratio (95% CI)* |
| **Road** | | |
| $L_{den}$max (dB): | | |
| <45 (reference) | 9718 | 1 |
| 45-50 | 13 500 | 1.09 (1.06 to 1.12) |
| 50-55 | 21 738 | 1.16 (1.13 to 1.19) |
| 55-60 | 24 533 | 1.17 (1.14 to 1.20) |
| 60-65 | 21 196 | 1.16 (1.13 to 1.19) |
| ≥65 | 12 815 | 1.16 (1.13 to 1.19) |
| $L_{den}$min (dB) | | |
| <40 (reference) | 18 136 | 1 |
| 40-45 | 30 306 | 1.12 (1.10 to 1.14) |
| 45-50 | 27 261 | 1.18 (1.16 to 1.20) |
| 50-55 | 18 005 | 1.21 (1.19 to 1.24) |
| ≥55 | 9792 | 1.18 (1.15 to 1.21) |

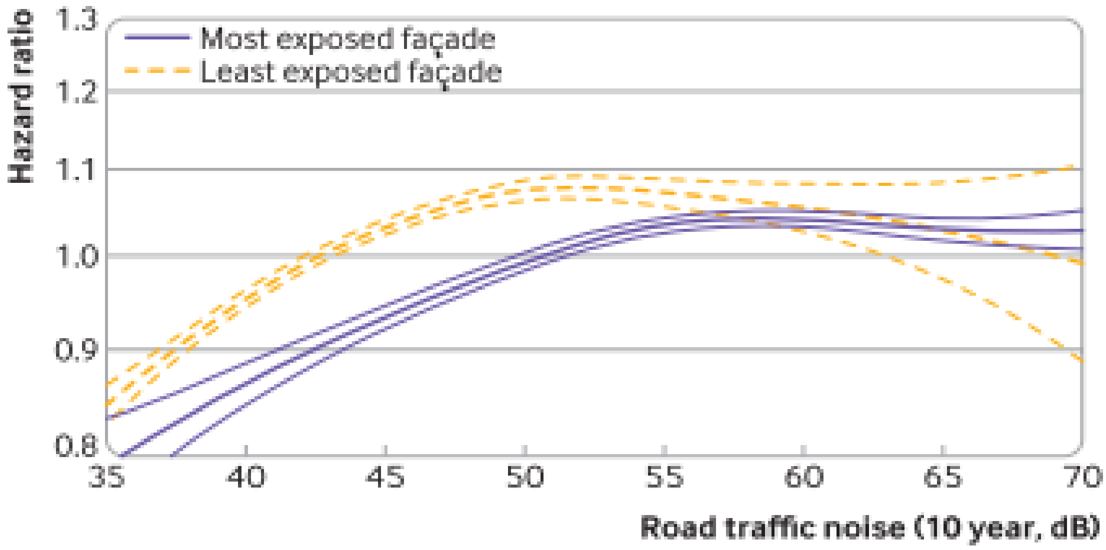



**Figure 32 Associations between 10 year mean exposure to road traffic noise at the most ($L_{den,max}$) and least ($L_{den,min}$) exposed façades of buildings and risk of all cause dementia (fully adjusted for many covariates, including $PM_{2.5}$ and $NO_2$). Figure shows hazard ratios and corresponding 95% confidence intervals (Cantuaria et al., 2021).**

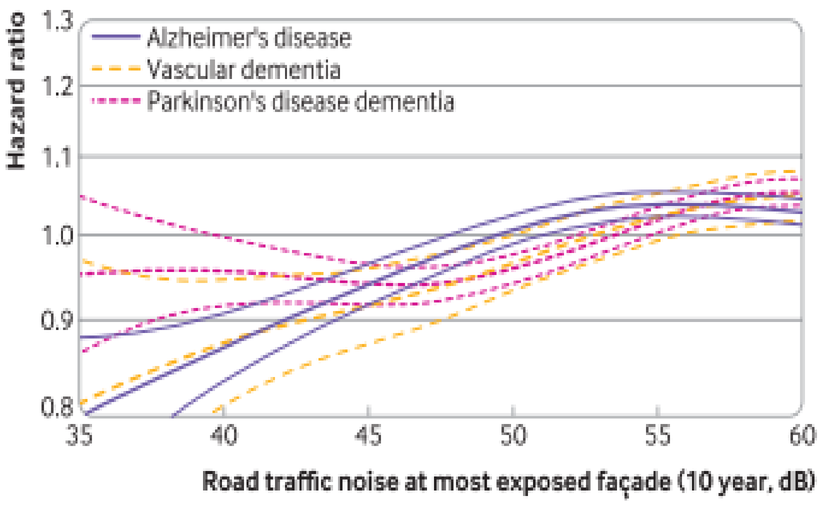



**Figure 33 Associations between 10-year mean exposure to road traffic noise at the most exposed façades of buildings and risk of dementia subtypes, using the fully adjusted model (model 2). Figure shows hazard ratios and corresponding 95% confidence intervals (Cantuaria et al., 2021).**

# OTHER HEALTH OBSERVATIONS (not particularly relevant for NSW)

## LOW-MIDDLE-INCOME-COUNTRIES

Based on a systematic literature review by Chen et al. (2022), of available epidemiological evidence of the effects of environmental noise on health in Low-Middle-Income-Countries (for the period 2009-2021), the strength of evidence for each outcome covered in the literature was assessed to be 'inadequate'. This highlights that high-quality epidemiological studies are needed in LMICs to strengthen the evidence base. It is unclear whether findings in high-income countries can readily be translated into policy contexts in low-middle-income-countries.

## RELATIONSHIPS BETWEEN TRANSPORTATION NOISE, SELF-REPORTED RESPONSES TO NOISE AND WIDER DETERMINANTS OF HEALTH

From a completely different perspective, Peris and Fenech (2020) reported an innovative review examining the relationship between noise, annoyance, and wider determinants of health. The latter were defined as: lifestyle (physical activity, smoking, and use of medication) community (home moving and social interaction), local economy, activities, built environment, natural environment, and global ecosystem. According to the authors, this was the first attempt at synthesising the current evidence on the associations and modification effects between transportation noise and the wider determinants of health. By bringing together evidence from many different published articles, they provide insight into the complex pathways through which sound and noise may be indirectly influencing people's health and wellbeing. The review looked for two-way evidence, *viz.* the effect of noise exposure and human response on these wider determinants of health, and the effect of these wider determinants on noise exposure and human response. Some 76 papers met the inclusion criteria for the synthesis, and a (narrative) summary of the main findings is below (citations were provided for these summaries in the original paper but have not been included here).

Main findings were:

- transportation noise could affect recreational activities through worsening the quality of leisure time, preventing people from practicing such activities and even stopping people walking in high streets.
- traffic noise is associated with a reduced engagement in physical activities. The annoyance response seems to play a role on the levels of physical activity – and through this path it may indirectly contribute towards cardiovascular disease, as well as to diabetes and obesity, by reducing physical activity.
- road traffic noise also seems to be associated with other non-healthy behaviours such as smoking, but these observations come from cross-sectional data and did not evaluate the relationship between smoking and the self-reported response to noise
- road noise also seems to influence property prices; however, this price devaluation is much lower for road than for aircraft noise, agreeing with previous reviews.
- some characteristics of the neighbourhood can improve mental health and well-being by reducing noise annoyance or other negative responses to noise. Provision of green spaces (i.e., having access or a view to green) nearby or providing a quiet side or quiet courtyard clearly decreases annoyance due to road traffic noise.

While these are tentative results, they do indicate that transportation noise may have the potential to affect health through various pathways.

# INTERVENTIONS & CHANGE EFFECTS (EXCESS RESPONSE)

As indicated in an earlier section of this document, '*2018 WHO Environmental Noise Guidelines for the European Region*', in addition to the seven systematic reviews of the relationship between environmental noise and health outcomes undertaken for the WHO ENG (annoyance; cardiovascular and metabolic effects; cognitive impairment; effects on sleep; hearing impairment and tinnitus; adverse birth outcomes; and quality of life, mental health, and wellbeing), an eighth systematic review assessed the effectiveness of environmental noise interventions in reducing impacts on health. The aim of this review was to examine if existing practices in the management of environmental noise exposure by barriers, speed reductions, window glazing etc, not only reduced noise levels, but also had an effect in reducing measurable health effects. The difficulty in the task was gathering evidence of such *measurable* changes in health effects associated with past interventions.

This work was reported by Brown and Van Kamp (2017). Their paper, '*WHO environmental noise guidelines for the European region: A systematic review of transport noise interventions and their impacts on health*', described the review of evidence (1980–2014) on effects of transport noise interventions on human health (all transport sources). Health outcomes included sleep disturbance, annoyance, cognitive impairment of children and cardiovascular diseases. As there had been no previous work in this field, Brown and Van Kamp (2017) needed first to develop a conceptual framework) to classify noise interventions and health effects and this is shown in Figure 34.

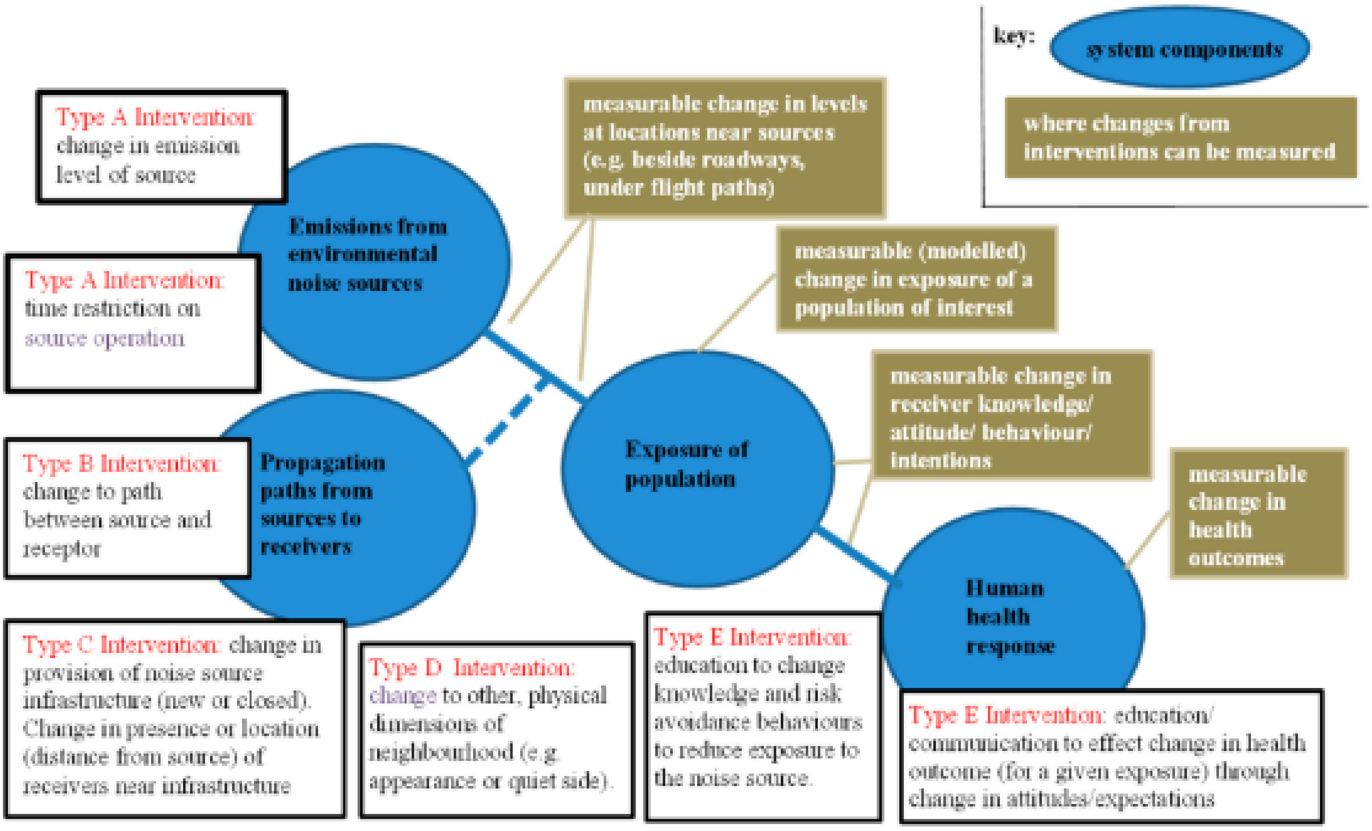



**Figure 34 Intervention framework showing: system components of the path between environmental noise and human health, where different types of noise intervention potentially act along that path, and points along the pathway where changes resulting from interventions can be measured (Brown & Van Kamp, 2017).**

The evidence was thinly spread across source types, outcomes, and intervention types. Further, diverse intervention study designs, methods of analyses, exposure levels, and changes in exposure did not allow a meta-analysis of the association between changes in noise level and health outcomes, and risk of bias in most studies was high. However, 43 individual transport noise intervention studies were examined (33 road traffic; 7 air traffic; 3 rail) as to whether the intervention was associated with a change in health outcome (see Table 13 for road traffic noise interventions only).

**Table 13 Number of Individual Road Traffic Noise Studies for each Outcome Measure and Intervention Type (Brown & Van Kamp, 2017).**

| | Number of Peer Reviewed Papers | Number of Non-Peer Reviewed Papers | Total Papers per Group |
|---|---|---|---|
| **ROAD TRAFFIC NOISE SOURCES** | | | |
| **Outcome: Annoyance** | | | |
| A Source Intervention | 7 | 3 | 10 |
| B Path Intervention | 4 | 2 | 6 |
| C New/Closed Infrastructure | 1 | 1 | 2 |
| D Other Physical | 6 | 1 | 7 |
| **Outcome: Sleep Disturbance** | | | |
| A Source Intervention | 1 | - | 1 |
| B Path Intervention | 1 | 1 | 2 |
| C New/Closed Infrastructure | 2 | - | 2 |
| D Other Physical | 1 | - | 1 |
| **Outcome: Cardiovascular Effects** | | | |
| D Other Physical | 4 | - | 4 |
| **Outcome: Modelled Change in Exposure/Effect *** | | | |
| A Source Intervention | 1 | 1 | 2 |

The findings for road traffic are in Table 14. Nearly all the interventions were associated with changes in health outcomes irrespective of the outcome or intervention type (source, path, or infrastructure).

**Table 14 Summary of evidence from the individual road traffic noise studies on the effect of the intervention on health outcomes Brown and Van Kamp (2017).**

| | Number of Papers | Evidence [1] That Health Outcome Changed | | | Observed Magnitude of Change in Health Outcome | | |
|---|---|---|---|---|---|---|---|
| | | YES | NO | n.a. | Magnitude *at Least* as Predicted by ERF | Excess [2] Response | n.a.[3] |
| ROAD TRAFFIC NOISE SOURCES (33) | | | | | | | |
| Outcome: Annoyance (23) | | | | | | | |
| A Source Intervention | 9 | ******* | | ** | ******* | ******* | ** |
| B Path Intervention | 6 | ****** | | | **** | ** ? | ** |
| C New/Closed Infrastructure | 2 | ** | | | ** | ** | |
| D Other physical | 6 | ****** | | | | | |
| Outcome: Sleep Disturbance (6) | | | | | | | |
| A Source Intervention | 1 | | | * | | | * |
| B Path Intervention | 2 | ** | | | | | ** |
| C New/Closed Infrastructure | 2 | ** | | | | | ** |
| D Other physical | 1 | * | | | | | |
| Outcome: Cardiovascular Effects (4) | | | | | | | |
| D Other physical | 4 | *** | * | | | | |

* Statistical significance of finding reported in the original study. * Finding interpreted by original, or current, authors based on data/tables/plots in original study. [1] Note that the evidence is indirect for Interventions Type D (Other Physical). [2] Excess response occurs where the total difference between the observed before and after outcomes is greater than the magnitude of the change in response estimated from an ERF, for a given change in exposure. [3] n.a. = not applicable/not available: no change in exposure or not reported. ? = unclear finding.

The Brown and Van Kamp (2017) review provided a positive answer to an important policy question: "*do environmental noise interventions change health outcomes?"* It shows that many current noise management strategies have a beneficial effect on human health. The caveat is that this evidence is not extensive or well distributed over all transport noise sources, intervention types, or health outcomes.

Another finding is that relevant ERFs for annoyance can provide an estimate of the minimum change in human outcomes that can be expected from a given change in exposure as a result of an intervention. This supports current noise management as ERFs for annoyance can thus provide a first conservative estimate for the health impact assessment of future interventions.

The review demonstrated that there was ***excess response*** to the intervention in 14 road traffic noise interventions. This 'Excess response' change effect occurs where the total difference between the before-outcome and the after-outcomes is greater than the magnitude of the change in response estimated from an ERF - for the given change in exposure (see graphic of excess response in Figure 35),

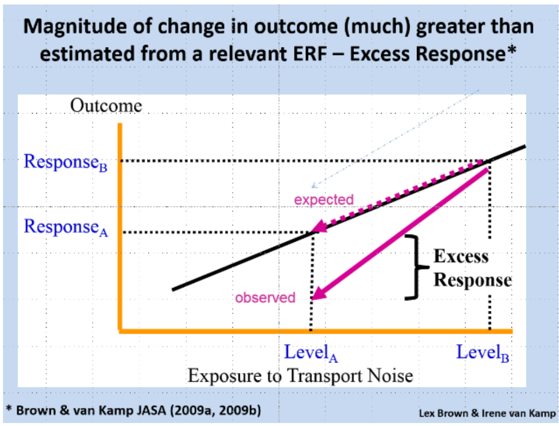



Figure 35 Graphic illustrating Excess Response to an intervention. B – before, A – after, the intervention.

The notion of excess response to interventions has been considered in depth by Brown and van Kamp (2009b, 2009a) where they examined (and rejected some of) the many explanations that have been proposed for this phenomenon. They reported:

> *'The evidence of the magnitude, and the persistence over time, of the change effect . . . and the existence of plausible explanations for it, suggest that it is a real effect and needs to be taken into account in assessing the response of communities in situations where noise levels change. Within the limitations of existing evidence on change, communities that experience an increase in noise exposure are likely to experience greater annoyance than is predicted from existing exposure–response relationships, and communities that experience a decrease in exposure experience greater benefit than predicted. Policy makers need to be informed of these potential change effects, particularly as situations in which noise levels increase as a result of infrastructure changes are always likely to be contentious. To do otherwise would be to deny them important information regarding potential community response in these contexts'.*

They also suggested that:

> *'Authorities proposing/funding interventions, whether at local, national, or international level, and whether or not the primary purpose of the intervention concerns noise, should be encouraged to include significant funding for the design and implementation of studies to evaluate outcomes from the interventions. At present, many of the evaluations appear to be addendums to, rather than integral components of, the interventions'.*

## AN URBAN ROAD TRAFFIC SPEED REDUCTION INTERVENTION: ZURICH

The city of Zurich progressively pursues a strategy of reducing road traffic noise by lowering the speed limit to 30 km/h on street sections that exceed the legal noise limits. Brink et al. (2022) evaluated the effects of the reduced speed limit on noise levels ($L_{day}$ and $L_{night}$), noise annoyance, self-reported sleep disturbance, perceived road safety. They surveyed about 1300 randomly sampled inhabitants, in a repeated measures study, before and after the speed rule changeover from 50 km/h to 30 km/h along 15 city street sections. Road traffic noise $L_{eq}$ at the loudest façade points dropped by an average of 1.6 dB during day and 1.7 dB at night (Figure 36). A statistically significant decrease of noise annoyance and of self-reported sleep disturbances was observed – reflecting the findings reported in Table 14 that physical traffic noise interventions lead to measurable change in noise effects (in this case, both in annoyance and in self-reported sleep effects, even though mean changes in exposure as a result of the intervention were small. A moderate but significant increase of perceived road safety was also measured in the study.

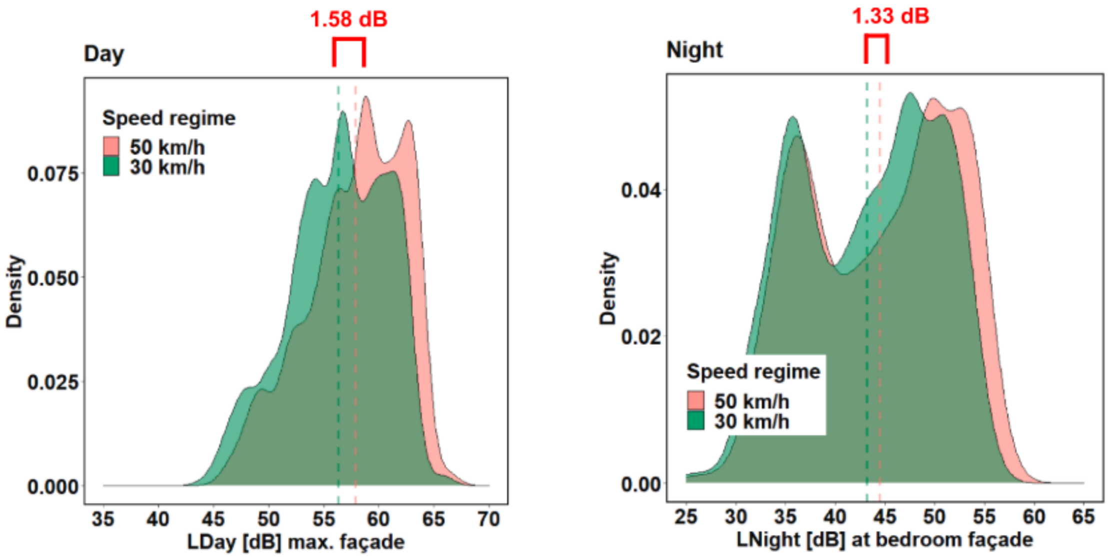



Figure 36 Estimated density distributions of the average road traffic noise level during the day (left) and at night (right) for the two different speed regimes. The dashed vertical lines indicate the respective mean value (Brink et al., 2022).

However, not only did the %HA and the %HSD drop significantly as a result of the relatively small change in exposures, but there was also an **excess response** whereby the annoyance and sleep disturbance scores dropped further than would have been expected from the 1.6 dB and 1.4 dB reductions respectively. This is illustrated most readily in Figure 37, where exposure–response relationships for annoyance and sleep disturbance are seen to be shifted towards lower effects in the 30 km/h condition by, depending on receiver point, between about 2 dB and 4 dB during the day and about 4 dB at night. These magnitudes of change effect conform broadly to those reported previously (Brown & van Kamp, 2009a).

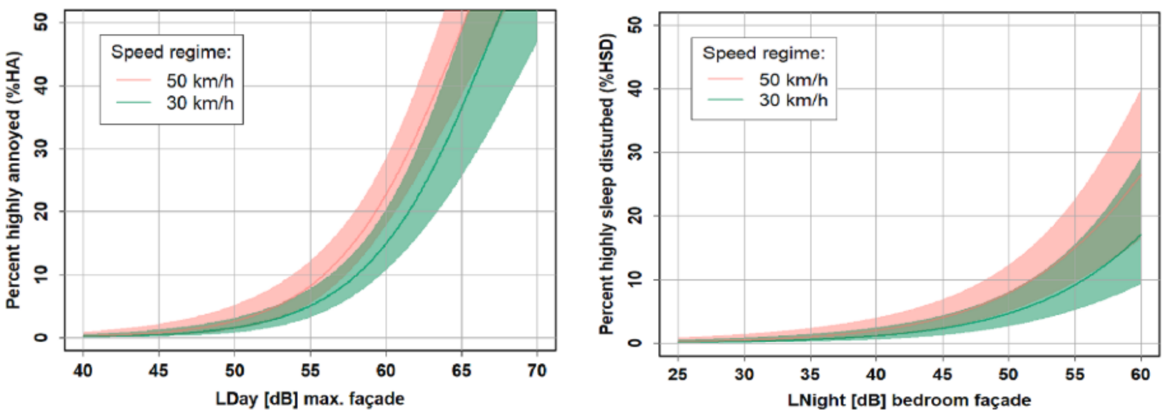



Figure 37 Exposure-response curves for the percentage highly annoyed (%HA, left) and highly sleep disturbed (%HSD, right) during the speed regime 50 km/h and speed regime 30 km/h (after the changeover). Scales used – Left: 5-point ICBEN scale with cut-off point 60 %; Right: 11-point ICBEN-type scale with cut-off point 73 %. Statistical modelling: Multilevel logistic regression with adjustment for age and sex; plotted are the centred curves (at the mean of age and sex) and the 95 % confidence intervals as shaded areas (Brink et al., 2022).

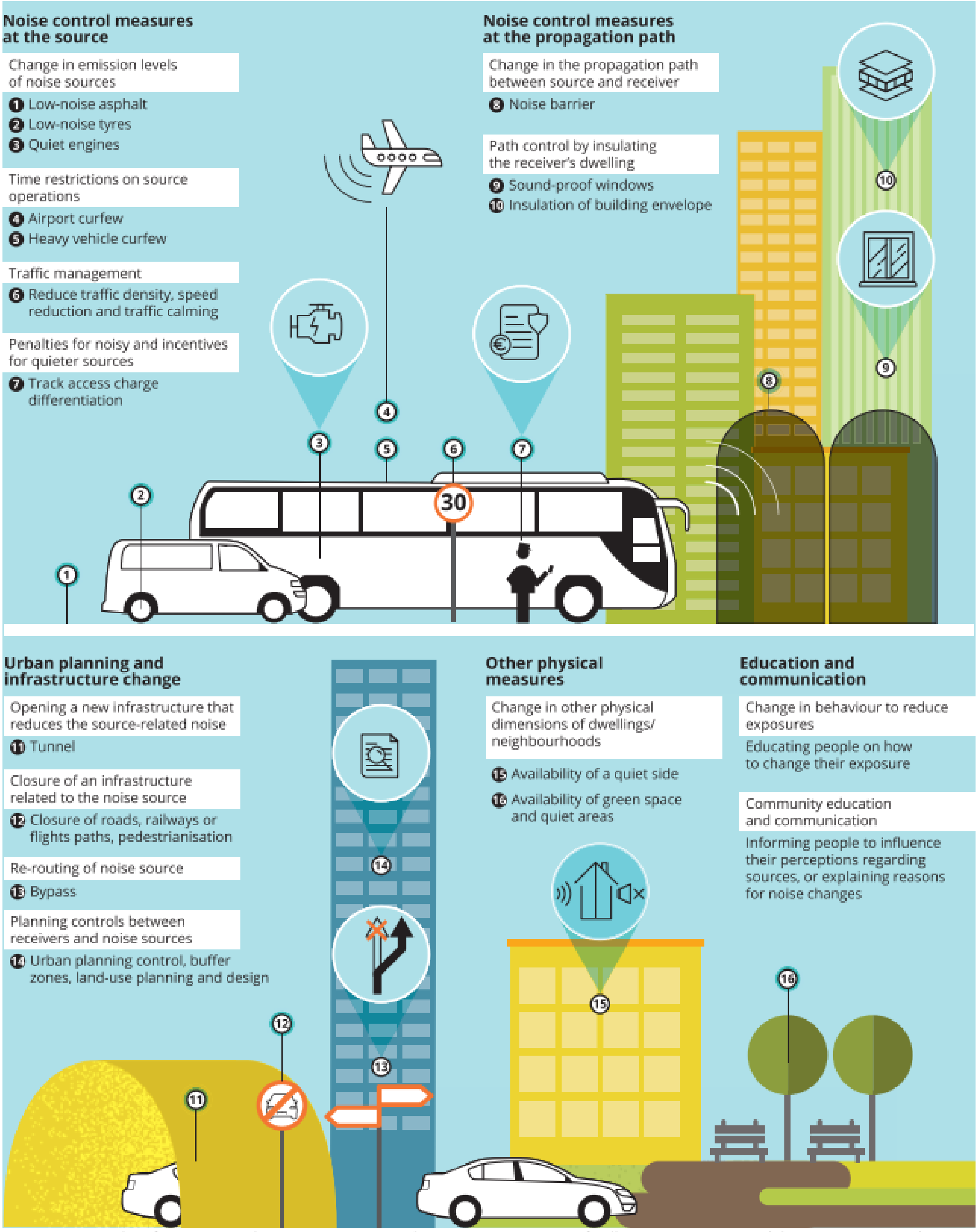
## EEA ADOPTION OF BROWN AND VAN KAMP INTERVENTION FRAMEWORK

The EEA (European Environment Agency, 2020) in *Environmental Noise in Europe - 2020,* adopted and slightly expanded, the framework of environmental noise management and mitigation proposed by Brown and Van Kamp (2017) – see Figure 38 – with five categories of intervention, viz.,

- noise control measures at source:
    - change in emission levels
    - time restrictions on operations
    - traffic management
    - penalties/incentives for different sources
- noise control measures on the propagation path:
    - between source and receiver
    - insulation of receiver's building
- urban planning and infrastructure change:
    - opening new or closing old infrastructure
    - re-routeing
    - planning controls between source and receiver (e.g., buffers)
- other physical measures:
    - quiet side
    - greenery
- education and communication:
    - change behaviour to reduce exposure
    - informing/explaining to alter perceptions.

This framework is shown as a graphic in Figure 38. While there is nothing in this list of measures that would be regarded as novel, what is novel is the categorization of such measures, clearly separating out the different possible action points. This figure could prove useful in the development of material for communication with the public regarding traffic noise management practices.

The utility of the classification above has been illustrated in a comparative survey of noise management strategies across eight European cities (see Van Renterghem et al., 2019).




**Note:** Examples of environmental noise management and mitigation measures are for illustration purposes and don't constitute an exhaustive list of measures.

**Source:** EEA, adapted from Brown and van Kamp (2017).

**Figure 38 Categorisation of intervention (noise management and mitigation) measures (from: European Environment Agency, 2020).**

# INSIGHT INTO THE WHO GUIDELINE DEVELOPMENT PROCESS:

The development of the WHO Environmental Noise Guidelines followed a rigorous methodology. The overall process is shown in Figure 39. Steps 1 and 2 in the figure - undertaken by the Systematic Review Teams (see Figure 8 for the Systematic Review Team authors) with review by the External Review Group – have been discussed extensively in preceding sections of this document. It is Step 3, 'Developing recommendations' by the WHO Guideline Development Group (GDG), that are examined here. This is where critical decisions were made concerning the WHO exposure guideline values.

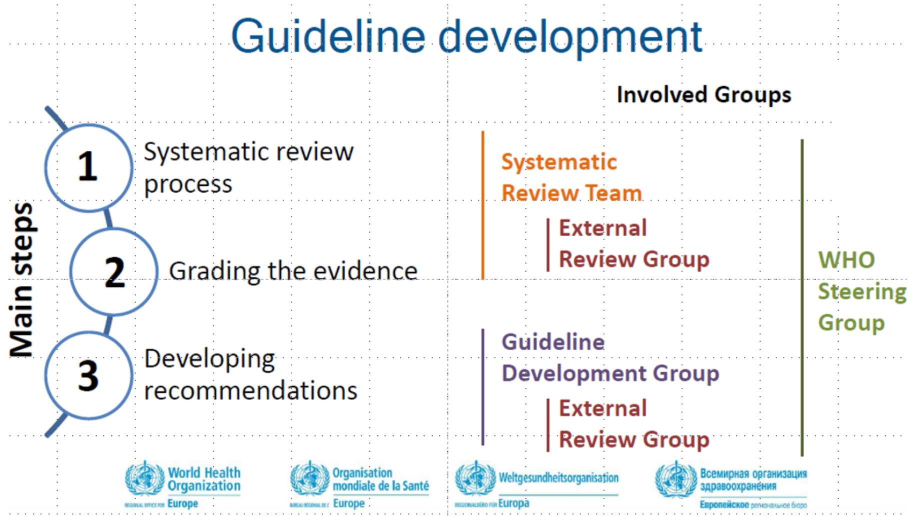



**Figure 39 Slide from Stephen Stansfeld, Chair of the WHO Guideline Development Group.**

The actual deliberations of the GDG are not recorded, but World Health Organization (2018) notes the following (paragraph 2.4.3)[53]. The objective was to set the guideline for road traffic ***at a level above which the GDG was confident that there is an increased risk of adverse health effects***. This required the GDG to (for road traffic noise):

1. assess the validity of ERFs from the systematic reviews for critical health outcomes (annoyance, sleep, cardiovascular and metabolic effects)
2. assess the lowest noise level measured in the studies examined in the systematic reviews
3. assess the smallest risk/relative risk (RR) increase in the ERF for each outcome
4. identify a level, ***above the lowest level (from 2. above) and where the ERF shows the smallest relevant risk increase (from 3. above)*** for ***each*** outcome.

This last activity (4.) is likely to result in a different level from each of the critical health outcomes (incidence of IHD, %HSD. incidence of hypertension, %HA).

5. Influenced by Disability Weights (DW) for each of these outcomes (DW represents agreed seriousness of the health outcome), and examination of each of the ERFs, the GDG then made a judgement from the various levels identified in 4. to specify **a single level** "***above which the GDG was confident that there is an increased risk of adverse health effects"***

For road traffic and sleep, this became the guideline level 45 dB $L_{night}$.
For road traffic and effects other than sleep, this became the guideline level 53 dB $L_{den}$.

This elaboration of the work undertaken demonstrates the difficulties, and risks, faced by the GDG associated with making judgements on appropriate guideline values. Secondly, it emphasizes, again, that the WHO guideline for $L_{den}$, was not based on determining a level at which a particular percentage of the population is highly annoyed, but instead on a level providing a much more multi-dimensional protection of health.

[53] The original has been paraphrased for this scoping review, and exclusive of noise sources other than road traffic.

## FURTHER CONSIDERATION OF THE GUIDELINES

Policy/regulatory authorities may wish to consider the following in making their judgements with respect to deciding whether, and if so what, to adopt from the WHO Environmental Noise Guidelines.

As development of the WHO Guidelines followed a rigorous methodology, WHO can note that they thus provide the scientific evidence on which national noise limits might be based. They believe that, for technical experts and decision-makers:

> *'…the guidelines can be used to provide exposure–response relationships that give insight into the consequences of certain regulations or standards on the associated health effects. They also can be useful at the national and international level when developing noise limits or standards, as they provide the scientific basis to identify the levels at which environmental noise causes a significant health impact. Based on these recommendations, national governments and international organizations can be better informed when introducing noise limits, to ensure protection of people's health.'*

In general, the criticisms of the Guidelines that have occurred since publication have largely been of the selection of studies used in some of the systematic reviews – that of annoyance in particular – and of the recommended guideline levels themselves (*viz*., that these were lower than previously recommended limit levels). The various commentaries on these issues, for road traffic noise, have been documented in earlier sections of this scoping review. These aside, there appears to be otherwise general acceptance of the overall WHO approach, the critical outcomes examined, and the scientific rigour of the exposure-response functions for each outcome that have been reported (the ERF reported for percentage highly annoyed may be the exception). In other words, the ERFs in the guidelines provide a good evidence base, at least for informed debate, regarding the health consequences associated with the setting of specific road traffic noise limits.

Additional studies and syntheses have continued to become available since the publication of the WHO ENGs. Again, these have been reported in earlier sections of this document, generally increasing confidence in the evidence for the various ERFs.

The WHO makes it clear that its guideline values should be regarded as *recommendations* only, and that any jurisdiction's policy decisions on noise limits for standards or legislation will have to also take other considerations into account. Its statement on this is *(World Health Organization, 2018, p. 105)*:

> *'The WHO guideline values are evidence-based public health-oriented recommendations. As such, they are recommended to serve as the basis for a policy-making process in which policy options are considered. In the policy decisions on reference values, such as noise limits for a possible standard or legislation, additional considerations – such as feasibility, costs, preferences and so on – feature in and can influence the ultimate value chosen as a noise limit.'.*

While the generic process followed in the ENG guideline development is listed (1. to 5.) in the previous section, it is unfortunate that the exact workings of the GDG – its discussions and decision practices in determining guideline values - are not unambiguously documented in World Health Organization (2018). One option for a policy/regulatory authority is to attempt its own 'replication' of GDG processes in determination of a guideline level for $L_{den}$. In doing this it would be appropriate to now use the ***updated*** ERFs of the WHO systematic reviews and, possibly, add an ERF for dementia. This would entail:

a. Examine ERF of reanalysis of %HA (Figure 16)
b. Examine ERFs of update of CV effects (Figures 26, 27 and 28)
c. Examine new ERF for dementia (Figure 32)

It is quite appropriate for an authority to undertake such a process, and form its own judgement, taking all local circumstances into account, as to whether the WHO recommended $L_{den}$ guideline of 53 dB has merit as the level of road traffic noise "***above which (one could be) confident that there is an increased risk of adverse health effects"*** of traffic noise exposure – and, if not, to adopt a different limit value.

# THE enHEALTH REVIEW

This review (enHEALTH, 2018) updates and revises a 2004 enHEALTH Australia report on the non–auditory effects of environmental noise. It evaluates some 200 research papers, publications and policies that were published in the period January 1994 to March 2014. It included a systematic review of international evidence on the influence of environmental noise on sleep, cardiovascular disease, and cognitive outcomes. Unlike the WHO review, enHEALTH (2018) did not include annoyance as a separate health outcome, choosing to consider it a mediating factor between environmental noise exposure and health outcomes. Outside of the acoustic community, there remains an opinion, within some health circles, that annoyance should not be considered a heath effect. Underlying this is the fact that there is no ICD code[54] for annoyance – nor for sleep disturbance[55].

The timing of the enHEALTH review was, perhaps, unfortunate vis-à-vis the reviews conducted for the WHO revision of its ENGs – occurring largely simultaneously, and independently.

Within the review objectives, the primary research question was:

> ***'What is the evidence for an effect of environmental noise on sleep, cardiovascular and cognitive outcomes?'***

Four sub-questions were:

1. Is there a dose–response relationship between environmental noise and sleep, cardiovascular and cognitive outcomes?
2. Is there any evidence that certain populations, such as children, are particularly vulnerable to the effects of environmental noise on sleep, cardiovascular and cognitive outcomes?
3. Does the association between environmental noise and sleep, cardiovascular or cognitive outcomes vary by noise source, such as rail, road, and aircraft?
4. Is there any evidence that annoyance is a mediator linking environmental noise exposure to sleep, cardiovascular and cognitive outcomes?

It is useful to first report, verbatim, the Summary of Findings:

- *There is sufficient evidence of a causal relationship between environmental noise and both sleep disturbance and cardiovascular disease to warrant health-based limits for residential land uses:*
    - *During the night-time, an evidence based limit of 55 dB(A) at the facade using the $L_{eq,night,}$ or similar metric and eight-hour night-time period is suggested.*
    - *During the day-time, an evidence based limit of 60 dB(A) outside measured using the $L_{eq,day}$, or similar metric and a 16 hour day-time period is suggested.*
- *There is some evidence that environmental noise is associated with poorer cognitive performance. However, findings were mixed, and this relationship requires further investigation.*
- *It is plausible that aircraft, rail, and road traffic noise have differential effects on sleep quality and cardiovascular health, but the evidence is not conclusive.*
- *It is possible that health impacts may be greater among certain vulnerable groups, but further investigation is needed before making conclusions.*
- *Research on the health impacts of environmental noise in the Australian context should be a priority. There is a particular lack of research on environmental noise exposure and health impacts in rural areas. Intervention studies examining the effects of change in noise exposure on changes in population health are also needed.*

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[54] ICD-10 is the 10th revision of the International Statistical Classification of Diseases and Related Health Problems (ICD), a medical classification list by the World Health Organization. Global Burden of Disease calculations (see later in this report) do not accept any health outcome that is not ICD-coded.

[55] There is, however, an ICD code for insomnia (see the latter part of the section *'Sleep')*.

Focussing on the evidence regarding thresholds, the document indicates:

for sleep disturbance:

| | |
|---|---|
| **Threshold** | There is consistency across higher quality studies to suggest sleep disturbance above 55 dB(A) ($L_{night,outside}$) at the façade. Some studies show physiological effects below 55 dB(A) ($L_{night,outside}$) but because of the studies' limitations, the evidence was not sufficient to say when these outcomes constitute an adverse health effect. |

for cardiovascular effects:

| | |
|---|---|
| **Threshold** | The larger studies that more comprehensively controlled for confounders suggested adverse effect on the cardiovascular system occur above 60 dB $L_{Aeq,day,16h}$ at the façade. Note that the $L_{Aeq,day,16h}$ metric measures sound from 7 am to 11 pm and is an outdoor value.<br>Given the variability in research designs and study quality, summary threshold effects could not be determined from the studies. Some studies offer findings that indicate levels at which adverse outcomes are observed, although these do not indicate clear thresholds. |

for cognition:

| | |
|---|---|
| **Threshold** | The systematic review did not provide a clear indication of a threshold but it suggested there may be distinct threshold effects for different cognitive outcomes. |

The enHealth review was conducted according to appropriate principles in terms of locating relevant international material for the 20-year period 1994-2014, selecting studies to be included (inclusion criteria are provided) and estimating quality of the studies according to GRADE and NHMRC guidelines/recommendations and providing GRADE estimates of quality of evidence of effect overall.

However, this author is concerned that in their use of noise metrics in the studies reviewed and/or in recommendations made, the enHEALTH authors may not have adequately distinguished between the European practice of exposure levels being free field ('at the façade') and the Australian practice of reporting exposures 1m in front of the façade, thus including the reflection in the measurement. They use the term 'at the facade' in the threshold boxes above for sleep disturbance and cardiovascular effects. While the enHEALTH authors indicate in their text that they are aware of this distinction[56], concerns remain because the measurement location/condition is not unambiguously defined in the enHEALTH document – even where one would expect to find them defined (as in Table 1-2: Common Noise Descriptors). Terms such as 'at the façade' and 'is an outdoor value' that are used in the enHEALTH document are problematical and carry the potential of 2.5/3dB error in any level specified – including in suggested thresholds. Determining whether this author's concern is appropriate or not would require reconstructing all analyses undertaken.

With the above comments in mind, enHEALTH (2018) is a useful document that confirms that road traffic noise exposure has a measurable effect on sleep disturbance and cardiovascular health, and that thresholds for physiological effects can be identified – but there is the above reservation around accepting the particular threshold levels suggested.

[56] enHEALTH (2018) p.31 does note the existence of this difference: '*Similarly, within the studies it is important to distinguish between façade noise levels, often used in Australia and France, and the free field noise levels often used in other countries. Free field noise levels account only for noise coming from a source. Façade levels account for both noise coming from a source and noise reflected back from a façade. A façade level is typically 2.5 to 3.0 dB higher than the corresponding free field*'. Despite this, the question remains as to whether this distinction was applied in all appropriate circumstances.

# BURDEN OF DISEASE AND HEALTH IMPACT ASSESSMENT

**S**everal topics below are inter-connected. Deciding the order in which to present them has been problematic.

**Burden of Disease** (BOD), and related **Health Impact Assessment** (HIA), for road traffic noise and interventions are described below – as a follow-on applications of the previous sections on health effects of road traffic noise. BoD was also introduced earlier in the section '*Changing Understanding of Effects of Road Traffic Noise'* (Hanninen et al., 2014) where it was termed EBD (environmental burden of disease) and results reported from the EBoDE project (Environmental Burden of Disease in European countries) – See Figure 5.

However, a critical component of both BOD and HIA is determination of exposure of the general population to road traffic noise in their homes. Rarely are these exposures measured. Instead, estimates are obtained by strategic road traffic noise level prediction techniques. Prediction modelling for BOD and HIA is referred to in this section, but there is also a separate section '*Road Traffic Noise Prediction Model*s' later in this scoping document that also includes such strategic noise modelling. Strategic traffic noise prediction modelling is applied in epidemiological studies of the health effects of road traffic noise. Several epidemiological studies of health that used strategic noise mapping to estimate road traffic noise exposures – viz., Vienneau et al. (2022) for CVD mortality; Thompson et al. (2022) for children's cognitive effects; and Cantuaria et al. (2021) for dementia - have already been examined in the earlier health effects sections.

Some initial studies of BOD and strategic noise mapping in Australia are also introduced here – though these, too, could equally have been reported in the later '*Road Traffic Noise Prediction Model*s' section.

Initial work on the disease burden of environmental noise was reported in Fritschi et al. (2011) based on a committee of experts assembled by the WHO – see Figure 40. A Burden of Disease calculation, for any specific disease, estimates deaths from that disease in terms of 'years of life lost' together with the number of years lived with a disability as a result of that disease (weighted for the severity of living with that disease). Burden can be measured in the units termed 'DALYs' – Disability Adjusted Life Years. The equations below for calculating DALYs are from Fritschi et al. (2011):

The burden of disease is expressed in DALYs in the general population through the equation

$$DALY = YLL + YLD$$

In this equation, YLL is the number of "years of life lost" calculated by

$$YLL = \sum_i (N_i^m \cdot L_i^m + N_i^f \cdot L_i^f)$$

where $N_i^m(N_i^f)$ is the number of deaths of males (females) in age group i multiplied by the standard life expectancy $L_i^m(L_i^f)$ of males (females) at the age at which death occurs. YLD is the number of "years lived with disability" estimated by the equation

$$YLD = I \cdot DW \cdot D$$

where I is the number of incident cases multiplied by a disability weight (DW) and an average duration D of disability in years. DW is associated with each health condition and lies on a scale between 0 (indicating the health condition is equivalent to full health) and 1 (indicating the health condition is equivalent to death).

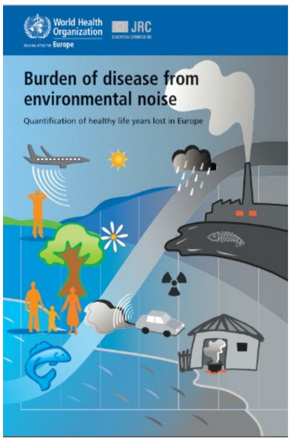



**Figure 40 Fritschi et al. (2011)**

The purpose of BOD calculations is largely to assist policy makers to quantitatively compare the burden resulting from different diseases, or in comparing the burden in different localities or sub-populations. An example has already been included in the earlier section: '*Changing Understanding of Effects of Road Traffic Noise*', where it was reported *(Hanninen et al., 2014)* that road traffic noise is considered the second most prevalent (in terms of DALYs) environmental risk factor, after fine particle pollution, to human health in Europe. Estimates of DALYs are best considered primarily as a comparative measure rather than an absolute one – some epidemiological colleagues even suggest that that a single, absolute DALY measure is of very little use.

| Health endpoint | Noise in Europe 2014 (EEA, 2014) | Gap filled data-set with 473 agglomerations | |
|---|---|---|---|
| | | Exposure >55 dB $L_{den}$ & >50 dB $L_{night}$ | Imputed full distribution & extended exposure-response |
| Annoyance (*million) | 10.9 | 21.5 | 29.5 |
| Severe annoyance (*million) | 5.0 | 10.0 | 12.5 |
| Sleep disturbance (*million) | 4.7 | 9.2 | 17.2 |
| Severe sleep disturbance (*million) | 2.2 | 4.3 | 7.3 |
| Hypertension (*million) | 0.49 | 1.0 | 1.1 |
| Hospital admissions (*thousand) | 25.1 | 47.4 | 52.6 |
| Premature mortality (*thousand) | 5.7 | 11.0 | 12.2 |

**Table 15 Results (Van Beek et al., 2015) of the health impact assessment for road traffic noise in agglomerations in Europe for 177 million residents: estimates of DALYs from road traffic noise for different health outcomes/endpoints compared to 2014 DALY estimates.**

**See the source document for specification of the health endpoints. The middle column of the table shows calculated DALYs with left-truncated ERFs; the right column uses the full ERF range.**

'Gap-filling' refers to accounting for those agglomerations/countries where required END data on road traffic noise exposure was not available.

However, absolute DALYs for road traffic noise have been estimated – *vid.* those calculated for Europe in a paper (Van Beek et al., 2015) entitled "*Towards an complete Health Impact Assessment for Noise in Europe*". These are shown in Table 15. There are other examples of DALY calculations for road traffic noise: e.g., Hegewald et al. (2021) reported cardiovascular disease, depressive disorders, annoyance, and disturbed sleep due to road-traffic noise cost the roughly 6 million residents of Hesse 26,501 years of healthy life (DALY) in 2015. This equates to about 4.3 years of healthy life lost per 1000 persons (population of Hesse, Germany, in 2015 = 6,093,888). Veber et al. (2022) provide another example by application to two Estonian cities: Tallin and Tartu. Each of these different applications provides increasing insight into the issues, sensitivities, and limitations, in estimating burden of disease for road traffic noise,

More recent estimates for Europe have been reported (European Environment Agency, 2020) both in terms of numbers suffering from various health outcomes from environmental noise for each noise source separately (Table 16) and, estimation of the burden of disease (BoD) in DALYs (Table 17) due to environmental noise for sources covered by the END

**Table 16 Estimated number of people suffering from various health outcomes due to environmental noise in 2017, EEA-33 - Turkey not included (European Environment Agency, 2020).**

| | | High annoyance | High sleep disturbance | Ischaemic heart disease | Premature mortality ([a]) | Cognitive impairment in children |
|---|---|---|---|---|---|---|
| **Inside urban areas** | Road | 12 525 000 | 3 242 400 | 29 500 | 7 600 | |
| | Rail | 1 694 700 | 795 500 | 3 100 | 800 | |
| | Air | 848 300 | 168 500 | 700 | 200 | 9 500 |
| | Industry | 87 200 | 23 400 | 200 | 50 | |
| **Outside urban areas** | Road | 4 625 500 | 1 201 000 | 10 900 | 2 500 | |
| | Rail | 1 802 400 | 962 900 | 3 400 | 900 | |
| | Air | 285 400 | 82 900 | 200 | 50 | 2 900 |
| | **Total ([b])** | **21 868 500** | **6 476 600** | **48 000** | **12 100** | **12 400** |

**Notes:** ([a]) Refers to mortality due to ischaemic heart disease.

([b]) There may be double counting for annoyance and sleep disturbance because of the combined effects of multiple sources. It is estimated to be no more than 13 % for annoyance and 16 % for sleep disturbance. Double counting for ischaemic heart disease and mortality is estimated to be negligible (ETC/ACM, 2018)

**Table 17 Estimation of the burden of disease (BoD) due to environmental noise for sources covered by the END, EEA-33 - Turkey not included (European Environment Agency, 2020).**

| Health effect | Public health impact (DALYs/year) and (DALYs/year) per million ([a]) | Health effects included in different approaches to estimating noise BoD | | |
|---|---|---|---|---|
| | | WHO and JRC (2011) | Hänninen et al. (2014) | IHME (2018) |
| High annoyance ([b]) | 453 000<br>900 per million people | ✓ | x | x |
| High sleep disturbance ([b]) | 437 000<br>800 per million people | ✓ | ✓ | x |
| Ischaemic heart disease ([c]) | 156 000<br>300 per million people | ✓ | ✓ | ✓ |
| Cognitive impairment in children ([d]) | 75<br>~0 | ✓ | x | x |

**Notes:** ([a]) The DWs used for the calculation of DALYs are those indicated in WHO (2018). Other sources of information suggest using smaller DWs for annoyance, sleep disturbance and reading impairment (van Kamp, 2018).

([b]) There may be double counting because of the combined effects of multiple sources. It is estimated to be no more than 13 % for annoyance and 16 % for sleep disturbance (ETC/ACM, 2018).

([c]) Includes incidence and mortality.

([d]) Impairment is calculated only for aircraft noise.

Khomenko et al. (2022) have also looked at the BOD from road traffic noise in Europe. The difference in their work is that they focussed in detail on the quality of the strategic noise maps available through the END or from local sources (724 cities and 25 greater cities in 25 European countries). The noise maps were categorized as high, moderate and low quality following a qualitative approach, finding that the majority of noise maps (i.e., 83.2%) were considered of moderate or low quality. Only two health outcomes, high annoyance and ischemic heart disease (IHD), using mortality from IHD causes as indicator, were utilized, using Exposure Response Functions retrieved from the literature. Based on the data provided, almost 60 million adults were exposed to road traffic noise levels above 55 dB $L_{den}$, equating to a median of 42% (Interquartile Range (IQR): 31.8–64.8) of the adult population across the analysed cities. They estimated that approximately 11 million adults were highly annoyed by road traffic noise and that 3608 deaths from IHD (95% CI: 843–6266) could be prevented annually with compliance of the WHO recommendation. The proportion of highly annoyed adults by city had a median value of 7.6% (IQR: 5.6–11.8) across the analysed cities, while the number preventable deaths had a median of 2.2 deaths per 100,000 (IQR: 1.4–3.1).

Salomons and Dittrich (2022) take such work even further, analysing effects in the EU over the period 2020-2035 – allowing for traffic growth and fleet development over that period. Health burden is estimated, for the 334 million population, by four different measures: %HA, %HSD, DALYs, Monetized health burden as shown in Table 18.

After establishing the '2035 baseline burden', Salomons and Dittrich (2022) proceeded to evaluate how a diverse set of traffic noise mitigation strategies might influence this BOD over the next decade and a half. The very detailed set of strategy scenarios included in their study is listed in Table 19.

**Table 18 Annual EU health burden of road traffic noise for the baseline scenario in 2030 (Salomons & Dittrich, 2022).**

| | Annual value in 2030 |
|---|---|
| Highly annoyed persons: | 31.2 million |
| Highly sleep-disturbed persons: | 14.6 million |
| DALYs: | 1669 thousand |
| Monetized health burden (method 1 / 2): | 58.4 / 14.6 billion Euro |

**Table 19 Scenarios noise solutions in EU2020–2035 evaluated for effect on BOD from road traffic noise across Europe (Salomons & Dittrich, 2022). In all cases, a final situation in 2035 (2024 for scenario B) is specified. For intermediate years, linear interpolation is applied.**

| | |
|---|---|
| A | **Quiet roads**. The fractions of roads with a quiet surface are increased, for road types 5–8 (thin top layers for types 5–6, porous asphalt for types 7–8). The length percentages are 22.5% in 2035, which is a factor of 4.5 higher than the baseline value of 5%. |
| B | **Quiet tires**. The tire labels for the three vehicle types C1/C2/C3 are gradually decreased from 70/72/75 (baseline) to 66/69/70 in the period 2020–2024 and remain constant after 2024. |
| C | **Vehicle limits**. Vehicles comply faster with new vehicle noise emission limits. For the year 2035, the percentages for the six vehicle-emission categories 2015/2016/2022–24/2024–26/hybrid/electric are changed as follows:<br>– cars (C1): from 15/15/30/10/9/21% (baseline) to 2/2/26/40/9/21%<br>– vans (C2): from 15/15/35/14/9/12% (baseline) to 2/2/30/45/9/12%<br>– buses (C3): from 15/15/25/7.5/10.5/27% (baseline) to 2/2/20/38.5/10.5/27%<br>– lorries (C3): from 15/20/30/9.5/24/1.5% (baseline) to 5/5/27.5/37/24/1.5%<br>– heavy trucks (C3): from 15/20/30/9.5/24/1.5 (baseline) to 5/5/27.5/37/24/1.5%. |
| D | **Electrification**. Electrification is enhanced, with more hybrid and electric vehicles in 2035. The compliance percentages for the six vehicle-emission categories 2015/2016/2022–24/2024–26/hybrid/electric in 2035 are changed as follows:<br>– cars (C1): from 15/15/30/10/9/21% (baseline) to 10/10/25/5/19/31%<br>– vans (C2): from 15/15/35/14/9/12% (baseline) to 10/10/30/9/19/22%<br>– buses (C3): from 15/15/25/7.5/10.5/27% (baseline) to 10/10/20/2.5/20.5/37%<br>– lorries (C3): from 15/20/30/9.5/24/1.5% (baseline) to 20/15/25/4.5/34/11.5%<br>– heavy trucks (C3): from 15/20/30/9.5/24/1.5 (baseline) to 20/15/25/4.5/34/11.5%. |
| E | **Barriers**. The fractions of roads with noise barriers are increased, for road types 5–8. The length percentages are 12.5% in 2035, which is a factor of 2.5 higher than the baseline value of 5%. |
| F | **Speed restriction**. Vehicle speeds in all urban areas are reduced. The vehicle speeds in 2035 are changed as follows.<br>– road types 1–4: from 30–50 (baseline) to 30 km//h<br>– road type 5 (main road): from 70–80 (baseline) to 50 km/h<br>– road type 6 (motorway):from 85–115 (baseline) to 80 km/h. |
| G | **Car-free zones**. New car-free zones in urban areas are created by means of traffic access restrictions and traffic rerouting. In 2035, there is a total of 2500 $km^2$ new car-free area in EU cities (e.g., 250 zones of 10 $km^2$); this is about 2.5% of the total area of 400 END cities (average about 250 $km^2$ per city). |
| H | **Quiet facades**. More quiet facades of dwellings are created. It is assumed that 30% of the dwellings in urban area that have no quiet facade in 2020 will have a quiet facade in 2035. |
| I | **Dwelling insulation**. More dwellings are insulated. The percentage of dwellings with insulation is increased by 10% in 2035 (compared with baseline), for road types 5–8. |
| J | **Reception limits**. Reception limits are introduced: 60 dB for Lden and 55 dB for Lnight in 2035. |
| ABC | **Combined scenario**. Combination of scenarios A, B, and C. |
| ABCD | **Combined scenario**. Combination of scenarios A, B, C, and D.<br>The compliance percentages for the six limits in 2035 are changed as follows:<br>– cars (C1): from 15/15/30/10/9/21% (baseline) to 0/0/18/32/19/31%<br>– vans (C2): from 15/15/35/14/9/12% (baseline) to 0/0/22/37/19/22%<br>– buses (C3): from 15/15/25/7.5/10.5/27% (baseline) to 0/0/12/30.5/20.5/37%<br>– lorries (C3): from 15/20/30/9.5/24/1.5% (baseline) to 0/0/22.5/32/34/11.5%<br>– heavy trucks (C3): from 15/20/30/9.5/24/1.5 (baseline) to 0/0/22.5/30/34/11.5%. |
| FGHI | **Combined scenario**. Combination of scenarios F, G, H, and I. |

The original publication should be examined for details of methods applied in scenario testing, including future predictions based on application of the CNOSSOS prediction model applied to input data reflecting each scenario being tested. It also should be examined for details regarding the economic modelling of the costs and benefits of scenarios (including two different methods of calculating the benefit-cost ratio of each scenario).

The conclusion is that for road traffic noise, EU health burden reductions in 2030 of 15–20% can be achieved with combined scenarios. For ABC and ABCD, the reduction is largely the effect of quiet tires (B). The reduction for single solution scenario B (13%) is much larger than scenarios A (quiet roads), C (vehicle limits), and D (electrification). The benefit-cost ratio for scenarios ABC and ABCD is in the range 1–5, the benefits are a factor of 1–5 higher than the costs. For scenario FGHI, the reduction is largely due to the effect of speed restriction (F). The effects of car-free zones (G), quiet facades (H) and dwelling insulation (I) are considerably smaller. The authors claim to have undertaken detailed modelling of all elements of the causal chain from noise exposure to health effects. They note that all results presented are presented at EU level but warn results at local levels may differ from the results at EU level.

The current author has not been able to attempt to examine/verify the details of inputs, data, models, and assumptions used in this work.  Further, the current author suspects that much of the changes in burden associated with the scenarios may be occurring at levels on the exposure-response functions below the criteria recommended by WHO. It would be useful to assess how much of these ‘global’ changes in burden is amongst those experiencing the highest exposure.

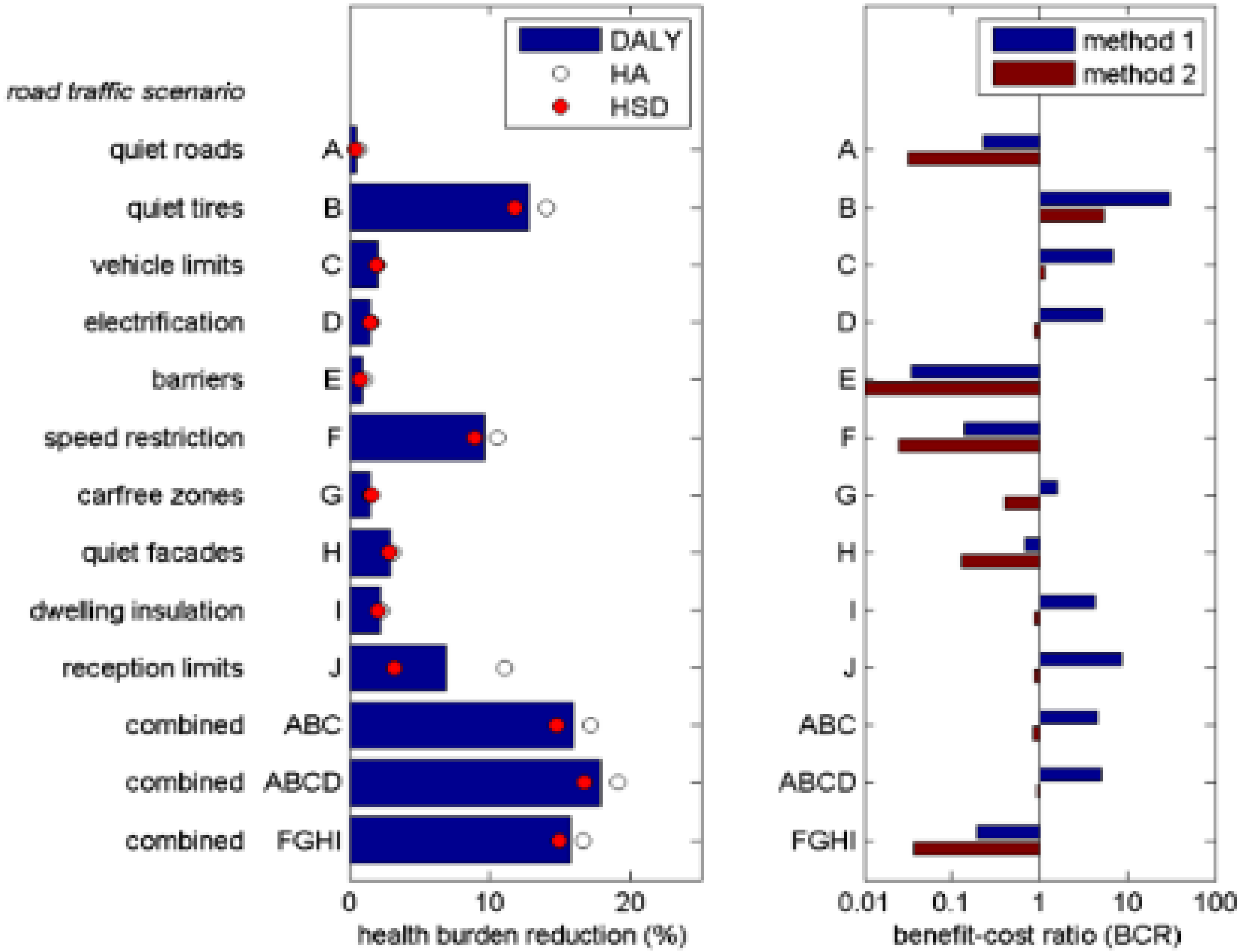



**Figure 41 Results of calculation for the thirteen road traffic noise mitigation scenarios (scenarios listed in previous table): whole of EU health burden reduction in 2030 (left) and benefit-cost ratio over 2020–2035 (right) (Salomons & Dittrich, 2022).**

## STUDY ON METHODOLOGY TO PERFORM AN ENVIRONMENTAL NOISE AND HEALTH ASSESSMENT-A GUIDANCE DOCUMENT FOR LOCAL AUTHORITIES IN EUROPE.

The Dutch RIVM was commissioned by the European Commission to provide guidance on the issues, methodology and application of these technique for burden of disease from environmental noise, by non-specialist users – nominally local and regional authorities. The document by van Kamp et al. (2018) describes the steps of a health impact assessment, one by one, and explains the accompanying decisions and required conditions. The actual calculation methods are further explained for two indicators: the number of healthy life years adjusted for disease, disability, and death (DALY) and the number of people that experiences adverse effects of noise ((NafP). This is a 110+ page document, which will not be examined further here, but it should be by looked at by any organization wishing to calculate the BOD for road traffic noise. Usefully, two aspects of the van Kamp et al. (2018) work were summarized at the Acoustics 2019 conference in Cape Schanck (van Kamp, 2019). These are reported below in full as they provide important guidance for conducting an HIA for road traffic noise, and example results of an HIA on road traffic noise interventions. The outline of the steps in undertaking a HIA for environmental noise is shown in Figure 42, and the results of the practical application of HIA to a simple road traffic noise reduction programme (other transport noise sources were also present) in Dusseldorf, Germany is reported in Figure 43, and in the Box, Figure 44.

Th following are reported , verbatim from van Kamp (2019):

> ***"Practical example of HIA*** *The actual process of quantifying the burden of disease discerns five steps: 1) selection of health outcomes 2) the assessment of population exposure 3) the identification of exposure response relations 4) the estimation of the number of cases and 5) the calculation of the disease burden. Following these steps, the Environment Agency of Düsseldorf (EAD), in Germany, calculated the anticipated impact of ground transportation noise interventions, including predictions of the change in the number of exposed residents for two road traffic noise mitigation actions, which will be taken as examples here. These two interventions both consisted in the implementation of noise reducing road surfaces and a speed limit change from 50 km/h to 30 km/h. The underlying main question was: 'What are the impacts of these selected noise actions on health?'*
>
> *"In preparing the quantitative HIA the following input data were obtained:*
>
> *a) The fraction of residents per noise category of the study area. By linking noise maps that were created in the framework of the END for the 3rd round in 2017, to the home addresses of the residents of Dusseldorf, the EAD provided population-exposure distributions. For aircraft, road traffic and railway noise. Numbers of exposed people were available in 1 dB bands for both $L_{den}$ and $L_{night}$. The $L_{den}$ bands ranged from below 40 to 78 dB for road traffic noise and to 90 dB for railway noise. The $L_{night}$ bands included noise levels from below 40 dB to 69 dB for road traffic noise and to 81 dB for railway noise. For aircraft noise data, the EAD provided the number of exposed people in 1 dB bands within the range 45 dB to 75 dB $L_{den}$ and to 66 dB $L_{night}$.*
>
> *b) Demographic data of the study area: Demographic and social distributions were extracted from an official report by the City of Düsseldorf, which included. In particular, data on age groups and gender in 2016. Data per age group was needed since the EERs applied in this assessment had to be applied to the adult population (18 years and older) and to children in the age of 7 to 17 years.*
>
> *c) Disease and mortality data valid for the study area: Since no local disease and mortality data were available, this data was obtained from (inter)national sources. Mortality data for coronary heart disease were extracted from the WHO Mortality database for Germany, the most recent data being from 2015. Hospital discharges (total) was used as the indicator for the incidence of coronary heart diseases (CHD) obtained from the European Hospital Morbidity Database of the WHO office for Germany for the year 2014. It should be noted, that although incidence data for CHD was also available, the calculation was based on hospital discharge rates. Disability-adjusted life years (DALYs) could then be calculated by summing up the years lived with a disease (YLD) with the year of life lost due to premature death (YLL). Both YLL and YLD due to coronary heart disease were derived from the WHO Burden of Disease project for 2016, for which disease and mortality data is referred the whole of Germany. For simplification in this exemplar HIA, the data was assumed to be valid for Düsseldorf.*
>
> *d) Exposure response relations. Part B of the RIVM guidance document (Van Kamp et al., 2018) recommends using local EERs when estimating the number of highly annoyed people and/or the number of highly sleep*

*disturbed people in a study area. Unfortunately, no reliable and valid exposure-response function (ERF) data from Düsseldorf or from elsewhere in Germany that could be applied in this case were available. Therefore, it was decided to apply the generalised ERFs for high annoyance and high sleep disturbance that were recommended and presented in the guidance document.*

*Combining the noise maps with the percentage of adults (18 years and older) obtained from the demographics report of the City of Düsseldorf, the number of exposed persons per 1 dB $L_{den}$ and $L_{night}$ bin was calculated. Figure 43 demonstrates the day and night-time noise exposure levels of people in the city district of Dusseldorf Bilk before and after the intervention (noise reducing road surfaces and speed limit reduction).*

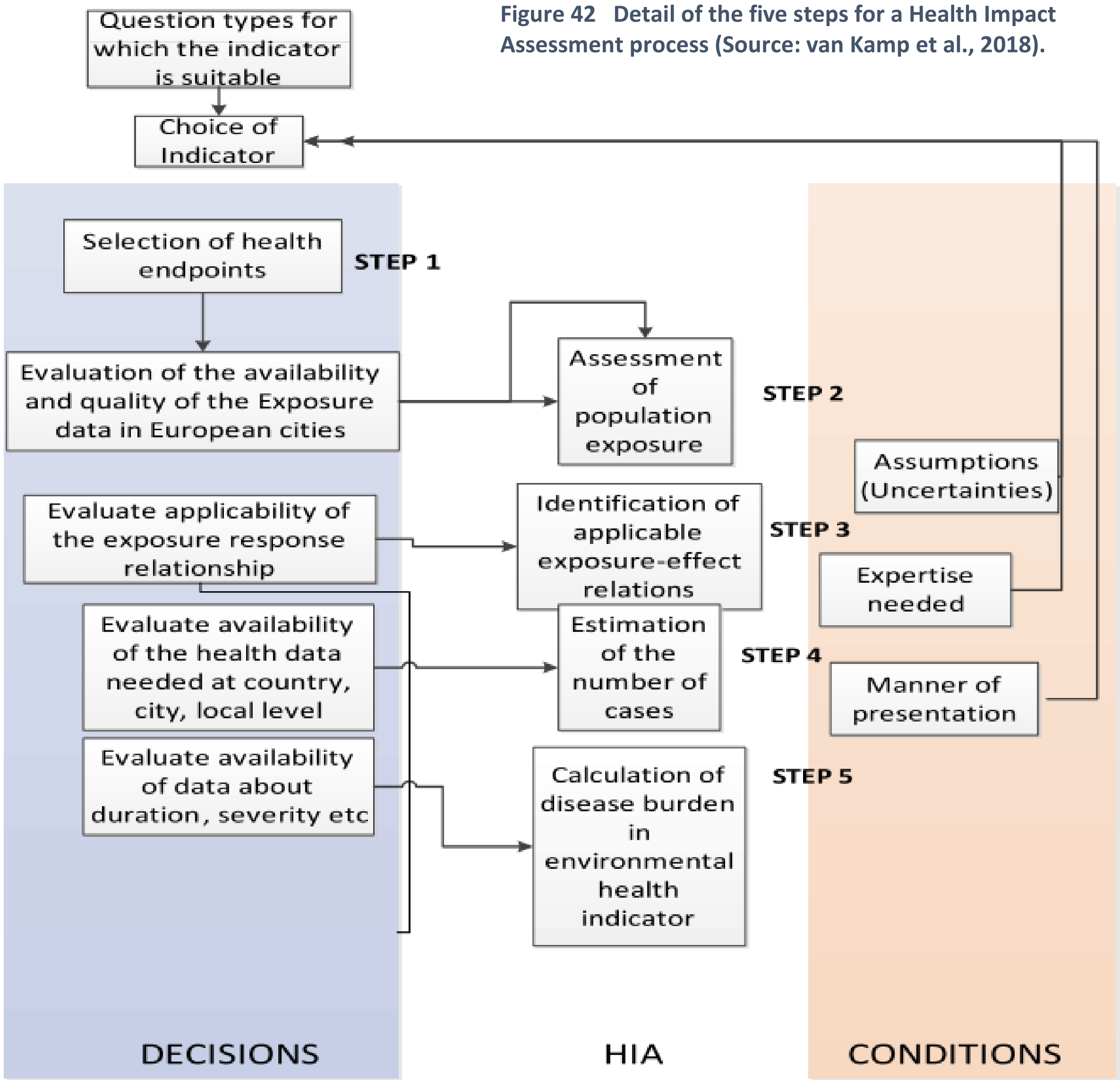
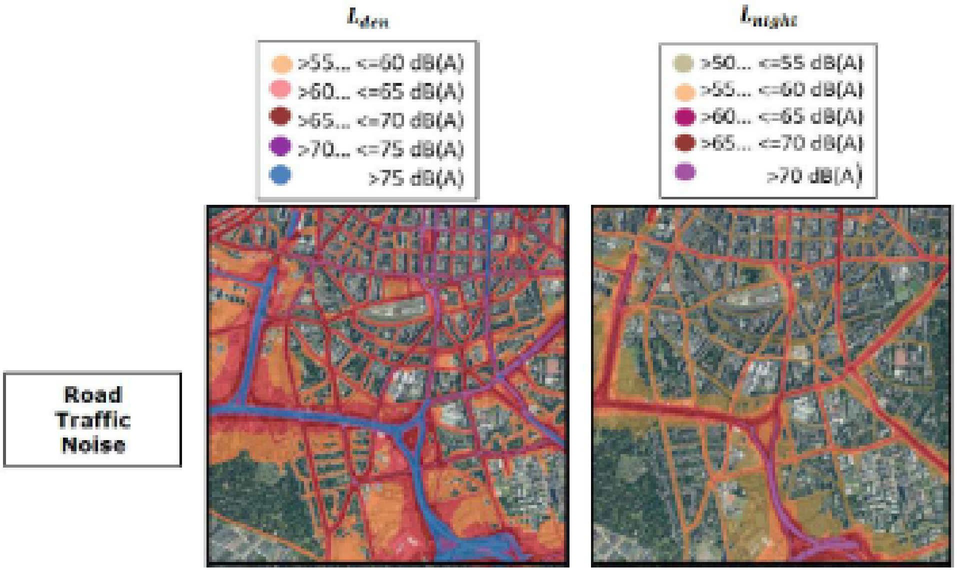



**Figure 42 Detail of the five steps for a Health Impact Assessment process (Source: van Kamp et al., 2018).**

*For the estimation for the second policy question (evaluation of action plans) the HIA assessment of the impact of noise abatement actions was carried out for two simulated interventions: the first addresses road traffic noise with noise-reducing road surfaces and speed limit reduction to 30 km/h (from 50 km/h) and the second consisted in mitigating tram noise. For brevity, only the results of the first intervention is presented here, using two methods of measurement: the number of people affected method (NafP, method 1) and the DALYs (method 2)."*




Source : (Van Kamp et al., 2018)
Figure 4.3: Noise maps for the city district of Düsseldorf Bilk

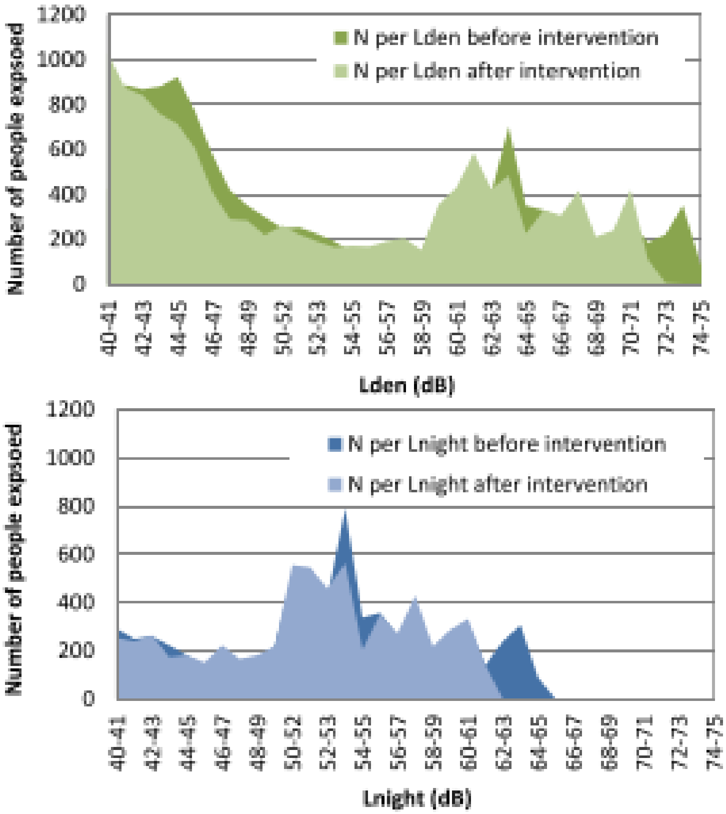



**Figure 43 Noise maps for the city district of Düsseldorf Bilk and Number of adult residents exposed of the city district Düsseldorf-Bilk before and after simulated road surface noise reduction and speed limit reduction interventions (separately for day and for night).**

As a next step, the generalised EERs for the percentage of highly annoyed persons (%HA), the percentage highly sleep disturbed (%HSD) was applied and the Relative Risk (RR) of CHD (incidence and mortality) per unit change in noise exposure level were derived as per the WHO evidence reviews (refer tables 4.1 and 4.2).

Table 4.1: Generalized EER for high annoyance (%HA) in relation to Lden for road traffic

| Noise source | Source-specific exposure-response equation for high annoyance |
|---|---|
| Road traffic | (78.927 – 3.1162*Lden+ 0.0342*Lden2)/100 |

Table 4.2: Relative Risks per 10 dB Lden for the association between road noise and coronary heart disease (EC: ecological study, CC: case-control study, CO: cohort study; RR: relative risk; CHD: coronary heart disease)

| End point | Noise source | RR (95%CI) per 10 dB Lden | Number and design of studies |
|---|---|---|---|
| Incidence CHD | Road | 1.12 (0.85 – 1.48)<br>1.08 (1.01 – 1.15) | 1, EC<br>7, CC CO |
| Mortality CHD | Road | 1.05 (0.97 – 1.13) | 3, CC CO |

The number of affected people (NafP) was calculated separately for each outcome and noise source that is included in the HIA. Further details on the determination of the NafP can be found in to section 3.5 of (van Kamp et al., 2018). The health impact, expressed in the number of people affected by the two interventions, aimed at reducing road traffic noise is shown in table 4.3.

Table 4.3: Theoretical display of number of affected people, NafP (method 1) and DALYs (method 2) before and after noise reducing measures: noise-reducing road surfaces and speed limit restriction to 30 km/h

| Health end Point | Average number of affected people | | | Average DALYs due to road traffic noise | | |
|---|---|---|---|---|---|---|
| | Before intervention | After intervention | Difference (after – before) | Before intervention | After intervention | Difference (after - before) |
| High annoyance | 1490 (11.3%) | 1385 (11.6%) | -106 (0.3%) | 14.90 | 13.85 | -1.06 (7.1%) |
| High sleep disturbance | 289 (2.2%) | 263 (2.0%) | -26 (0.2%) | 6.66 | 6.05 | -0.60 (9.0%) |
| CHD Incidence | 2 | 2 | -0 | 0.38 | 0.34 | -0.04 10.5%) |
| CHD mortality | 8 | 7 | -1 | 9.76 | 8.81 | -0.95 (9.7%) |
| DALY II | -- | -- | -- | 31.70 | 29.06 | -2.65 (8.4%) |

In line with the calculated reduction in noise levels, the health gain of the two interventions is evident

The road traffic intervention causes a reduction of 0.3% (106 persons) in high annoyance and 0.2% in high sleep disturbance. For the two noise abatement actions for road traffic noise, the number of DALY II – defined as DALYs due to CHD, noise annoyance and sleep disturbance – would decrease by 8.4% and one person less would die due to coronary heart disease caused by noise exposure.

**Figure 44 Calculations for the HIA of road traffic noise interventions in Düsseldorf Bilk, showing, in the bottom table, before and after Numbers of People Affected, and before and after DALYs. Both methods of measuring the change show the health gain from each of the interventions (Source: van Kamp, 2019).**

# BURDEN OF DISEASE & STRATEGIC NOISE MAPPING– AUSTRALIA

## BACKGROUND

The notion of the burden of disease from environmental noise has been referred to, but to this author's knowledge, has had limited application in any Australian jurisdiction. It was raised as a concept in VicRoads Noise Policy Review (Vic Roads, 2015) and in the enHEALTH report on the health effects of environmental noise (enHEALTH, 2018).

The Public Health Association of Australia has also sought the following action: '*A **burden of disease relevant to environmental noise pollution** should be established within constraints of current scientific knowledge*' in its *Environmental Noise Policy Position Statement* (Public Health Association Australia, 2020).

Hinze et al. (2022), in Acoustics Australia, published a Technical Note '*Development of Australia-Wide Transportation Noise Maps: An Application in the Estimation of Population Exposure in Victoria*'. In his historical account of noise mapping in Australia, he noted that the first city noise map was of the Adelaide CBD in 2006, and then Central Perth in 2014. Both were for road traffic noise and allowed for estimation of exposures of each building in the study areas.

In 2014, the Greater Melbourne Noise map was completed. Building from the Melbourne Freeways noise map, this model calculated the noise exposure for the day, evening, and night-time periods from road, rail, and industrial noise, representative of the year 2011. Roads within the model included the highways and freeways developed within the Melbourne Freeways noise map in addition to major local roads. Local/minor local roads were not considered. Hinze et al. (2022) references these studies as below (note the '*obtained through freedom of information*' tags, suggesting some difficulties in accessing information).

> Hinze, B.: AECOM. VicRoads. Development and Verification of a Road Traffic Noise Model. Obtained through freedom of information, Melbourne. (2011)
>
> Hinze, B.: WSP Australia, Victoria EPA. Estimation of Environmental Noise Exposure: Greater Melbourne. Obtained through freedom of information, Melbourne. (2014)

## STUDIES

Hinze et al. (2022) reports the building of an "*Australia-Wide Transportation Noise Map*". Three-dimensional road, rail, and aircraft noise models were built for Australia's 537 Local Government Area (LGA) regions using SoundPLAN, an environmental noise modelling software suite from SoundPLAN GmbH. Tallied movements of vehicles from global positioning system devices (known as probe counts) were collated across all road segments (approximately 3.47 million) throughout Australia, spanning a period of 215 weekdays between February and November 2018 … .. This data includes the number of passenger vehicles and their average speed, and the number of fleet/heavy vehicles and their average speed, on each road segment during each hour. A relationship between the probe counts and the road traffic volumes (obtained from commercial and State Road traffic models) was developed with separate relationships for urban and rural areas. Separate relationships were also determined for passenger and heavy vehicles. Approximately 71.6 billion data points were processed to obtain the relevant modelling inputs for Australia's 877,000 km of road network to be modelled. The noise model presented includes over 1.2 million km of road centreline, as many roads were assessed as dual carriageway with two source lines.

Hinze et al. (2022) observes that SoundPLAN implements the German Richtlinien für den Lärmschutz an Straben 1990 (RLS 90) standard …. RLS 90 was chosen because of its ability to calculate an $L_{Aeq}$ for all the time periods required (day, evening, and night). Additionally, SoundPLAN's adaption of RLS 90 complemented the available datasets by enabling the road traffic volumes and speeds for passenger vehicles to be input separately from those of fleet/heavy vehicles. Due to the scale of the project, RLS 90 was also more efficient regarding calculation time. CNOSSOS-EU may be considered in future works. Figure 45 provides a sample, with road traffic noise levels layered with digital elevation models and the location, footprint, and height of over 15 million buildings across Australia was sourced from the PSMA Australia GeoScape© dataset.

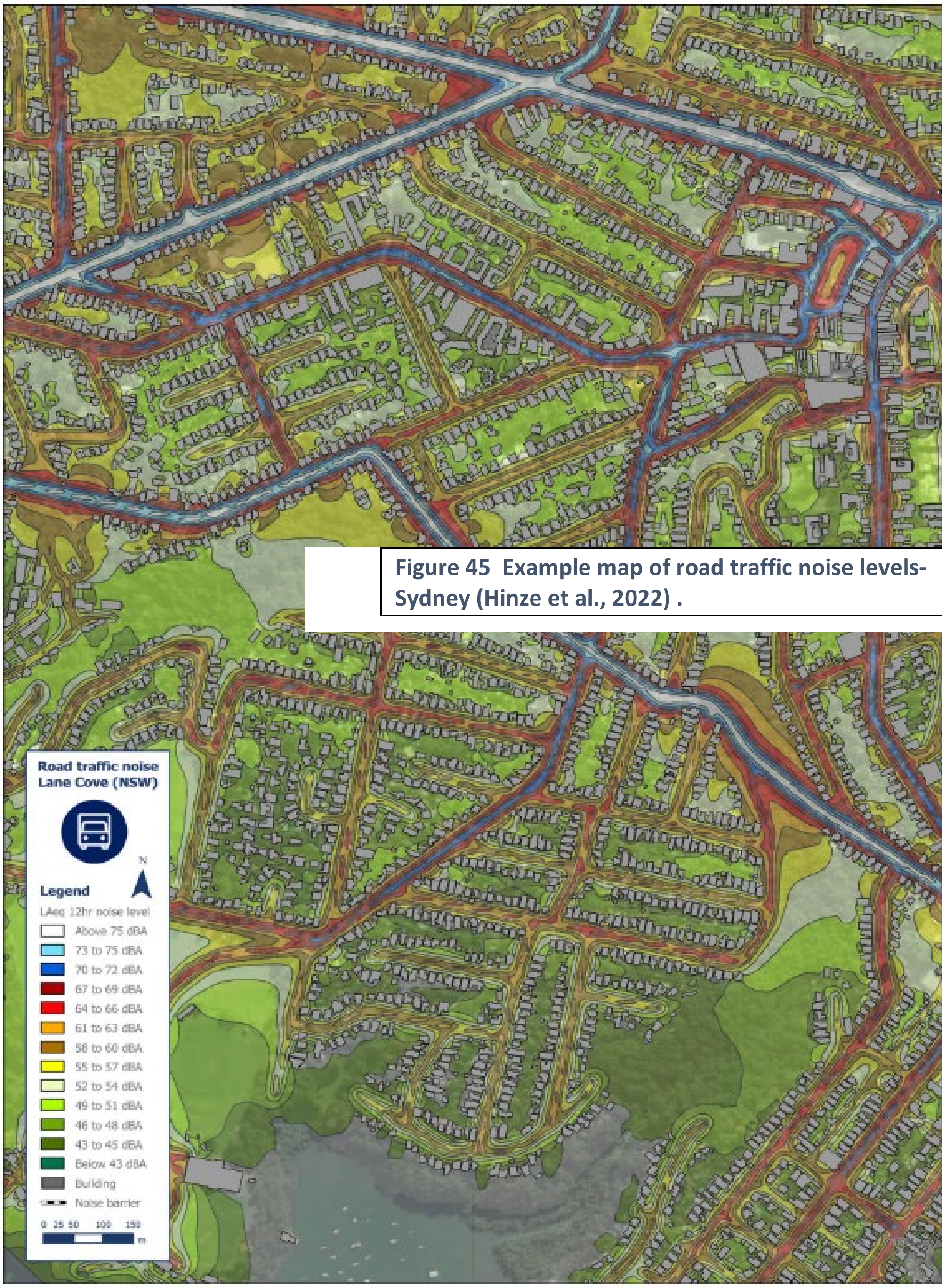



**Figure 45 Example map of road traffic noise levels- Sydney (Hinze et al., 2022) .**

The census data were collated in the form of mesh blocks[57]. The number of inhabitants of each dwelling was based on the volume of each dwelling compared to the volume of the other dwellings within each mesh block. This approach allowed a distribution of population based on the physical volume of the dwelling, allowing a larger proportion of residents to be allocated to apartment buildings that typically have a larger footprint and multiple storeys. Population exposures were estimated and are shown for $L_{den}$ in Table 20.

---

[57] Mesh Blocks are the smallest geographical area defined by the Australian Bureau of Statistics (ABS) and form the building blocks for the larger regions of the Australian Statistical Geography Standard (ASGS). All other statistical areas or regions are built up from, or approximated by, whole Mesh Blocks. The Mesh Block is the smallest geographical unit for which Census data are available and are identified with a unique 11-digit code. In 2016, there were approximately 358,000 Mesh Blocks covering the whole of Australia without gaps or overlaps. They broadly identify land use such as residential, commercial, primary production and parks, etc. (from: ABS web site, 2074.0 - Census of Population and Housing: Mesh Block Counts, Australia, 2016. https://www.abs.gov.au/AUSSTATS/abs@.nsf/Lookup/2074.0Main+Features12016?OpenDocument).

**Table 20 Numbers of inhabitants exposed in Melbourne - two different methods (Hinze et al., 2022).**

**Table 2** Number of inhabitants estimated to be exposed to $L_{den}$ transport noise levels (dB(A))—most exposed façade

| Noise level (dB(A)) | Population exposure from each transport noise source | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Roads | | Rail | | Aircraft | | Combined | |
| | Pop'n | Perc't (%) | Pop'n | Perc't (%) | Pop'n | Perc't (%) | Pop'n | Perc't (%) |
| ≤ 40 | 138,036 | 2.4 | 4,168,303 | 71.4 | 4,810,550 | 82.5 | 122,658 | 2.1 |
| 40–44 | 326,874 | 5.6 | 406,168 | 7.0 | 462,142 | 7.9 | 257,792 | 4.4 |
| 45–49 | 946,911 | 16.2 | 351,419 | 6.0 | 339,140 | 5.8 | 759,292 | 13.0 |
| 50–54 | 2,596,993 | 44.5 | 351,028 | 6.0 | 162,609 | 2.8 | 2,297,417 | 39.4 |
| 55–59 | 1,137,753 | 19.5 | 297,841 | 5.1 | 53,481 | 0.9 | 1,432,397 | 24.6 |
| 60–64 | 393,167 | 6.7 | 163,116 | 2.8 | 5440 | 0.1 | 566,196 | 9.7 |
| 65–69 | 234,352 | 4.0 | 59,093 | 1.0 | 675 | 0.0 | 295,037 | 5.1 |
| ≥ 70 | 59,954 | 1.0 | 37,072 | 0.6 | 4 | 0.0 | 103,251 | 1.8 |

**Table 3** Number of inhabitants estimated to be exposed to $L_{den}$ transport noise levels (dB(A))—50% of the building occupants allocated to the most exposed façade, with 50% allocated to the noise levels at the least exposed

| Noise level (dB(A)) | Population exposure from each transport noise source | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Roads | | Rail | | Aircraft | | Combined | |
| | Pop'n | Perc't (%) | Pop'n | Perc't (%) | Pop'n | Perc't (%) | Pop'n | Perc't (%) |
| ≤ 40 | 921,910 | 15.8 | 4,591,337 | 15.8 | 4,810,550 | 82.5 | 617,366 | 10.6 |
| 40–44 | 1,394,725 | 23.9 | 369,355 | 23.9 | 462,142 | 7.9 | 1,160,567 | 19.9 |
| 45–49 | 1,090,812 | 18.7 | 299,975 | 18.7 | 339,140 | 5.8 | 1,196,735 | 20.5 |
| 50–54 | 1,480,102 | 25.4 | 251,140 | 25.4 | 162,609 | 2.8 | 1,533,952 | 26.3 |
| 55–59 | 599,543 | 10.3 | 181,392 | 10.3 | 53,481 | 0.9 | 823,192 | 14.1 |
| 60–64 | 199,497 | 3.4 | 91,329 | 3.4 | 5440 | 0.1 | 300,790 | 5.2 |
| 65–69 | 117,433 | 2.0 | 30,815 | 2.0 | 675 | 0.0 | 149,597 | 2.6 |
| ≥ 70 | 30,018 | 0.5 | 18,696 | 0.5 | 4 | 0.0 | 51,839 | 0.9 |

Australia lags well behind most European countries in terms of its mapping of road traffic noise levels, estimation of populations exposed, and thus estimates of BOD from road traffic noise. This modelling (by Hinze et al., 2022) appears to be a commendable initiative, particularly in terms of its novel estimation procedures for road traffic flows on the complete network – in the absence in Australia of readily available traffic flow information on the ***complete*** road network of urban areas (as against just the main roads of networks). However, the methodology does not appear to be described as yet so as to reproducible, and further large-scale verification (of vehicle flows, noise levels at facades, and population exposures) is required.

Some components of the above mapping described by Hinze were utilized (by Hanigan et al., 2019) to produce interactive maps of excess risk of ischemic heart disease death due to noise, at the small neighbourhood scale. The traffic noise maps[58], and the estimate of noise level exposures at dwelling facades, appear to be problematic as noise levels from road traffic were modelled using data from the Victorian Integrated Traffic Model which includes traffic volumes for major arterial and feeder roads only. Amongst other matters, omission of smaller roads in the modelling, and spatial averaging of exposure across mesh blocks, is likely to have contributed to underestimates of noise exposures. Apparently there was a noise level validation program, but the efficacy of this in determining the accuracy of the building façade estimates of exposure is unclear.

[58] Hinze, B.: WSP Australia, Victoria EPA. Estimation of Environmental Noise Exposure: Greater Melbourne. Obtained through freedom of information, Melbourne. (2014) - **but** - described (by Hanigan et al., 2019) as: Hinze B, Victoria EPA. Estimation of environmental noise exposure— Greater Melbourne. 2013.

# NOISE EVENTS & RELATED LITERATURE

It was noted in the section '*Supplementary Noise Indicators'* that little progress appears to have been made, in the review period, in clarifying the role that *noise events* might have in road traffic noise management.

The brief required:

> *'…examination of whether community response to road traffic noise is adequately evaluated using equal energy indicators drawn from dose response approaches and/or whether maximum event assessment is needed. This is to identify what metrics should be considered to describe road traffic noise and its impact',*

and the following is a collection of observations related to this topic rather than a comprehensive response. It will be seen that there is no agreement how to consider noise events appropriately. However, this review sets out to clarify why there is continuing interest in the consideration of noise events for road traffic in terms of human response, and some relevant new material from the review period[59].

Many authors have suggested that time-averaged noise metrics (equal energy $L_{eq}$ or exceedance measures $L_{10}$) of road traffic noise streams need to be supplemented by other indicators to reflect the effect these characteristics have on human response to a road traffic stream. The measures suggested are most often those related to **eventfulness of the traffic noise** (such as the values of individual peaks $\mathbf{L_{Amax}}$**,** the statistics at the higher end of the noise level distribution – e.g., $\mathbf{L_{A01}}$, or some count of the **number of events** over a period - **NNE** (Brown, 1981) - or even **number of heavy vehicles** - **NHV** (Langdon, 1976) – to note some of the earlier papers on this topic. The NSW *Environmental Criteria for Road Traffic Noise (Environmental Protection Authority, 1999) also included the Traffic Noise Level (**TNL**)* given by: TNL = Leq(24hr) + 0.1 MNH where MNH represented the mean number of heavy vehicles per hour between 10 pm and 7 am – a combination of noise level and heavy vehicle numbers. Other authors identify components that describe the macro-temporal pattern of sounds as constituted by quiet periods (noise breaks).

A noise event does not have only an $\mathbf{L_{max}}$ but, *inter* alia, other dimensions such as **duration**, **rise time**, and **emergence** of the event from background levels. Further, **number of event** type measures may equally be described in terms of fluctuation or **variability** of the traffic noise signal. Several composite measures have appeared in the history of this field *viz*. **Traffic Noise Index** (**TNI**: background + 4x inter- decile difference), and **Noise Pollution Level** ($L_{np}$: $L_{eq} + k\ \sigma$ where **σ** is the **standard deviation of the noise level distribution**).

It is flagged here that the absence of unambiguous ways in which to appropriately identify, measure, and count, events and variations in road traffic noise signals – particularly as they would be heard indoors in bedrooms – is a major impediment to attempts to use noise events from road traffic as limit criteria. It is this problem that Brown and De Coensel have attempted to address through simulation modelling (see more below). It can also be noted that, during the review period, a novel, but ultimately not successful, attempt was made to model noise eventfulness in traffic noise streams in Switzerland using the **Intermittency Ratio (IR)** – described further later in this section.

## NOISE EVENTS and SLEEP

There is strong evidence of acute sleep effects (acute sleep effects are those that occur in temporal proximity to the event) of noise events. Acute sleep effects can be objectively measured by polysomnographic or other devices (e.g., actimetry). These do not necessarily have to involve awakening, but always a measurable change in sleep stage. By contrast, non-acute effects of sleep are assessed by self-reports or morning-after tests. Basner et al. (2010), extended their observations on the effect of aircraft noise on sleep to a road traffic study and reported:

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[59] Some of the observations in this section are made by the author as a researcher with a long-term interest in the concept of events as an important dimension in human response to road traffic streams. At this stage of their development, these do not necessarily constitute a body of knowledge that has wide acceptance amongst peers.

*'END suggests the equivalent noise level $L_{night}$ as the primary exposure variable, and our own simulations of single nights with up to 200 noise events based on a field study on the effects of aircraft noise on sleep support using … $L_{night}$ … for risk assessment. However, the precision of risk assessment may be considerably improved by adding information on the number of noise events contributing to $L_{night}$'.*

This evidence was extended in the WHO systematic review of sleep (Basner & McGuire, 2018):

*'A pooled analysis of polysomnographic studies on the acute effects of transportation noise on sleep was also conducted and the unadjusted odds ratio for the probability of awakening for a 10 dBA increase in the indoor $L_{max}$ was significant for aircraft (1.35; 95% CI 1.22–1.50), road (1.36; 95% CI 1.19–1.55), and rail (1.35; 95% CI 1.21–1.52) noise.'*

This identifies $L_{max}$[60] as the acoustic parameter of interest with regard to effect on sleep, with the outcome measure being the probability of the noise event resulting in a sleep stage change or an awakening – as measured by polysomnography. For road traffic noise, the resultant equation for the probability of additional awakenings is:

$$\text{Road: Prob. of Wake or S1} = -3.3188 - 0.0478 * L_{AS,max} + 0.0037 * (L_{AS,max})^2$$

Basner argued that the acute effects of traffic noise events should be included in the WHO guidelines. However. World Health Organization (2018) decided not to use awakening reactions or other polysomnography parameters on the basis that it is currently unclear how acute physiological reactions that affect the microstructure of sleep are related to global sleep parameters, such as total sleep time, and to long-term health impediments. The WHO thus determined to use only $L_{night}$ as a limit value for transport noise, and not to recommend limits either for the $L_{max}$ of road traffic noise events or the number of noise events.

However, this does not change the fact that noise events in road traffic streams are known to disturb sleep, and it is not unreasonable for authorities to separately impose event limits on road traffic noise, outside of the WHO Guidelines, to protect sleep. This has been foreshadowed, for several decades, by reference to *Supplementary Noise Indicators* in the END However, before this can be done , it raises the fundamental issue as to how to measure level, and number, of road traffic noise events

A recent study (Sanok et al., 2021) examined how road traffic noise impacts sleep continuity in suburban residents, providing exposure-response quantification of noise-induced awakenings from vehicle passes-by at night. This was a study with high ecological validity, with participants sleeping in their own homes and largely with no artificial manipulation of noise exposures or sleeping conditions. A new methodology for attributing awakening reactions to a vehicle pass-by was developed accounting for the intermittent nature of road traffic noise in residential areas. Noise events (at the sleepers' ears), and polysomnographic measurement of sleep, were measured simultaneously for five consecutive nights. Participants were free to choose whether to sleep with windows open, or shut, or tilted, on each night of the study. A total of 152 sleep recordings and 11,003 road traffic events were used in the analysis. The results of this study provide the best estimate available of the ERF between maximum level of a road traffic noise event (measured inside, at the sleeper's ear) and probability of a road traffic noise-induced awakening (Figures 46 and 47), and how this varies by sleeper's age. The overall finding was that sleep continuity of residents was found to be impacted by road traffic, even at moderate vehicle densities, resulting in more than one noise-associated awakening per participant and night.

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[60] The origins of work on noise events and sleep were largely in the context of aircraft noise. Time-weighted S response was used in certification of the maximum flyover levels from individual aircraft. In aircraft noise and sleep studies, these maxima were measured with S time weighting $L_{ASmax}$ (see Brown, 2013) whereas road traffic noise measurements use F response. The German study of road traffic noise (Sanok et al., 2021) noticeably reported traffic noise maxima as $L_{AFmax}$.

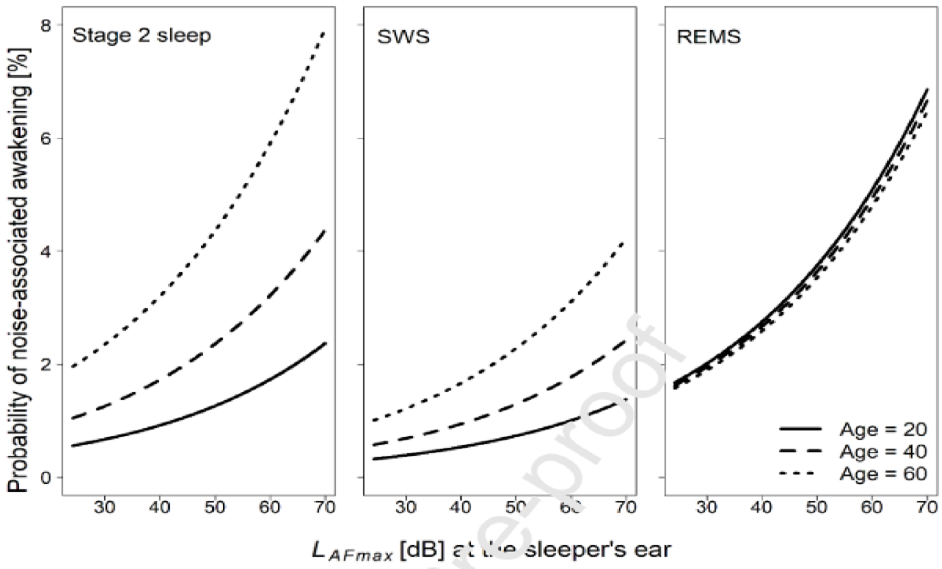


**Figure 47 Estimated probability of noise-associated sleep stage change to wake, or to S1, as a function of the $L_{AFmax}$ of a road traffic noise event, the participant's age, and the sleep stage from which the participant awakened. Stage S2 sleep (left), SWS - slow wave sleep, stages S3 and S4 - (middle), and REMS, rapid eye movement sleep (right) (Sanok et al., 2021).**

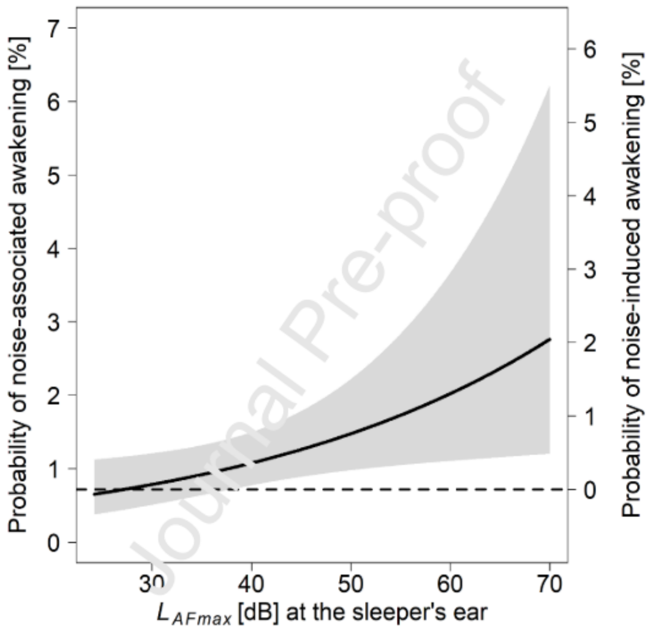


**Figure 46 Estimated probability of a *noise-induced* sleep stage change to wake or S1 as a function of the $L_{AFmax}$ of a road traffic noise event. (Assumptions, *inter alia*: median age 25 years, prior sleep stage S2). Exposure-response relationship (and 95% confidence interval ) (Sanok et al., 2021).**



There are some additional observations from this study (italics in the dot points below are direct quotations) that are useful to progress work in noise events[61]:

- duration of event, and sleeper's sex did not affect results
- '*…the nightly $L_{Aeq}$ did not correlate with the number of awakenings at night. Therefore, an event-related analysis is so important…*'
- '*The awakening probability increased with the $L_{AFmax}$ of a vehicle pass-by and the $L_{SlopeAFmax}$ of a noise event.*'
- '*The effect of the $L_{AFmax}$ on the awakening probability was influenced by the $L_{Aeq1min}$ prior to the noise event. This finding is consistent with earlier studies…and substantiates the concept that the emergence of a noise event from its background is particularly relevant for its impact.*'

[61] Figure 47 and related quotes are replicated in Box 4 (Figure 57) below, for convenient comparison with results of the Brown and De Coensel road traffic noise event simulation modelling.

- *'…the variable $L_{SlopeAFmax}$ improved the overall model fit. A steeper slope likely reflects a higher speed of vehicle pass-bys. Thus, limiting the general speed of road traffic in residential areas especially at night could be beneficial, reducing not only the $L_{SlopeAFmax}$ but also the $L_{AFmax}$ of a vehicle pass-by*
- *'The variable "vehicle class of the noise event" was not selected during the model selection process. This appears to reflect the fact that the event levels varied considerably less across vehicle classes[62] than within.*

## INTERMITTENCY RATIO (IR): A METRIC REFLECTING SHORT-TERM TEMPORAL VARIATIONS of TRANSPORTATION NOISE (SLEEP and ANNOYANCE)

Wunderli et al. (2016) noted that most environmental epidemiological studies model health effects of noise by regressing on acoustic exposure metrics that are based on the concept of average energetic dose over longer time periods. Recognizing that average energetic noise exposure measures (i.e., the $L_{eq}$ and related measures) often cannot satisfactorily predict annoyance and somatic health effects of noise, particularly sleep disturbances, it has been hypothesized that effects of noise can be better explained when also considering the variation of the level over time and the frequency distribution of event-related acoustic measures, such as for example, the maximum sound pressure level. However, it is unclear how this is best parametrized in a metric that is not correlated with the $L_{eq}$ but takes into account the frequency distribution of events and their emergence from background. Wunderli et al. (2016) present a calculation method that produces a metric which reflects the intermittency of road (and also rail and aircraft) noise exposure. The metric, termed ***intermittency ratio*** (IR), expresses the proportion of the acoustical energy contribution, in the total energetic dose, that is created by individual noise events above a certain threshold. The paper calculates the IR based on the distribution of maximum pass-by levels from information on geometry (distance and angle), traffic flow (number and speed) and single-event pass-by levels per vehicle category.

Figure 48 is an example of calculating IR from a road traffic time history. Highly intermittent traffic noise exposure situations consist of consecutive passes-by of vehicles (cars, aircraft, trains and so on) which acoustically stand out from the background (noise) by a certain degree. Wunderli et al. (2016) define such parts of the level-time course as

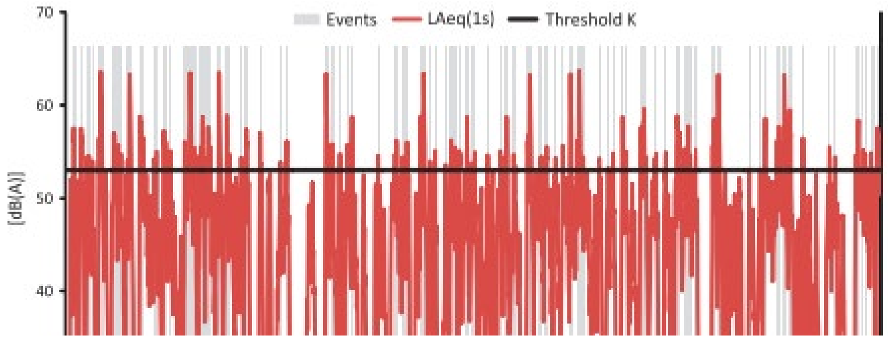


**Figure 48 Example of IR calculation for road traffic noise: Road traffic noise at 100m distance from a highway with a speed limit of 120 km/h. A total of 315 vehicles per hour with 12% heavy traffic. Intermittency ratio (IR) =62%. The portions of the curve shown 'gray' are used to calculate $L_{eq,T,Events}$ (Wunderli et al., 2016).**

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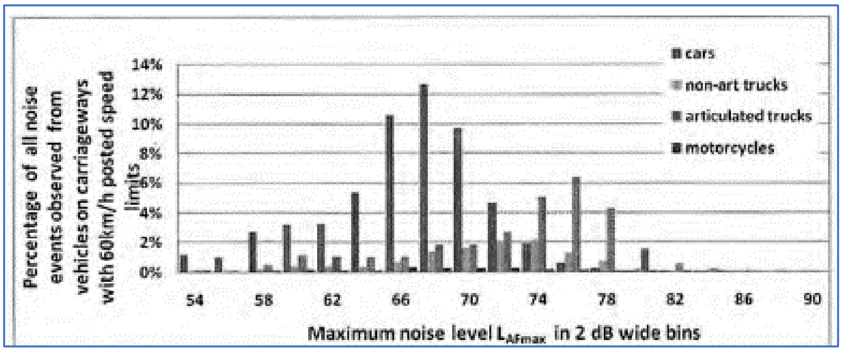
[62] This observation is supported by an extensive measurement program of maximum pass by level events of different types of vehicles in Australia. The figure in this footnote shows the distribution of noise events for different vehicle types (Brown & Tomerinni, 2011) , and that there is as much variation in $L_{AFmax}$ ***within*** vehicle types as there is ***between*** them.



"noise events". A noise event can be characterized by its maximum level, its sound exposure level, the emergence from background noise, its duration, or by the slope of rise of the level. For an integral characterization of the "eventfulness" of an exposure situation over a longer period of time, the authors introduce the event-based sound pressure level $L_{eq,T,Events,}$ which accounts for all sound energy contributions that exceed a given threshold, that is, clearly stand out from background noise. This event-based sound pressure level $L_{eq,T,Events}$ can now be compared with the overall sound pressure level $L_{eq,T,tot}$. The IR is defined as the ratio of the event-based sound energy to the overall sound energy. A single pass-by only contributes to $L_{eq,T,Events}$ if its level exceeds a given threshold K, which is C dB above the $L_{eq,T,tot.}$ In practice, the value of C was set to 3 dB.. IR can be mapped, as in Figure 49.

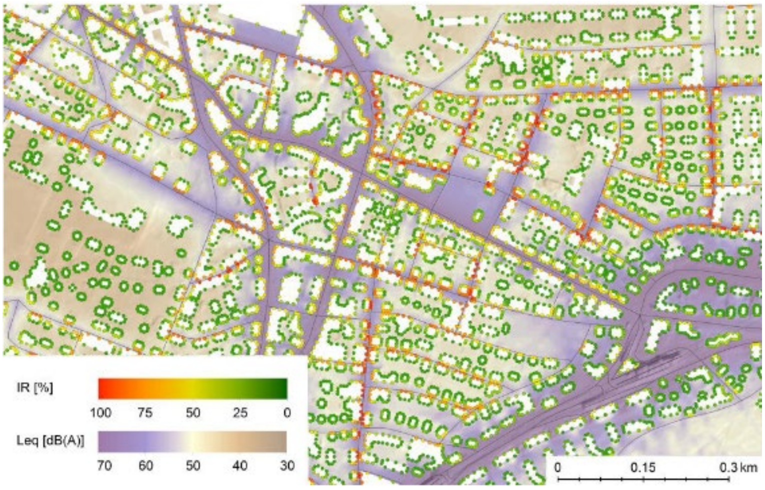



Figure 49 Intermittency Ratio (IR) and road traffic noise ($L_{eq,day}$) in an area around Zurich Wunderli et al. (2016). IR has been calculated at individual façade points on the map, marked as small dots.

IR has been included in several studies related to the SiRENE data set in Switzerland (which constitutes the whole Swiss population) including 1) '*A survey on exposure-response relationships for road, rail, and aircraft noise annoyance: Differences between continuous and intermittent noise*' (Brink, Schaffer, Vienneau, Foraster, et al., 2019), and 2) '*Self-Reported Sleep Disturbance from Road, Rail and Aircraft Noise: Exposure-Response Relationships and Effect Modifiers in the SiRENE Study*' (Brink, Schaffer, Vienneau, Pieren, et al., 2019).

The latter study has been examined earlier in the sleep section of this report – where eventfulness (IR) in the noise signal was analysed as a confounder. However, it was found to have an effect on sleep disturbance only above about 60dB. In the annoyance study, the effects of the eventfulness, measured as IR, was counterintuitive in terms of the %HA as shown in Figure 50. The expectations are that %HA should increase with increasing IR – but, as the mid and right panels show, this was definitely not the case.

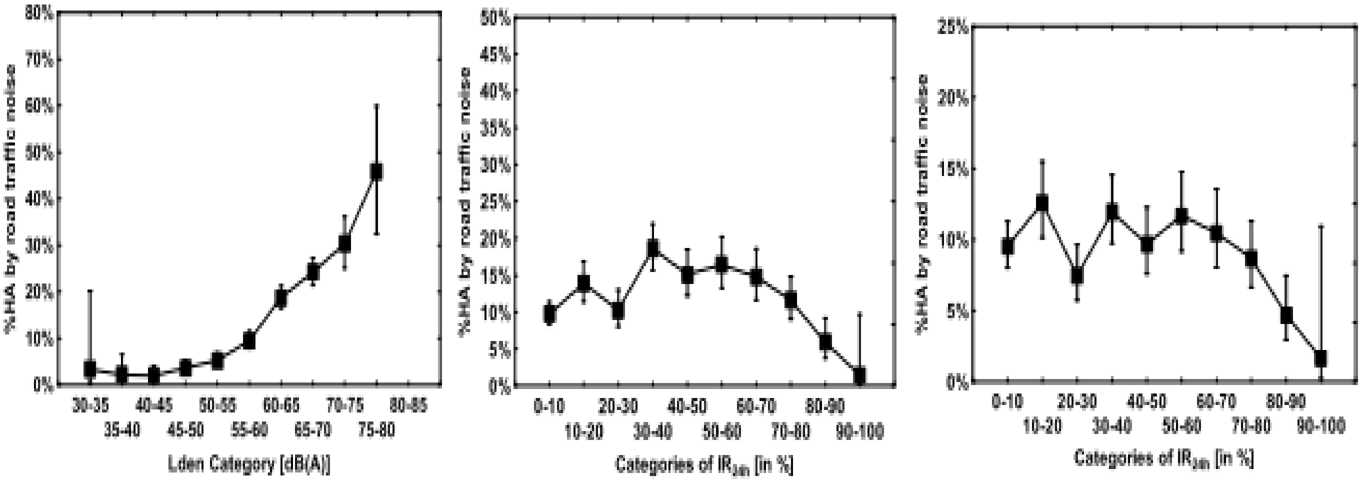



**Figure 50 %HA by $L_{den}$ (left) and IR (mid and right). Mid panel: IR only as predictor. Right panel: additionally includes $L_{den}$ as predictor. (Brink, Schaffer, Vienneau, Foraster, et al., 2019). Vertical bars are 95% CIs.**

## NOISE EVENTS and HYPERTENSION

In a reanalysis of alpine exposure-response studies for hypertension, Lercher et al. (2018) added measures of traffic noise ‘emergence’ and ‘fluctuation’ to the conventional $L_{den}$ and $L_{night}$ exposure measures (see references in the publication for exact definitions of these). The alpine situation has two roadway sources which they initially modelled separately – ‘Highways’ running down the valley, and other ‘Main Roads’ in the valley. Figure 51 shows the effect of temporal fluctuation and emergence in sound exposure on self-reported hypertension

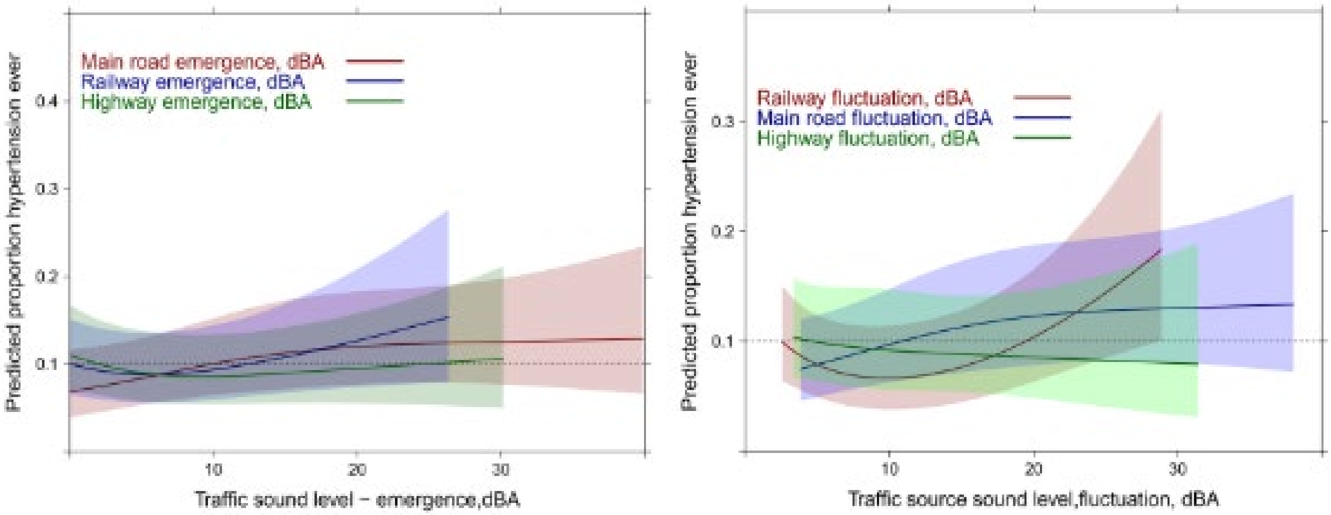



**Figure 51 Relationship between emergence (left) and fluctuation (right) of traffic noise (two different road sources, one railway source) and reported hypertension (Lercher et al., 2018).**

The authors analysed a subset of respondents who were exposed only to **both** highway and main road traffic noise, and examined the relationship between $L_{den}$ (combined from both road sources) and self-reported hypertension (blue line and CI shading) and self-reported hypertension medication (orange line and CI shading) AFTER adjusting, separately for fluctuation or for emergence (Figure 52).

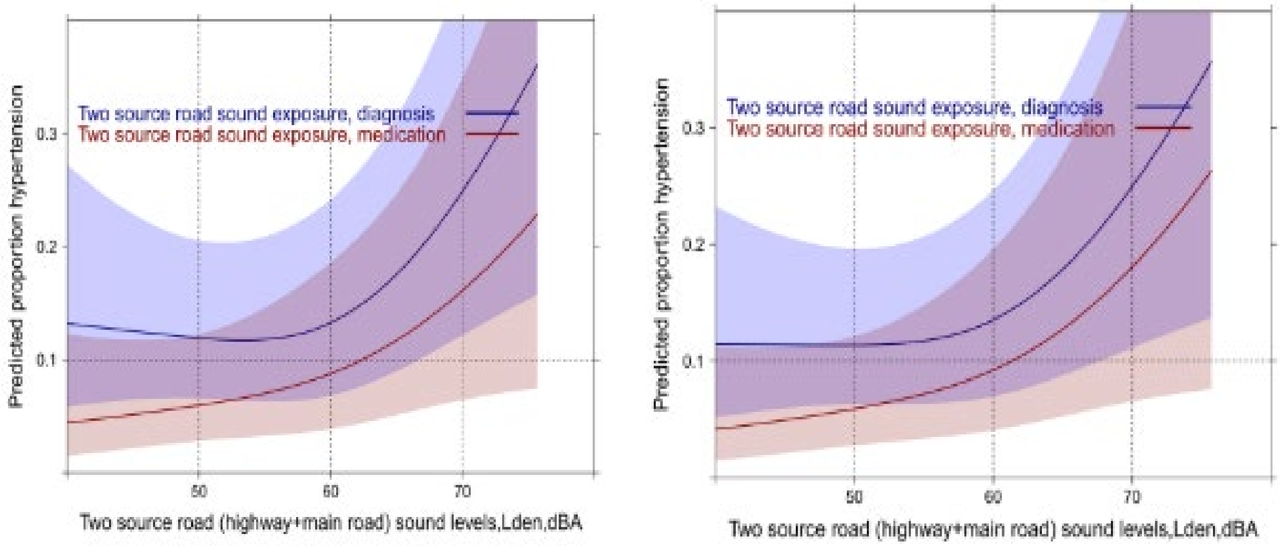



**Figure 52 Relationship between hypertension diagnosis and medication and combined $L_{den}$ from both roadway sources and adjusting for fluctuation (left) and emergence (right) in the traffic noise signal (Lercher et al., 2018)**

The authors concluded, somewhat tentatively, that ‘the results indicate the application of additional acoustic indicators is useful in health effect studies. Further methodological ideas are required to avoid the high correlations with the standard indicators. This means using indicators known to show less correlation with $L_{den}$’. In other words, in this study there were still somewhat high intercorrelations between the particular measures of fluctuation/emergence and the $L_{den}$ The conclusion was that alternative measures of eventfulness that avoid these correlations need to be identified.

The same study setting and subjects were used again in another analysis (Lercher et al., 2019), this time exploring whether different supplementary indicators, intermittency ratio (IR) and ($L_{eq}$-$L_{50}$) could be useful in estimating hypertension. By way of introduction, the authors note with respect to IR:

> *'The noise indicator Intermittency Ratio (IR) has been recently applied within the SIRENE study with annoyance as outcome. IR showed lower correlation with $L_{den}$ as compared to other noise metrics. The observed additional contribution was significant but small compared with the $L_{den}$. Furthermore, no moderation was observed between these two indicators. Also, IR was tested in the nationwide Swiss National Cohort (SNC) with cardiovascular mortality (CVM) and diurnal variability of noise exposure in focus. While the effects on CVM of noise exposure were confirmed with $L_{den}$ as indicator, except for stroke, the IR did not show a clear picture, and revealed even an opposite trend with mortality due to hypertensive disease. This raises the question, whether the IR, better reflecting the temporal features of noise, may be a helpful indicator for annoyance assessment, but not for possible far distal effects of noise, such as mortality, where the immediate perception of noise is not important anymore.*
>
> *As hypertension is a precursor (intermediate) for more serious cardiovascular illness (IHD, stroke), and such persons are known to be vulnerable to stress reactions by noise exposure – the perception of temporal pattern may be more important for the induction of biologic reactions.'*

The conclusion of the study was that neither of these 'eventfulness' indicators, IR or ($L_{eq}$-$L_{50}$) were critical in the results of this work. It is informative to note that the originator of the IR concept (Wunderli) was one of the co-authors in this (Lercher et al., 2019) reporting that IR had not met expectations of it in terms of predicting human response.

## TRUCK NOISE - MANAGEMENT INTERVENTION to REDUCE NUMBER of EVENTS (TRUCK PEAKS)

Brown (2015) reported a study of longitudinal annoyance responses to a road traffic noise management strategy that reduced heavy vehicles at night. The traffic management strategy was designed to reduce trucks using an urban corridor. The intervention had potential to affect night-time truck flows, but did not target truck traffic in the day, or vehicles other than trucks at any hour. A two-year long panel study measured the community's response to this intervention, using five repeated measurements of response. There were significant reductions in the panel's response to noise, both for night-time annoyance and for interference with activities. This was remarkable given that noise monitoring showed that the intervention produced no change in conventional traffic noise indicators.

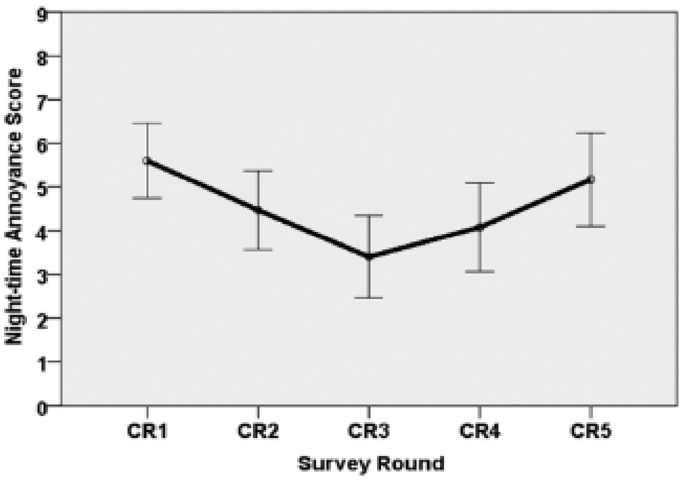



**Figure 53 Reduction in annoyance responses as a result of night-time truck restrictions (reduction in truck numbers only - no change in non-truck flows or $L_{10,18h}$ or $L_{night}$ (Brown, 2015). The figure shows changes in the estimated marginal means of the late night and early morning annoyance scores (y axis) of a panel of respondents across five rounds of survey.**

However, there were measurable changes in the number of articulated truck movements at night, and the benefit can be attributed to reduction in the number of noise events from heavy vehicles. The parallel tracking of changes in reported noise effects and the numbers of heavy vehicles in the night hours in this longitudinal study provides strong support to the notion that noise effects at night depend on the number of noise events experienced, not only on the overall level of traffic noise. Conventional indicators appear to be unresponsive in terms of assessing the noise-effect benefit of heavy vehicle reduction strategies. The increase in Night-time Annoyance Scores after CR3 is likely a response to overall growth in vehicle numbers using the roadway as the study progressed over several years.

## NOISE EVENTS, READING and COGNITIVE PERFORMANCE (and ANNOYANCE)

Lavandier et al. (2022) reported the Influence of road traffic noise peaks on reading task performance and disturbance in a laboratory context. It examined the influence of fluctuating road traffic noise on perceived disturbance during a reading task as well as on performance, inside a living room. The comprehension rate of the texts decreases as the background sound level increases. The reading speed decreases during the rising front of a peak and goes back to its former regular reading speed during the descending front of the peak. This slowdown during the rising front reaches 14% for slow readers. The declared disturbance is higher for slow readers compared to fast readers, and higher for noise sensitive persons compared to non-sensitive persons. The influence of the acoustic indicators on the reading disturbance is very small compared to the influence of personal factors. There is no influence of the number of events on this disturbance.

There are also persistent opinions that human annoyance with road traffic noise may also depend on the pattern of events in the traffic noise signal. A recent study (Schäffer et al., 2022) examined how the macro-temporal pattern of road traffic noise affect both noise annoyance and cognitive performance – though this was restricted to a laboratory simulation of office workers, not of people in dwellings, and human response was measured as 'short-term' annoyance. The following indicators were used to quantify the macro-temporal pattern of the road traffic noise situations (see original paper for definitions of the macro-temporal pattern:

- Number of events (N),
- Relative Quiet Time (RQT),
- Intermittency Ratio (IR),
- Centre of Mass Time (CMT),
- A-weighted and FAST-time weighted maximum sound pressure level ($L_{AFmax}$), and
- Quiet Time Distribution (QTD).

The study found: '*annoyance decreased with increasing total duration of quiet periods. Also, the distribution of the quiet times affected annoyance. Shorter but more regular breaks [in the noise signal] were found to be less annoying than longer but irregular breaks of identical total duration; a minimal necessary duration of noise breaks as proposed in literature could thus not be confirmed. Cognitive performance in an attention-based task, in contrast, did not systematically vary with the macro-temporal pattern of the situation*'.

## INVESTIGATION of NOISE EVENTS THROUGH MODELLING – WORK by BROWN and De COENSEL

In a series of papers (De Coensel & Brown, 2011; Brown, 2014; Brown & De Coensel, 2017; Brown & De Coensel, 2018; De Coensel & Brown, 2018) a novel approach was taken to advance the understanding of noise events in road traffic streams and the appropriate way in which events should be measured[63].

The overall methodology involved testing how different formulations of event detection perform across the population of acoustic conditions that can exist near roadways. A modelling approach had to be adopted as this provided the ability to test a generalized algorithm for event detection against time histories of traffic noise generated for all likely situations. Traffic flow variables, and the propagation distance from roadway to receiver, covered a full range of realistic values, and these could be varied independently—something that would be impractical using data gathered in field measurement studies. The modelling study had a two-stage approach. Firstly, the time history of instantaneous sound level was simulated for a population of road traffic noise exposure scenarios. Secondly, an extensive set of alternative exceedance algorithm were utilized to detect and count noise events in these time histories.

Figure 54 shows the range of the traffic and distance variables used in the time history simulations – a total of (2 x 10 x 5 x 5) 500 unique simulation settings. Time histories of road traffic noise were modelled, using the AIMSUN microscopic traffic simulator, and Noysim2 noise emission and propagation modelling (De Coensel & Brown, 2011). Each simulation was run for the equivalent of one hour of measurement time (with a simulation time step of 125ms) and each unique scenario was simulated 30 times. Effectively, this was equivalent, in the real world, to 30 independent hours of measurement of road traffic noise for any one given combination of traffic and distance variable shown in Figure 55. Simulations were completed for all 500 unique traffic and distance simulation settings – which effectively represents all (unshielded) road traffic noise exposure conditions that are likely to be of relevance in urban areas in terms of significant effects on people in dwellings. For illustration, a snapshot of 10 minutes of one-time history of road traffic noise levels simulated for just one combination of traffic and distance settings (30m from a two-lane road with a speed limit of 60 km/h, and a traffic volume of 50 vehicles/h, of which 20% is heavy vehicles) is shown in Figure 55.

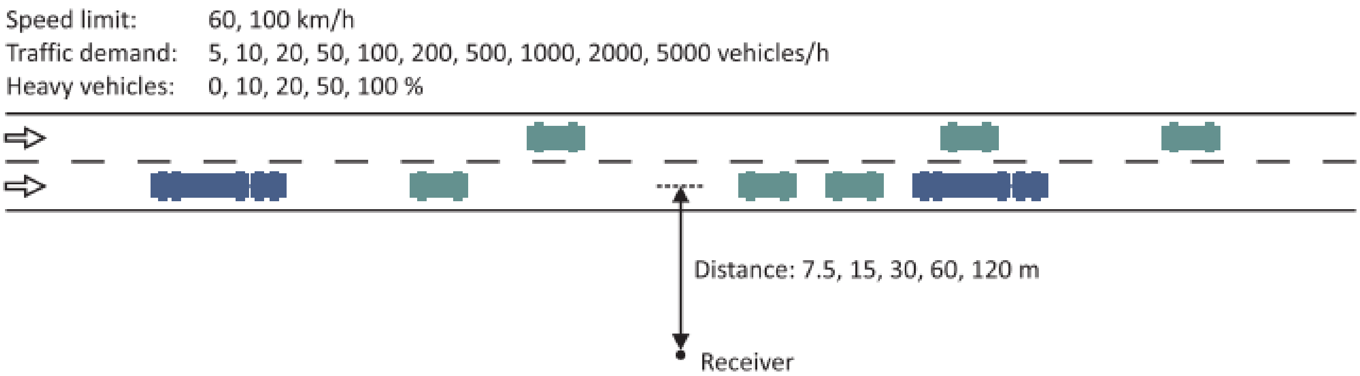



**Figure 54 Simulation setting and ranges of each variable included in the simulations (Brown & De Coensel, 2017)**.

These simulated time histories of noise levels at receiver locations were then searched to identify noise events. It is important to note that, because sleep researchers report exposures at the sleeper's ears, it is indoor noise levels that need to be modelled – and this is even more critical for detections that include 'emergence', as an event emerges from a different background indoors than it does outdoors – and detection algorithms have to be identified for two different conditions – open windows and closed windows. The same 10 minutes of simulation is shown in each of the four panels in Figure 56 – both outdoors and indoors, with windows opened and closed. The vertical red bars

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[63] The reason publications from this promising line of research has not continued beyond 2018 - despite the bulk of the complex simulations and analyses having been completed by then - was that both chief investigators were diagnosed with serious illnesses – sadly leading, in the case of Bert de Coensel, to his passing.

indicate when two different algorithms detect the number of noise events in that simulated road traffic noise time history. One event in panels (a) and (d); three events in panel (c) and none in (b).

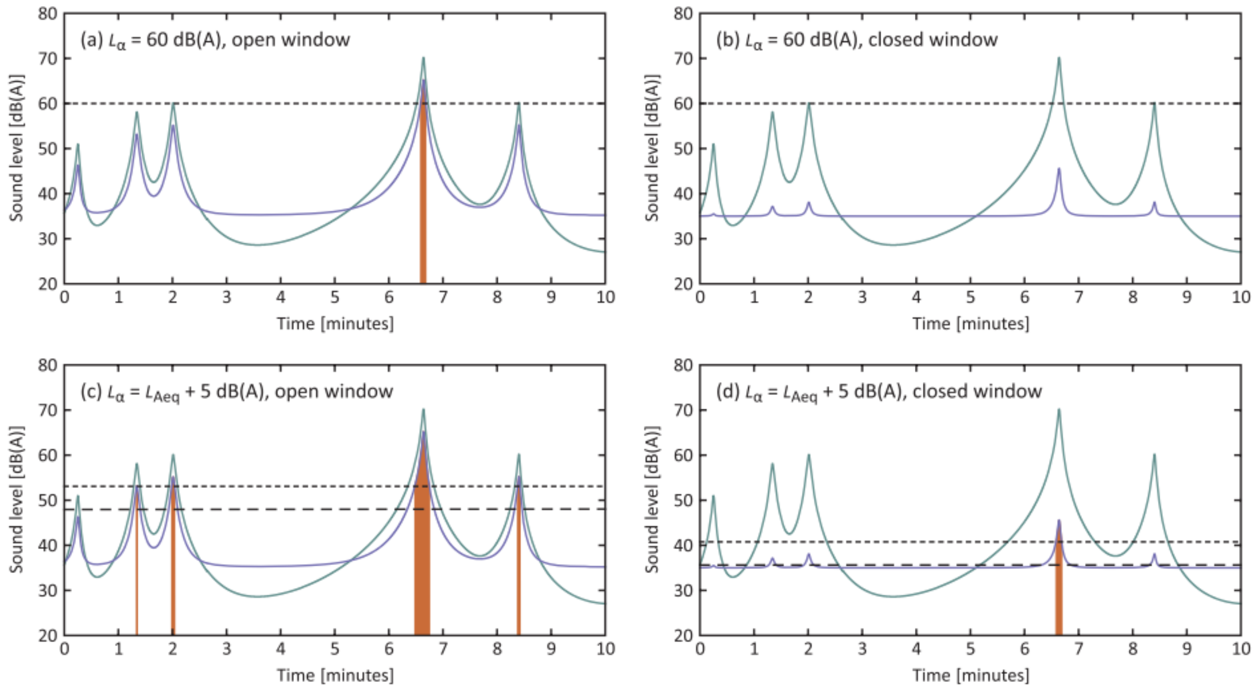


**Figure 55 Simulated time history of the outdoor sound level (free-field, solid green line) at a distance of 30m from a two-lane road with a speed limit of 60 km/h, and a traffic volume of 50 vehicles/h, of which 20% are heavy vehicles. The indoor sound level time history is shown in a solid blue line, for open windows on panels (a) and (c), for closed windows on panels (b) and (d). The detection of noise events within the indoor sound level time history is illustrated using two different algorithms - on panels (a) and (b) with Lα set to 60 dB(A) (short, dashed line), and on panels (c) and (d) with Lα set to $L_{Aeq}$ +5 dB(A) (long dashed line represents $L_{Aeq}$). In both algorithm settings, the minimum time gap between events is set to 3 s. Detected noise events are tagged using a vertical red bar, for the duration of the event. There is also a minimum background level from non-transport sources set for indoors. (Brown & De Coensel, 2017).**

For 'event detection' a generalized exceedance-based detection algorithm was constructed on the basis of an extensive literature review (Brown & De Coensel, 2018). With this algorithm, the onset of a noise event is detected when the instantaneous sound level exceeds a threshold level Lβ with an emergence of at least E. Noise events are only retained when the time gap (or noise-free interval) since the previous event is larger than $\tau_g$.

There are many possible algorithms, and the way in which this was statistically reduced to just a prototypical seven (as in Table 21) – which met validity, reliability and reasonableness criteria is described in full in Brown and De Coensel (2018).

| Label | $L_\beta$ | $E$ | $\tau_g$ | Window |
|---|---|---|---|---|
| NA50 | 50 dB | 0 dB | 5s | opened |
| NA55 | 55 dB | 0 dB | 5s | opened |
| NA65 | 65 dB | 0 dB | 5s | opened |
| NA70 | 70 dB | 0 dB | 5s | opened |
| NAL50E03 | $L_{A50}$ | 3 dB | 5s | closed |
| NAL50E10 | $L_{A50}$ | 10 dB | 5s | opened |
| NALEQE03 | $L_{Aeq}$ | 3 dB | 30s | opened |

**Table 21 The seven selected prototypical algorithm parameter sets, together with the naming convention (see Brown & De Coensel, 2018).**

**NAxx = number above xx dB**

NAL50Exx = number xx dB above $L_{50}$

NALEQExx = number xx dB above $L_{eq}$

Table 22 shows the correlation between $L_{Aeq}$ outside the dwelling, and the number of noise events detected inside. The NA50, NA55 and NAL50E10 algorithms result in low correlations with $L_{Aeq}$, indicating that these measures can provide useful supplementary information beyond that provided by $L_{Aeq}$. The NA65 and NA70 fixed parameter algorithms, and the other two adaptive algorithms, NAL50E03 and NALEQE03, have high rank order correlations with $L_{Aeq}$, suggesting that they would not be useful as supplementary indicators. Only NA50, NA55 and NAL50E10 meet the I-INCE criterion (I-INCE, 2015) that the product-moment correlation with $L_{Aeq}$ must not exceed 0.5, and are thus primary candidates for further consideration as event-counting supplementary indicators.

| Label | Spearman's rank corr. coefficient | Pearson corr. coefficient |
|---|---|---|
| NA50 | -0.244 | 0.003 |
| NA55 | 0.123 | 0.216 |
| NA65 | 0.787 | 0.573 |
| NA70 | 0.868 | 0.628 |
| NAL50E03 | 0.917 | 0.824 |
| NAL50E10 | 0.384 | 0.454 |
| NALEQE03 | 0.633 | 0.611 |

**Table 22 Correlations between $L_{Aeq}$ outside the dwelling and the number of events detected inside the dwelling, for the 7 selected algorithm parameter sets.**

These three noise event detection algorithms, NA50, NA55 and NAL50E10 are considered further below.

A more detailed view of the relationships between the number of noise events detected and the $L_{Aeq}$ across the 500 traffic/distance scenarios is shown in Figure 56. The panels plot the number of noise events as they would be detected inside a dwelling with open windows for all traffic flows modelled, with separate panels for different distances from the roadway, against the outside $L_{Aeq}$.

The $L_{Aeq}$ is, as would be expected, linearly related to the logarithm of traffic flow, and decreases with increasing distance from the roadway. For clarity, the effects of variation in the percentages of heavy vehicles and in the traffic speed on the indicators is not shown in these figures. Figure 57 confirms the observation above that NA50 and NA55 both have distinct non-monotonic relationships with $L_{Aeq}$ and are therefore potential supplementary indicators. These algorithms detect events from the lowest traffic flows up to traffic flows of 1,000 to 2,000 vehicles per hour, with no events detected at higher flow rates. Both indicators also detect events at all the distances modelled, with decreasing numbers of events as the distance from the roadway increases. Figure 57 also confirms the observation that NAL50E10 too has a non-monotonic relationship with $L_{Aeq}$, and thus may also be a potential supplementary indicator. It detects events across the full range of traffic flows, with the maximum number occurring at flow rates of 500 to 2,000 vehicles per hour. But unlike the NA50 and NA55, the maxima occur at lower flow rates as distance from the roadway increases. The number of noise events also decreases with distance[64].

**Summary:** The study reported above is based solely on physical modelling of traffic noise time-histories. But any actual use of noise event measures to supplement conventional noise indicators such as $L_{Aeq}$ for management of road traffic noise will, in the end, have to be determined through human effects research. This will entail examining if and how levels and events contribute, independently or in combination, to an association between road traffic noise and human response. There are indications, both from sleep research and from other studies of human effects of noise, that human response may also depend on noise events as well as on level – the problem to date is that there is no agreement as yet as to how noise events should be measured.

This work finds that NA50 and NA55, that detect the number of events exceeding 50 dB(A) and 55 dB(A) respectively, and NAL50E10, that detects the number of events exceeding $L_{A50}$ with at least 10 dB(A), all have the potential to be considered for practical application as event detection measures. All three are found to produce valid, reproducible, and sensible (in terms of intuitive expectation) counts of events. They each apply to the detection of noise events heard indoors with the windows of the dwelling open.

[64] The original paper should be consulted to see some more subtle observations with respect to the functioning of these event detection algorithms as distance from the roadway and traffic flow parameters change.

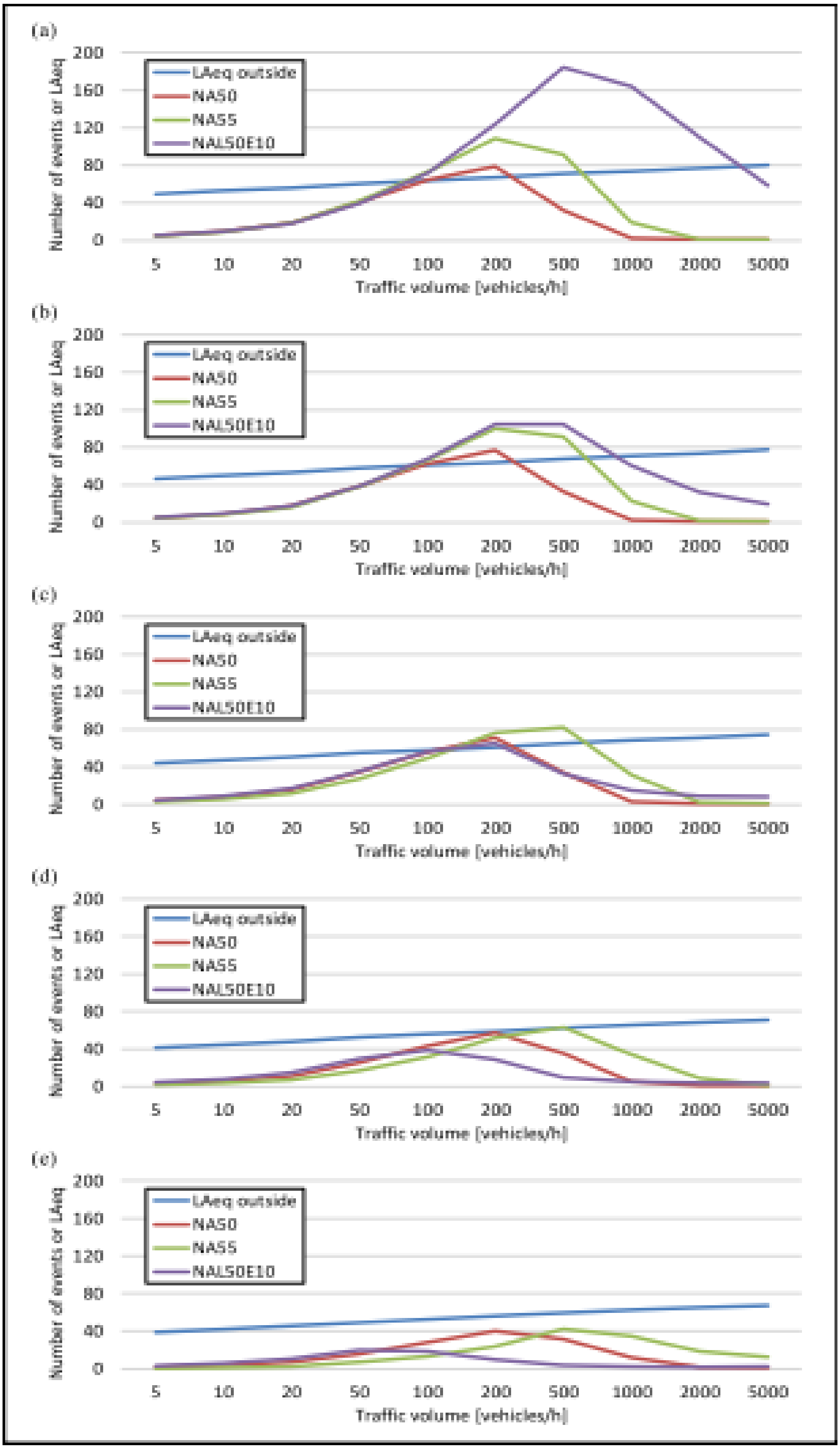


**Figure 56 Comparison of outdoor $L_{Aeq}$ with NA50, NA55 and NAL50E10 as a function of traffic volume, for different distances to the roadway: (a) 7.5m, (b) 15m, (c) 30m, (d) 60m, (e) 120m (De Coensel & Brown, 2011).**

Further to this summary, it seems useful to ***speculate*** about the apparent consistency in the behaviour of the NA50 and NA55 algorithms in Figure 56, with observations made previously about the effect of $L_{AFmax}$ on acute sleep disturbance. Those observations on human response to noise event maxima have been duplicated, from the sleep section, to Box 4 in Figure 57 below. It is also useful to speculate whether the NAL50E10 algorithm may have utility in reflecting the measure suggested in the third dot point in Box 4 *(viz. 'The effect of the $L_{AFmax}$ on the awakening probability was influenced by the LAeq1min prior to the noise event. This finding is consistent with earlier studies…and substantiates the concept that the emergence of a noise event from its background is particularly relevant for its impact.')*

As already indicated, whether the event detection algorithms will be useful will have to be substantiated through human effects research. The major advance made in the current work is that one now has a strong lead as to which noise event measures to use in future work.

**The $L_{AFmax}$.** There is one other useful observation from the above modelling work. Figure 58 shows the correlation between $L_{Aeq}$ and $L_{AFmax}$ in simulations across the population of urban traffic noise conditions – as described above, with both measured outside the dwelling. The correlations between the two indicate that the maximum measure is unlikely to be a useful supplementary indicator.

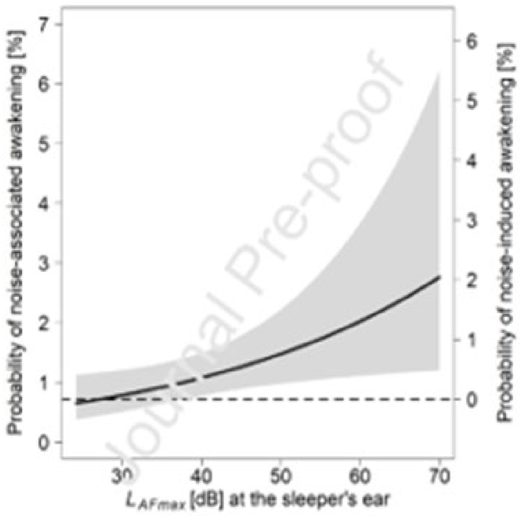


Figure 47 Estimated probability of a **noise-induced** sleep stage change to wake or S1 as a function of the $L_{AFmax}$ of a road traffic noise event (Assumptions, *inter alia*: median age 25 years, prior sleep stage S2). Exposure-response relationship (and 95% confidence interval ) (Sanok et al., 2021)

There are some additional observations from this study (italics in the dot points below are direct quotations) that are useful to progress work in noise events[56]:

- duration of event, and sleeper's sex did not affect results
- *'…the nightly $L_{Aeq}$ did not correlate with the number of awakenings at night. Therefore, an event-related analysis is so important…'*
- *'The awakening probability increased with the $L_{AFmax}$ of a vehicle pass-by and the $L_{SlopeAFmax}$ of a noise event.'*
- *'The effect of the $L_{AFmax}$ on the awakening probability was influenced by the $L_{Aeq1min}$ prior to the noise event. This finding is consistent with earlier studies…and substantiates the concept that the emergence of a noise event from its background is particularly relevant for its impact.'*
- *'…the variable $L_{SlopeAFmax}$ improved the overall model fit. A steeper slope likely reflects a higher speed of vehicle pass-bys. Thus, limiting the general speed of road traffic in residential areas especially at night could be beneficial, reducing not only the $L_{SlopeAFmax}$ but also the $L_{AFmax}$ of a vehicle pass-by*

**Figure 57 BOX 4: Observations on the effect of $L_{AFmax}$ on acute sleep disturbance - copied here, for ease of comparison, from text associated with Figure 47 in the 'Noise Events and Sleep' section.**

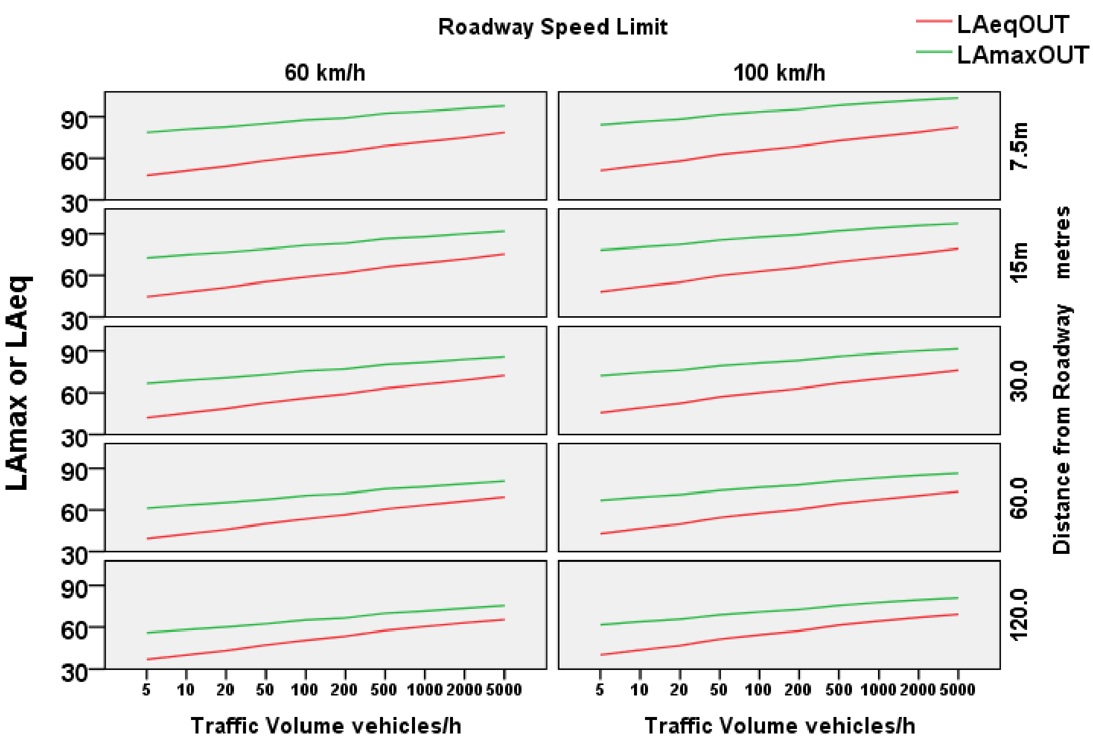


**Figure 58 Modelled relationship between LAeq and LAFmax in simulations across the population of urban traffic noise conditions - both measured outside the dwelling.**

## OTHER WORK on NOISE EVENTS

Can and co-workers (e.g., Can et al., 2008; Can et al., 2016; Can & Aumond, 2018) have written extensively about road traffic noise dynamics, including events. This work is recorded here for completeness, but without further comment For example,.

> *'The "Number of Noise Event" NNE and "Mask Indexes" indicators are often used to describe emergences. NNE and MI are defined as the number of events, and the percentage of the time, that exceed a given threshold, respectively. The threshold can be a fixed value (i.e., 70 dB) or can be set adaptively, e.g., based on a noise indicator (i.e., LAeq+10, L10+10). These indicators can be defined to measure either noisy or quiet periods. Thus, they are more adapted than the previously cited indicators to describe sound levels variations. Nevertheless, each of the NNE or MI offers only a partial view of the emergences: the NNE takes the same value regardless the duration of events, and the MI takes the same value regardless the number of events. These two indicators can be merged into a map of emergences, which nonetheless loses in enforceability. However, the calculation of NNE and MI has not been standardized. Different algorithms to detect and count noise events within a sound level time history can result in vastly different values of NNE and MI ; the mechanics behind the detection of events has to date not been investigated in detail.'*

# ROAD TRAFFIC NOISE PREDICTION MODELS

Before examining post-2010 literature on road traffic noise prediction, it is firstly useful to rehearse the range of applications of these models. This is because the applications have evolved and expanded over time. Different applications can have quite different requirements in terms of accuracy of predicted level, data input requirements, and associated output formats. There are at least four recognizable applications of prediction modelling:

1. Utilisation in roadway design, in impact assessment and preparation of mitigation strategies in action plans. This requires predictions of levels with known accuracy, at specific receptor locations, or over specific areas (such as the area of a road project) in a form where the model can estimate the level at noise-sensitive uses, and also evaluate the effect of different mitigation interventions.
2. Determination if predicted level exceeds some limit value for an individual location (e.g., a dwelling, or a classroom) where mitigation or compensation could be considered. This is largely the same as 1) but has been listed as a separate application because it likely requires a higher, known, standard of certainty because of potential of legal testing for erroneous predictions – and the possibility the prediction may be tested with field measurements at a later date.
3. Mapping noise exposure over some (usually urban) area in a way that facilitates counting of the numbers of people exposed in their dwellings to different levels of road traffic noise (so-called Strategic Noise Mapping). This is currently the most common use of road traffic noise prediction modelling in Europe, and driven by the noise mapping requirements of the Environmental Noise Directive (EU, 2002). Extensive requirements, and guidelines, have been prepared for such mapping work. The outputs, widely called *Strategic Noise Maps* are not an end in themselves but are primarily a tool for estimating the numbers of sensitive uses/people, exposed to different levels of road traffic noise over the mapped entity. A useful review of Strategic Noise Mapping is provided in a chapter by Murphy and King (2022a).
4. The fourth application is similar to (3), but rather than counting numbers exposed through an agglomeration, the task is to assign accurate estimates of road traffic noise exposures to all residents in a study area/residential building for epidemiological studies. Effort is directed at minimizing error (termed *misclassification* in such studies) in the assignment of exposures to the subjects to reduce uncertainty in epidemiologically determined exposure-health-response relationships[65].

It is flagged here that examination of the literature shows that much international research on traffic noise prediction modelling in the last decades has, almost exclusively, focussed on Applications (3) and (4) above - noise mapping, and epidemiology exposure studies - rather than on Applications (1) and (2) - roadway and mitigation design and testing if limit values are exceeded. This is pertinent given that most interest in prediction modelling in Australia/NSW has, to date, been the latter. See, for example, papers by Peng and co-workers which are focussed on the application and accuracy of CORTN modelling in Australia. These are described below in the section '*Recent Evaluations/Commentaries on CORTN in Australia*'.

Modelling has successively evolved through applications (1) to (4) over the last 50 or so years. While the nature of the output, and purpose, of prediction modelling may have evolved, it should be noted that the approach/algorithms that underlie any particular model may have undergone refinement/evolution rather than comprehensive change. There has, of course, been computational improvement: moving from hand calculation to algorithms in programmable calculators and computers, to integrated models linking roadway infrastructure, traffic flow, terrain, propagation, and population densities. Modelling outputs have also changed, driven by sophisticated developments in GIS and mapping software.

[65] There could also be an emerging fifth purpose (or perhaps better seen as an extension/refinement) - a so-called receiver-centric mapping approach, introducing *noise-exposure sensitivity maps* to assess the potential impact from a specific vehicle moving in a given network, quantifying the associated exceedance over the prevailing background noise, under varying spatiotemporal traffic conditions (see Baclet et al., 2022).

## A PLETHORA of ROAD TRAFFIC NOISE PREDICTION MODELS

While this work aims to examine recent literature on prediction models, it became clear that some overview of model evolution was necessary because so much of current practice (including Australian practice) continues to reference models that originated much earlier. A light perusal of operational models (not a comprehensive search) identifies a large range of models in use internationally. Many of the models have *national* origins.

Steele (2001) enumerated those models in use at the time (excluding Scandinavian models). This is part reproduced in Table 23 below. The Table compares various inputs and outputs of each model, and validation information if available. Steele (2001) listed user groups for these models as: roadway engineers; acoustical engineers; expert witnesses; preparers of Environmental Impact Statements; and consultants working for owners of building affected by road traffic noise – clearly all users originally involved in applying prediction models to applications (1) and (2) listed above – well before the introduction of the END requirement for noise mapping.

Later expansion of models to application (3) was driven by the requirement in the END to produce Strategic Noise Maps. For this, European Member States were allowed to use their own prediction models. Tables 24 and 25 list different noise calculation methods utilised across the EU during the first (2007) and third (2017) rounds of strategic noise mapping. Models listed in Tables 24 and 25 overlap with those in Table 23, though do not include FHWA models[66]. In fact, there is little published in recent years[67] on FHWA/TNM models.

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[66]The TNM method is the FHWA accepted calculation method to predict noise from highways and was not meant to be applied to the mapping of complex city environments involving a grid of many receiver points, and with more reflection and diffraction than would be experienced near a highway. Thus, it has not been widely used for noise mapping. King et al. (2014) assessed its applicability to mapping studies and found that, in its current form, it is not suitable for the development of noise maps in urban areas (Murphy & King, 2016).

[67] Exceptions includes National Academies of Sciences (2014) that provides supplemental guidance on the application of the TNM to various specific situations such as structure-reflections and expansion-joints; intersections; median and parallel barriers; multi-lane highways; tunnel openings; and a range of propagation effects including wind and temperature gradients. Another exception was Sun et al. (2014) who compared the implementation of the TNM model on SoundPLAN and Cadna/A in predictions of the exposures of the first row of buildings on road projects in Canada.

Table 23 Principal road traffic noise prediction models in use around 2000 in North America, Europe, Australia and Japan (extracted and adapted from Table 1 of Steele (2001)).

| | FHWA STAMINA | FHWA TNM version 1.0 | 01 db MITHRA | CoRTN | RLS 90 | STL-86 | ASJ-1993 |
|---|---|---|---|---|---|---|---|
| Government users | USA, Canada, Japan, Mexico | In course of preparation. Replaces STAMINA | France, Belgium | UK, Australia, Hong Kong, New Zealand | Germany | Switzerland | Japan |
| Applications | Highway ($L_{eq}$), not architectural. Grid. Road networks | Highway ($L_{eq}$), not architectural. Grid, excellent source base. Road networks | Highways and railways ($L_{eq}$), not architectural. Grid, Good propagation. Simple streams | Highways (quasi $L_{10}$).Point. Single traffic streams only. | Highways and car parks ($L_{eq}$), not architectural. Point. Good propagation. Simple streams only. | Highways and Trams, light rail ($L_{eq}$) not architectural. Point. Simple streams only. | Highway barriers. |
| Predicts traffic volumes? | No | No | Yes | No | Yes | Yes | No |
| Traffic conditions | Constant speed, grades | Constant speed, acceleration, grades, and interruption | Constant speed, grades | Constant speed, grades. | Constant speed, grades, quasi-intersections, interruptions | Constant speed, grades. | Constant speed. |
| Input data | Traffic speed, flow, road and environs data. Local characteristics | Traffic type, flow, speed, whether interrupted, road and environs data. Local characteristics. | Traffic type, flow, road and environs data. | Heavy/light ratio, flow, speed, road and environs data. | Traffic type, flow, park or road data, and environs. | Traffic type, flow, road and environs data. | Traffic type, flow, speed, barrier geometry |
| Type | Hybrid, consistent/ inconsistent | Mathematical/ hybrid. | Hybrid, consistent. | hybrid, inconsistent. | Hybrid, consistent. | Hybrid, inconsistent. | Mathematical. |
| Noise descriptor | $L_{eq}$/quasi$L_{10}$ | $L_{eq}$ | $L_{eq}$ (8–20 H) | Quasi- $L_{10}$ (18 h) | $L_{eq}$ | $L_{eq}$ | $L_{eq}$, quasi- $L_{50}$ |
| Type of mapping | Point → grid | Multiple dual points → grid | Line → grid | Line → point | Line → point | Line → point | Multiple points → point |
| Source | Simple streams | Simple stream | Simple stream | Single stream | Simple stream | Simple stream | Simple straight stream |
| Propagation | Energy type | Energy type | Ray tracing | Energ by type | Energy type | Energy type | Mathematical (velocity potential) |
| Vehicle types | Automobile/ medium trucks/ heavy trucks | Optional spectra for automobile/ medium trucks/ heavy trucks/ buses/motor cycles. | Light vehicles, heavy vehicles, and trains. | Light vehicles/ heavy vehicles | Light vehicles/ heavy vehicles/ car parks. | Light vehicles/ heavy vehicles/ trams and suburban trains on roadways. | Light vehicles/ medium vehicles/ heavy vehicles |
| Validation | Contingent; 0.58 to −1.3 dBA @ 15 m to 60 m. | Not readily available. | Not readily available. | +1.4 @ 50 to 54.9 dBA (Delany): −1.2 @ 80 to 84.9 dBA. +1.7 @ facades (Saunders). | Not readily available. | Not readily available. | Not readily available. |
| Weighting: Source/receiver | dB(A)/dB(A) | 1/3 octave/dB(A) | octave/dB(A) | dB(A)/dB(A) | dB(A)/dB(A) | dB(A)/dB(A) | dB(A)/dB(A) |
| Major faults | No. $L_N$, no interruptions. | No $L_N$, simple interruptions only. | No $L_N$, no interruptions, no local characteristics | No $L_{eq}$, $L_{10}$ not rigorous, no interruptions, single traffic streams only, no local characteristics | No $L_N$, simple interruptions only, no local characteristics | No$L_N$, simple interruptions only, no local characteristics | No $L_N$ restricted to quasi $L_{50}$, no interruptions, long roadside barriers only. |
| Authors opinion | Obsolescent. | $L_{eq}$ only; allows local vehicle types. Accurate. | Use for complex buildings and unknown traffic flows. | Obsolete. | Use for car parks and unknown traffic flows. | Use for trams and light rail and unknown traffic flows. | Use for free-flowing traffic with long roadside barriers |

**Table 24 Noise Calculation Methods Utilised Across the EU During the First (2007) Round of Strategic Noise Mapping (after Murphy & King, 2022a).**

| **Model** | (Country) |
|---|---|
| RVS 3.02 | (Austria) |
| NMPB/XPS 31-133 | (France) |
| Temanord 525 | (Nordic) |
| RLS90 | (Germany) |
| CRTN | (UK) |
| RMW2002(SRM I+II) | (Netherlands) |
| StL 86 | (Switzerland) |

**Table 25 'Examples' of Noise Calculation Methods Utilised Across the EU During the Third Round (2017) Strategic Noise Mapping (after European Environment Agency, 2020).**

| **Model** | (Country) |
|---|---|
| RVS 4.02 | |
| NMPB-Routes-96 | |
| NMPB-Routes-2008 | |
| Nord2000 | |
| VBUS | (Germany) |
| CRTN | |
| RMW 2002 (SRM II) | |
| RTN 1996 | (Japan) |
| SKM2 | |
| sonROAD | (Switzerland) |
| CNOSSOS-EU 2015 | |

This large number of models will not be examined here, but a summary of each of them is readily accessible in Garg and Maji (2014). Their review provides an exhaustive comparison of many of the above models highlighting similarities and differences, including source modelling algorithms and sound propagation algorithms. They concluded, as at 2014, that empirical models used in earlier road traffic noise prediction tasks have now been replaced by newer ones based on scientific principles, appending numerical methods because of increasing computer skills. They comment that newer models separate the source and propagation parts to allow independent updating of source and propagation components of the models. A summary of source-specific calculation models for road traffic noise can be found at Section L.1 of Annex 1 in ISO (2017), as in Table 26 (included below in the section describing the development of the CNOSSOS-EU mode).

The following sections discuss more details regarding post-2010 prediction modelling literature. As flagged above, the emphasis in these tends to be their application to strategic noise mapping and to mapping exposures for epidemiological studies (i.e., Applications (3) and (4) in the list above) rather than to mitigation studies or modelling whether or not limit values have been exceeded.

## STRATEGIC NOISE MAPPING & MAPPING FOR EPIDEMIOLOGICAL STUDIES

These two applications of prediction modelling are closely related topics and will be discussed together in this section. Strategic noise mapping maps noise exposure over some (usually urban) area estimating the noise exposure at the façade of dwellings (or, in some cases, several different facades of a dwelling, facilitating estimation of the numbers of people exposed to different levels of road traffic noise (an END requirement). For epidemiological studies, the task is similar, but specifically directed at assigning accurate estimates of road traffic noise exposures to all subjects in the study. In this, effort is directed at minimizing error in the assignment of exposures to subjects that minimize uncertainty (termed misclassification) to allow for estimation of epidemiologically determined exposure-response relationships

Australia lags well behind many European countries in terms of road traffic noise mapping. A very useful review of Strategic Noise Mapping is provided in a chapter by Murphy and King (2022a). The Introduction to that Chapter reads:

> '*For environmental noise research, mapping is an extremely important part of the process of quantifying and visualising noise pollution levels. Indeed, environmental noise pollution is an inherently spatial phenomenon. It varies across geographic space depending on the location of the noise source, the receiver, and the intervening obstacles (e.g., the terrain, buildings, barriers). Understanding how noise pollution varies across space, how many people it affects and how it can be mitigated are all part of the process of strategic noise mapping. The process of digitally mapping environmental noise across geographic space allows researchers and policymakers to identify locations that are subject to excessive noise levels and investigate if there are individuals residing in those areas that are affected by excessive pollution. Thereafter, steps can be taken to reduce noise levels so that public health is protected. The mapping process also allows for the identification of areas of good sound quality - often referred to as quiet areas - so that these can be protected into the future as amenity areas for rest or recreation that are free from noise disturbance. Accordingly, this chapter provides an outline of the strategic noise mapping approach developed and utilised across EU member states. Focus is placed on the EU precisely because it is the world leader in environmental noise policy and related legislation. Relevant noise mapping studies and associated research in other jurisdictions are also presented to demonstrate the wide adoption of the EU approach to assessment and mitigation of environmental noise across the globe. In outlining the strategic noise-mapping process, emphasis is placed on the principles of mapping, modelling, estimating exposure and action.'*

It should be noted that strategic noise mapping is not particularly useful if it applied to just some classifications of roadways in the system – e.g., freeways and major arterial roads. It tends to be these roadways that will have full information known about their traffic flows – either by direct/remote monitoring of flow rates, or by transport system modelling of the network. The critical issue in much strategic noise mapping, in Europe as in Australia, is the absence of data for the local roads in the transport network. This tends to be the case, even where the traffic data source for modelling is the output of transport modelling of the whole network. There have been various attempts to estimate the flows on the non-counted parts of the road network (see, for example, the 'probe' technique described earlier (Hinze et al., 2022) counting movements of vehicles from global positioning system devices (not clearly validated as yet). Morley and Gulliver (2016) reported another method to improve traffic flow and noise exposure estimation on minor roads. Quoting their abstract:

> *'Address-level estimates of exposure to road traffic noise for epidemiological studies are dependent on obtaining data on annual average daily traffic (AADT) flows that is both accurate and with good geographical coverage. National agencies often have reliable traffic count data for major roads, but for residential areas served by minor roads, especially at national scale, such information is often not available or incomplete. Here we present a method to predict AADT at the national scale for minor roads, using a routing algorithm within a geographical information system (GIS) to rank roads by importance based on simulated journeys through the road network. From a training set of known minor road AADT, routing importance is used to predict AADT on all UK minor roads in a regression model along with the road class, urban or rural location and AADT on the nearest major road. Validation with both independent traffic counts and noise measurements show that this method gives a considerable improvement in noise prediction capability when compared to models that do not give adequate consideration to minor road variability (Spearman's rho. increases from 0.46 to 0.72). This has significance for epidemiological cohort studies attempting to link noise exposure to adverse health outcomes'.*

Figure 59 illustrates a validation study in Norwich, comparing predicted AADT at 161 sites using the minor roads routing model to observe AADT.

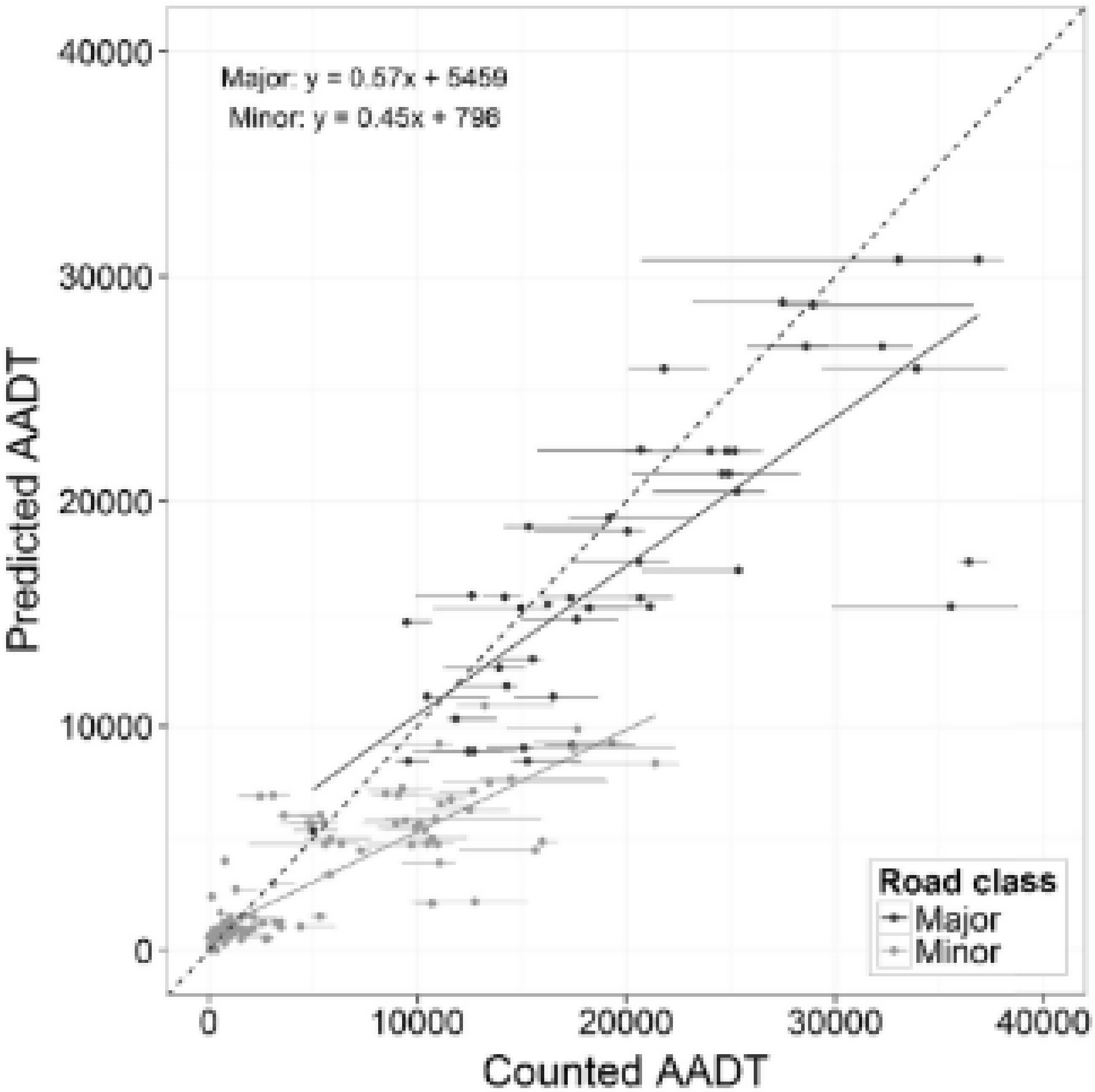



**Figure 59 Predicted AADT at the 161 count sites in Norwich using the minor roads routing model compared to observed AADT estimated from 30-min traffic counts. The median observed AADT is shown of three visits together with the minimum and maximum error bars. Sites are categorised as either being situated on a major or minor road. The dashed line represents a one-one relationship between measured and modelled values. Solid lines represent the best fit for the relationship between counted and predicted AADT split by road class (Morley & Gulliver, 2016).**

Apart from the traffic flow data availability, and the choice of traffic noise prediction model, a study of different population exposure methods was reported (Murphy & Douglas, 2018). The following is their abstract:

> *'This paper assesses four methods for estimating population exposure to road traffic noise within the terms of the strategic noise mapping process laid down in the Environmental Noise Directive (END). Employing a case study in central Dublin, Ireland, the methods – MAX, MIN, VBEB/ CNOSSOS and AVE – are tested utilising the Lden and Lnight indicators. The study first investigates the extent to which exposure estimates may vary depending on the method utilised while controlling for the noise calculation method. Second, it investigates how estimates of exposure vary depending on the calculation method used; in this case, CRTN and NMPB. The results show that controlling for noise calculation method and employing the same input data, estimations of population exposure differ substantially depending on the exposure method employed. Furthermore, the potential variability in estimated night-time exposure and the potential for under or over-estimation of the health effects of environmental noise on a given population when different methods are*

*utilised is clearly demonstrated. The results also show that the method of noise calculation employed has an effect on estimated exposure, particularly for Lden measures. Values for Lnight were found to be very similar regardless of the calculation method employed. The results therefore suggest that it is the exposure estimation method rather than the noise calculation method which has the greatest effect on population exposure estimation and should therefore be of greatest concern for understanding population health impacts and for achieving consistent results.'*

## UNCERTAINTIES in STRATEGIC ROAD TRAFFIC NOISE PREDICTION MODELLING

Several papers have addressed the uncertainty associated with the calculated noise levels in strategic noise modelling, Liepert et al. (2017) report the uncertainty of calculated noise levels and its influence on exposure-response-relationship in the NORAH-project (primarily an aircraft noise study, but with road and rail traffic too). Exposure-response relationships are often only depicted with an indication of the uncertainties in the response effect, but uncertainties also exist in the determination of exposure. The uncertainties of the calculated noise levels were estimated for road traffic noise (and other sources) to show the influence on the exposure-response-relationship. The uncertainty of the relevant calculation parameters was estimated for source emissions, transmissions paths and the receiver point for each traffic noise source for different distance classes between source and receiver. The outcome was that the uncertainty of the level values determined is between 3 and 5 dB, (depending on the transport source). They suggest that the influence of the uncertainty of the acoustic level values on the position of the lines of regression of the exposure-response relationship is only slight.

The Swiss SiRENE study (Short and Long Term Effects of Transportation Noise Exposure) was a large scale nationwide assessment of Switzerland's road, railway, and aircraft noise exposure conducted for the year 2011. One component was an opportunity for validation of the noise exposure modelling by long-term measurements (Schlatter et al., 2017). The authors noted that large-scale noise exposure modelling is used in epidemiological research projects as well as for noise mapping and strategic action planning. Such calculations should always be accompanied by an assessment of uncertainty in noise levels, both in terms of magnitude and sources.

In SiRENE. road noise modelling was performed using the sonROAD emission model and the sound propagation model of StL-86. The sonRoad emission model predicts the noise emission as a function of vehicle type, driving speed, slope, and road surface properties. StL-86 accounts for geometrical divergence, air absorption, an idealised ground effect, as well as shielding by obstacles. The reason for using a significantly older (and thus simpler, non-spectral) propagation model was primary due to computational efficiency. The decision to rely on StL-86 was supported by the fact that the propagation core of StL-86 had proven to be highly reliable in the past - the model is still recommended by the Swiss Federal Office for the Environment (FOEN)]. sonROAD requires traffic information such as speed, volume, and traffic composition. A Federal agency maintains a monitoring network and captures detailed annual traffic statistics. However, this system delivers traffic data for only a very small number of streets, whereas the noise emission model needs traffic volumes and the driven speed of all roads. Therefore, a traffic model was also used which links population census data with available information such as street width, number of lanes and road classification to predict the daily amount of traffic. Noise was modelled at façade points of all buildings in Switzerland for three different years (1991, 2001 and 2011), resulting in a total number of 5,430,000 receiver locations at 1,813,000 buildings

This SiRENE follow-up study (Schlatter et al., 2017), equipped 180 sleeping and/or living room windows with sound level meters for one week. This dataset was used to validate noise exposure modelling within SiRENE. Results were that, for the noise metric $L_{den}$, the comparison revealed a difference between the SiRENE predictions and measurements of 1.6 ± 5 dB(A) when taking all measurements into account. After removing measurement sites with noise mitigation measures not considered in the modelling, the difference to the calculation was reduced to 0.5 ± 4 dB(A). These results are shown in Figure 60 for four periods, and for hourly $L_{eq}$ as well. While the overall conclusion was that large-scale exposure modelling of SiRENE was in quite good agreements with measurements, the authors identified several sources of systematic deviations and/or scattering: position accuracy and topicality of infrastructure and building geometries; traffic modelling; acoustic source and propagation modelling; and noise measurements. See the original paper for suggestions as to how these can be improved.

There are two additional notes regarding this study:

1) The monitoring data at the 180 sites provided data regarding window attenuation of road traffic noise: see section '*Window Attenuation for Road Traffic Noise'* below, and the article by (Locher et al., 2018) described there.
2) The SiRENE study also measured the Intermittency Ratio, IR (see Noise Events discussion in this document). The current study also evaluated the prediction of the IR.

**Table 2:** Comparison of the differences (calculation - measurement) of the different datasets for four noise metrics.

| | $L_{DEN}$ | $L_{Day}$ | $L_{Evening}$ | $L_{Night}$ |
|---|---|---|---|---|
| | | [dB(A)] | | |
| $\mu_{Dataset\#1}$ | 1.6 | 2.3 | 1.7 | 1.5 |
| $\mu_{Dataset\#2}$ | 0.8 | 1.6 | 0.9 | 0.5 |
| $\mu_{Dataset\#3}$ | 0.5 | 1.5 | 0.7 | 0.2 |
| $\sigma_{Dataset\#1}$ | 5.0 | 4.8 | 4.9 | 5.4 |
| $\sigma_{Dataset\#2}$ | 4.2 | 4.2 | 4.0 | 4.5 |
| $\sigma_{Dataset\#3}$ | 4.0 | 4.1 | 3.9 | 4.3 |

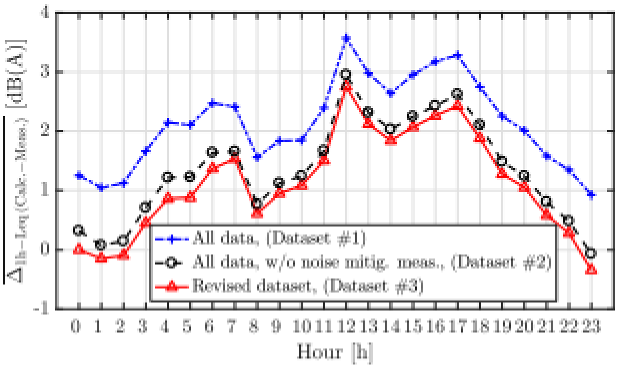



**Figure 60 Table shows comparison of the differences (calculation - measurement) of the different datasets for four noise metrics in the SiRENE validation study, and the figure shows average differences between calculated-measured one hour $L_{eq}$ (Schlatter et al., 2017).**

Thacher, Poulsen, et al. (2020) reported a h**igh-resolution assessment of road traffic noise exposure in Denmark.** They modelled road traffic noise exposure across the entire Danish population and investigated its distribution in relation to area-level socioeconomic indicators and green space. Based on the Nordic prediction method, they estimated road traffic noise for all Danish residential addresses, in total 2,761,739 addresses, for the years 1995, 2000, 2005, 2010, and 2015 at the most and least exposed façades. The latter was an unusual extension of these types of studies. Correlations between most and least exposed façades varied based on population density and building type, with the highest correlations between the most and least exposed façades found for semidetached homes and lowest for multistorey buildings. Increasing median noise levels were observed across increasing levels of higher education, lower income, and higher unemployment. A decreasing trend in median noise levels with increasing levels of green space was observed. The authors concluded that it is feasible to estimate nationwide, address-specific exposure over a long time-period. Furthermore, the low correlations found between most and least exposed façade for multistorey buildings, which characterize metropolitan centres, suggests that the most exposed façade estimation used in most previous studies predicts exposure at the silent façade relatively poorly.

An example of the application of this approach has already been reported in this document (see Cantuaria et al., 2021 in the section on Dementia).

## DEVELOPMENT of CNOSSOS-EU

With this diversity of models, it is not surprising that there were perceived inconsistencies in the results of END mapping across (European) countries (Murphy et al., 2020). This has resulted in the development of a prediction modelling system for noise mapping across Europe known as CNOSSOS-EU (**C**ommon **NO**ise a**SS**essment meth**O**d**S**) applicable to road traffic, railway traffic, aircraft, and industrial noise). Following an update of Annex II of the Environmental Noise Directive published in 2015, all EU Member States as of 1 January 2019 are required to use CNOSSOS when preparing noise maps in accordance with the END. It is to be expected that much more will be published in forthcoming years on experience with this model.

**Table 26 National and European road traffic prediction models as at 2017 (from ISO, 2017).**

Austria: Austrian standard RVS 3.02 Lärmschutz, December 1997.

Denmark, Finland, Iceland, Norway, Sweden:

— Road Traffic Noise — Nordic Prediction Method, TemaNord 1996:525, ISBN 92 9120 836 1, ISSN 0908-6692 (No longer used in Denmark, official method in the other countries).

— Nord2000 Road. New Nordic Prediction Method for Road Traffic Noise (It can be downloaded from www.delta.dk, official method in Denmark, unofficially used in the other countries)

European Union: CNOSSOS-EU Model[12]

France: French noise prediction method for road, rail and industry NF S 31-133:2011 Acoustics — Outdoor noise — Calculation of sound levels, AFNOR, February 2011. Reference software is available from http://www.setra.developpement-durable.gouv.fr/les-bibliotheques-logicielles-de-a5604.html

Germany: RLS-90. Richtlinien für den Lärmschutz an Straßen

Japan: Road traffic noise prediction model ASJ RTN-Model 2013. The technical report, published in Acoust. Sci. and Tech. Vol. 36 (2015), can be downloaded from http://www.asj.gr.jp/eng/

The Netherlands: Reken- en Meetvoorschrift Wegverkeerslawaai 2012, specifying a basic method (Standaard Rekenmethode I) and an advanced method (Standaard Rekenmethode II)

Switzerland: StL-86

United Kingdom: CRTN-88. The 18 h day time $L_{10}$ is calculated.

USA: TNM 1998: geometrical ray theory and diffraction theory - 1/3 octave band spectra

**Note**: CNOSSOS-EU uses four categories of vehicle for its road traffic noise emission modelling. See Table 27 below (plus a 5th open category). Note that CORTN modelling of emissions is based on just two vehicle categories (light and heavy…the latter >1525kg tare weight). Effective utilisation of a model with multiple vehicle classes requires that counts for each class be available. Improved precision in modelling the emissions from vehicles in a traffic stream is of little value if data on the counts for each vehicle category are not available. The fifth category has been kept open – presumably for electric vehicles – See section '*Electric Vehicles'* towards the end of this document.

**Table 27 Vehicle categories used in the CNOSSOS-EU model (Kephalopoulos et al., 2012).**

| Category | Name | Description | | Vehicle category in EC Whole Vehicle Type Approval[1] |
|---|---|---|---|---|
| 1 | Light motor vehicles | Passenger cars, delivery vans ≤ 3.5 tons, SUVs[2], MPVs[3] including trailers and caravans | | M1 and N1 |
| 2 | Medium heavy vehicles | Medium heavy vehicles, delivery vans > 3.5 tons, buses, touring cars, etc. with two axles and twin tyre mounting on rear axle | | M2, M3 and N2, N3 |
| 3 | Heavy vehicles | Heavy duty vehicles, touring cars, buses, with three or more axles | | M2 and N2 with trailer, M3 and N3 |
| 4 | Powered two-wheelers | 4a | mopeds, tricycles or quads ≤ 50 cc | L1, L2, L6 |
| | | 4b | motorcycles, tricycles or quads > 50 cc | L3, L4, L5, L7 |
| 5 | Open category | To be defined according to future needs | | N/A |

Kephalopoulos et al. (2014) (and, in more detail. Kephalopoulos et al. (2012)) outline the process behind the development of CNOSSOS-EU including emission modelling, sound propagation modelling, and assigning noise levels and populations to buildings. The methodological framework underpinning CNOSSOS-EU is based on noise assessment methods in use in Austria, Denmark, Finland, France, Germany and Sweden (Murphy & King, 2022a). The CNOSSOS-EU road source emission-model component is derived from the Harmonoise/IMAGINE project (Peeters & Blokland, 2007), while the propagation model component was derived from the French NMPB-2008 (Dutilleux et al., 2010).

In 2017 a study was conducted in The Netherlands to evaluate if the new calculation method could also be used to replace the Dutch national method. That study found errors that would lead to implausible results, and a working group from various Member States prepared a proposal for the improvement of the calculation method itself (Kok, 2019; Kok & van Beek, 2019). Many issues were raised regarding errors or lack of clarity in the model for outdoor propagation calculations, including ground attenuation and diffraction. All issues were resolved except for a remaining disparity between ground attenuation calculated by CNOSSOS-EU and by other common models (particularly ISO-9613, mainly at larger distances). These improvements in the CNOSSOS-EU calculation methods were formally adopted into the END (EU, 2020b). I am advised (pers. comm.) that the new version of CNOSSOS-EU will be, and is being, used in the current 2022 round of noise mapping.

With respect to the range of applications for prediction modelling (1) through (4) described above, Kephalopoulos et al. (2012) write: '*As CNOSSOS-EU has been designed to make cost-efficient calculations of A-weighted outdoor sound pressure levels for strategic noise maps* [Comment by this author: not noise action planning and achieving legal limits where accuracy requirements of assessment may be higher] *it is not necessarily the optimum method for other purposes…'*. Personal communications from several colleagues in Europe seems to suggest that whist they will use CNOSSOS for meeting the mapping and reporting requirements of the END, many countries are continuing to use their own prediction systems for other purposes. NMPB is used in France for other purposes besides noise mapping (assessing legal limits), and CNOSSOS-EU is largely based on the French NMPB method – and thus presumably producing similar predictions. The Dutch method (similar to ISO 9613-2) is still used in permitting and many legal processes in that country. In summary, the best advice this author has been able to glean is that: Australian authorities could use CNOSSOS (with local adaptations for emissions, road surfaces etc) for assessing legal limits and related purposes, but it is likely there may be more accurate methods available.

Given that the detailed traffic and land cover data required as input to CONOSSOS_EU may not always be available, Morley et al. (2015) assessed the feasibility using the CNOSSOS-EU road traffic noise prediction model with coarser input data in terms of model performance. Starting with a model using the highest resolution datasets, they progressively introduced lower resolution data over five further model runs and compared noise level estimates to measurements. They concluded that a low-resolution noise model should provide adequate performance for exposure ranking (Spearman's rank = 0.75; $p < 0.001$), but with relatively large errors in predicted noise levels (RMSE = 4.46 dB(A)).

## TRANEX - a MODIFIED IMPLEMENTATION of CORTN

For an epidemiological study 'Traffic and Health in London', Gulliver et al. (2015) developed a model TRANEX (**TRA**ffic **N**oise **EX**posure). This was an open-source traffic noise prediction model, which broadly follows a modified CoRTN (Department of Transport Welsh Office, 1988) method using city-wide traffic flow data, topographic layers, transport networks, building heights and a landform DTM. The modifications were for two purposes: firstly, so that the treatment of source geometry, traffic information (flows/speeds/spatially varying diurnal traffic profiles) and receptors matched as closely as possible those of the air pollution modelling being undertaken in the project; and secondly to be able to produce noise estimates at several million address locations across London with limited computing resources, from 63,000 road links. The authors explain in some detail the modifications made to the CORTN model and procedures including the model's input data, and this is not repeated here. But in describing limitations they identified: *'…not included elevated road sections…no existing noise barriers...modelled noise propagation on a flat world. We assume minor roads to have a constant traffic flow of 600 vehicles day… single value of average vehicle speed for each road… Noise estimates are universally made at 4 m. above ground…'.* They also incorporated available conversions (Abbott & Nelson, 2002) to estimate the standardised European $L_{Aeq,16hr}$ and $L_{night}$ values from hour-by-hour predictions of $L_{10}$. Gulliver et al. (2015) provided a somewhat limited evaluation (Figure 61) of results from TRANEX, not from London but from Leicester (38 x 30-minute noise measurements) and Norwich (35 x 30-minute measurements). Comparing measured and modelled 30-minute $L_{eq}$ showed modelled levels to be an average of 3.1 dB higher than measured. The authors also noted that studies comparing levels generated by prediction models with measured levels are relatively scarce. TRANEX has been developed so that it is transferable to other cities. In the UK this means that it can be applied in most areas (as long as there is sufficiently detailed information on traffic flows, composition and speeds).

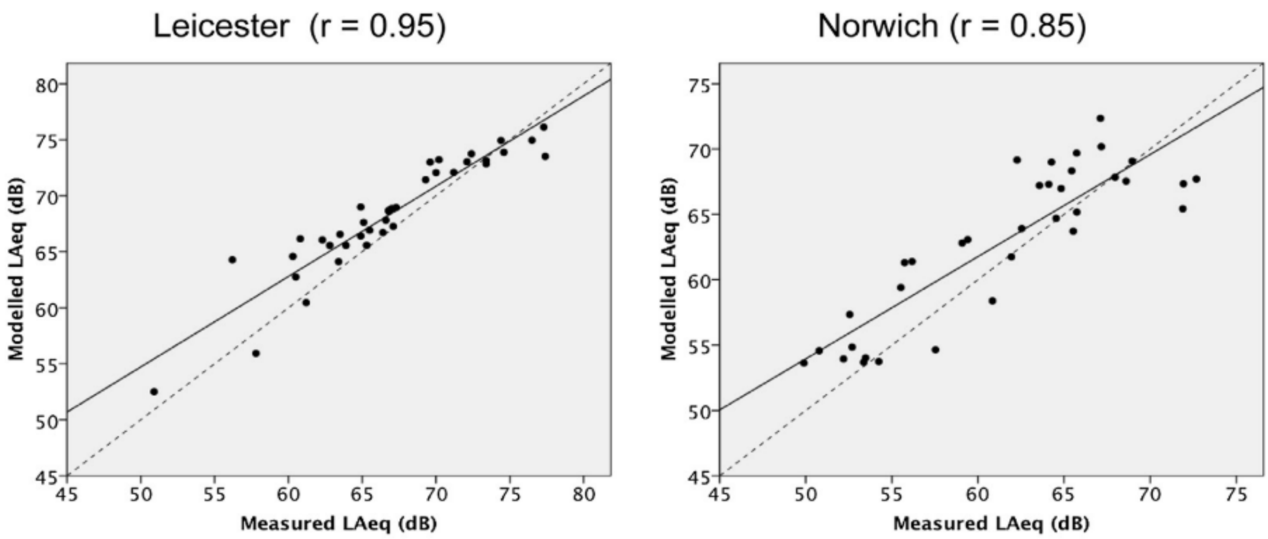



**Figure 61 Some evaluation results of the TRANEX model (Gulliver et al., 2015).**

## SELECTED COMPARISONS BETWEEN ROAD TRAFFIC NOISE PREDICTION MODELS.

### CNOSSOS-EU, Nord2000 and TRANEX

Some studies provide comparisons of the levels predicted by several different models. These tend to be inter-model comparisons which compare the levels predicted by one model with the levels predicted by others when they are applied to the same set of conditions. Each model is applied within an identical context, using the same, or appropriate, data inputs – appropriate given that differences in the formulations of the emission and propagation components of the various models require different inputs. This is a relatively weak form of validation. Better validation would be achieved by comparing predicted levels to measured levels – but, in fact, studies of direct comparisons of levels predicted by a model with levels that have been measured, are largely absent from the literature.

Khan et al. (2021) compared three road traffic noise prediction models: CNOSSOS-EU, Nord2000 and TRANEX (the modified version of the UK model CORTN, as described above). They compared the models in terms of their source and propagation characteristics, and the modelled $L_{eq}$ for 111 test cases. The results of the comparisons are complex, and the original paper needs to be examined to understand the various caveats, but some example results are shown in Figure 62 below. CNOSSOS and TRANEX reproduced $L_{Aeq}$ levels of Nord2000 within 3-5 dBA for most of the cases (CNOSSOS: 87%; TRANEX: 94%). As an extreme measure of fit, absolute differences among predicted $L_{eq}$ over the three models ranged from 0 to 7.3 dBA in all test-case conditions.

The CNOSSOS model utilised in this study had not yet incorporated the changes suggested by the Dutch as described earlier, and apparently the Nord2000 version used similarly did not include recent modifications to it.

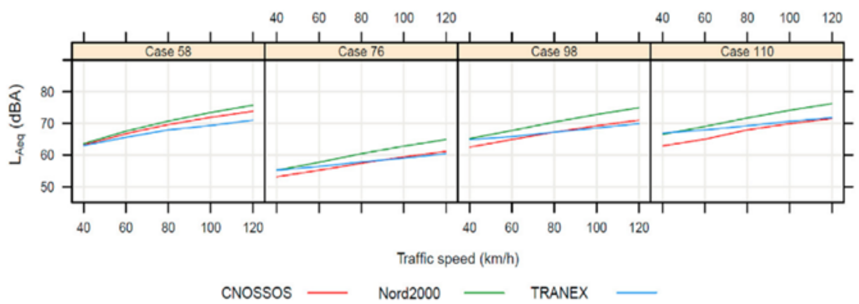
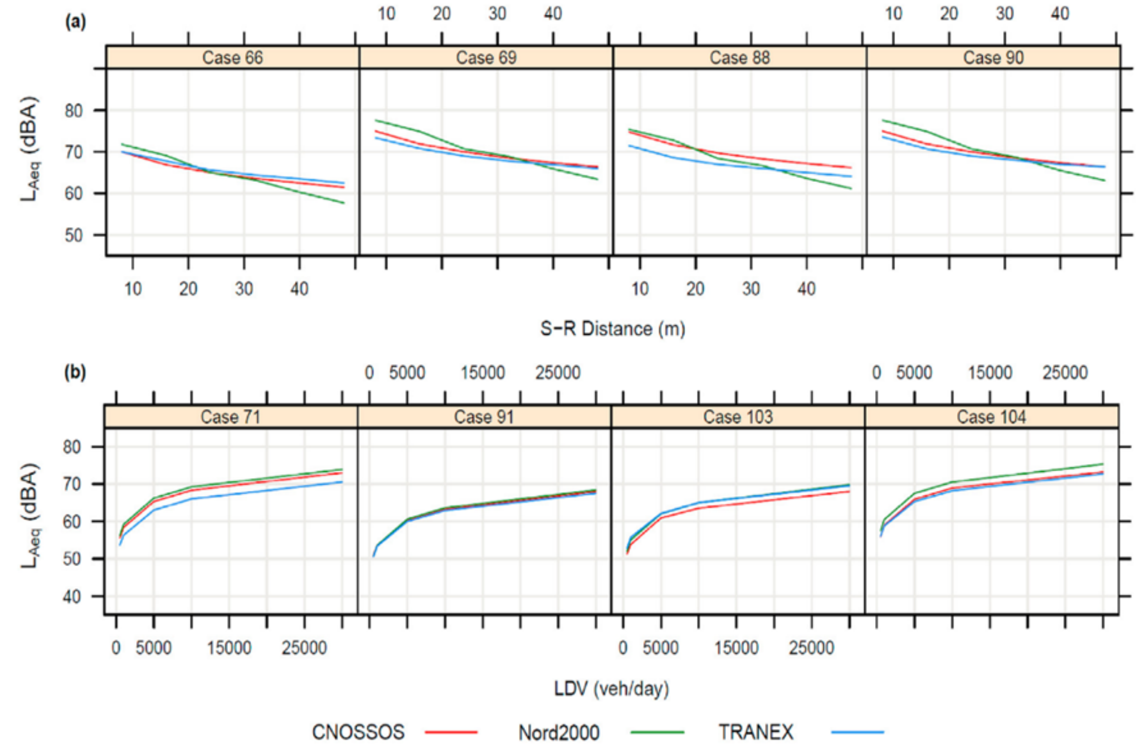



**Fig. 3.** The comparison of predicted $L_{Aeq}$ of CNOSSOS, Nord2000 and TRANEX at varying traffic speed for the selected test cases having similar geographical setting. The above results are for the "with building" scenario: Sound propagation over the straight road with building reflections. Note: Case 58: LDV = 10,000 veh/day; Case 76: LDV and HDV = 500 veh/day; Case 98 and 110: LDV and HDV = 5000 veh/day. The S − R distance for all the above cases is 10 m.




The comparison of predicted $L_{Aeq}$ of CNOSSOS, Nord2000 and TRANEX at varying (a) S − R distance (m), and (b) LDV (veh/day) for the selected test cases having similar geographical setting. The above results are for the "with building" scenario: Sound propagation over straight road with building reflections. Note: Case 66: LDV and HDV = 10,000 veh/day, speed = 70 km/h; Case 69: LDV and HDV = 10,000 veh/day, speed = 120 km/h; Case 88: LDV = 10,000 veh/day, speed = 70 km/h; Case 90: LDV and HDV = 10,000 veh/day, speed = 120 km/h; Case 71: speed = 70 km/h; Case 91: speed = 40 km/h; Case 103: speed = 40 km/h; Case 57: speed = 70 km/h. The S-R distance for test cases in (b) is 10 m.

**Figure 62 Comparison of levels predicted by CNOSSOS Nord2000 and TRANEX for randomly selected test cases (from Khan et al., 2021).**

## CNOSSOS-EU and CRTN-TRL

Faulkner and Murphy (2022) recently experimented with comparing CNOSSOS-EU and CRTN-TRL[68] traffic noise prediction model application in Ireland. Comparisons were based on both roadside measurements and on population exposure estimation analysis. Results from the (very limited) roadside measurement experiment (Table 28) indicated that the CNOSSOS-EU model predicts measured levels more accurately compared to CRTN-TRL (under the very limited test conditions of the study: e.g., only at 7.5m and 30m from the centre of the measured lane). Population exposure comparisons using the two noise models separately were used to estimate the noise exposures for the City of Dublin, some 122 km$^2$ with 2031 km of roadways. Comparisons of exposure estimates from the two models is shown in Figure 63. Implementation of the CNOSSOS-EU road traffic prediction model would substantially increase estimates of population exposure to road traffic noise levels >55 dB(A) $L_{den}$ and >50 dB(A) $L_{night}$ compared to if CRTN-TRL was used for these estimates. While it is useful to know that there are inter-model differences, there is no light shed on what the causes of them might be.

**Table 28 Comparisons of road traffic noise level measurements (LAeq,1h levels, 15:15h to 16:15h on weekday from a small roadside measurement experiment with levels predicted by each of: CRTN-TRL/CNOSSOS-EU models (Faulkner & Murphy, 2022).**

Roadside Measurements (R108) and Modelling Results for CNOSSOS-EU and CRTN-TRL Method 2 Traffic Flow Analysis - $LA_{eq}$ dB(A).

| Microphone location | Microphone height | Sound Level Meter (SLM) | CNOSSOS-EU Model | CNOSSOS-EU differential | CRTN-TRL model | CRTN-TRL differential | CNOSSOS-EU/CRTN-TRL differential |
|---|---|---|---|---|---|---|---|
| ***West Lane*** | | | | | | | |
| 1 | 1.5 m | 73.9 | 71.9 | −2.0 | 69.4 | −4.5 | −2.5 |
| 2 | 1.5 m | 73.8 | 72.0 | −1.8 | 69.5 | −4.3 | −2.5 |
| 3 | 1.5 m | 73.9 | 72.0 | −1.9 | 69.6 | −4.3 | −2.4 |
| 4 | 1.5 m | 74.4 | 72.4 | −2.0 | 70.2 | −4.2 | −2.2 |
| ***East Lane*** | | | | | | | |
| 5 | 1.5 m | 72.6 | 71.9 | −0.7 | 69.7 | −2.9 | −2.2 |
| 6 | 1.5 m | 72.8 | 71.9 | −0.9 | 69.3 | −3.5 | −2.6 |
| 7 | 1.5 m | 72.6 | 71.9 | −0.7 | 69.6 | −3 | −2.3 |
| 8 | 1.5 m | 72.6 | 72.4 | −0.2 | 70.4 | −2.2 | −2 |
| **Propagation** | | | | | | | |
| 9 (30 m) | 1.5 m | 64.8 | 65.9 | 1.1 | 60.1 | −4.7 | −5.8 |
| 10 (30 m) | 4 m | 66.3 | 65.7 | −0.6 | 61.5 | −4.8 | −4.2 |

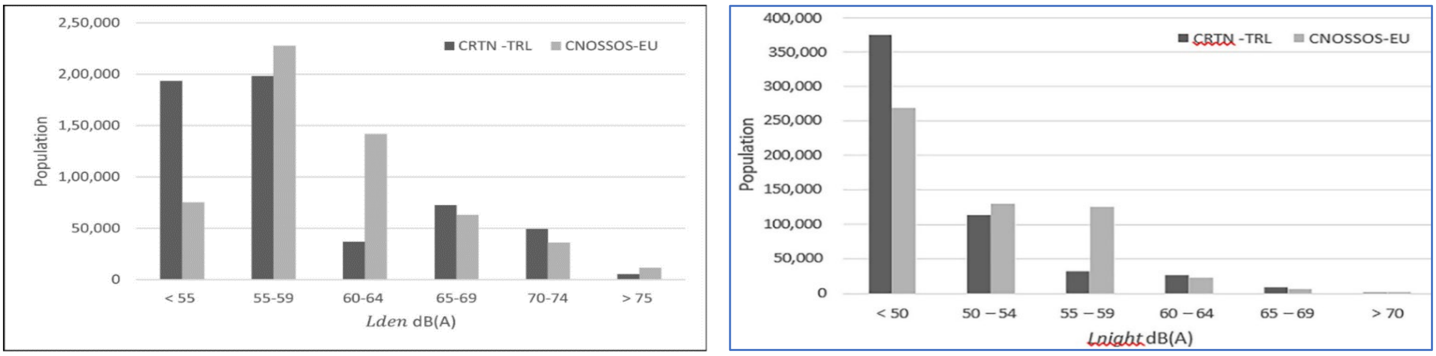



**Figure 63 Population exposures estimated for END compliance in Dublin City based on city-wide traffic noise prediction modelling (left: $L_{den}$, right $L_{night}$) – comparing the results from two modelling systems: CRTN-TRL/CNOSSOS-EU (Faulkner & Murphy, 2022).**

[68] In Ireland, road traffic noise has been calculated using the UK's CRTN method (also called CORTN) (Department of Transport Welsh Office, 1988).

## NORDIC PREDICTION MODEL SP48, CORTN, GERMAN MODEL RLS-90, and SWISS MODEL STL-86

Another comparative study had been reported earlier, from Santiago, Chile (Suarez & Barros, 2014). It is included here because it, too, used CoRTN as one of the models compared. Again, while it was the city-wide noise exposure that was modelled, noise measurements ($L_{eq,\ 15min}$) were available for 52 points. While the representativeness of these points was unclear in the paper, the authors reported the absolute differences between measured and modelled levels, for each of the four different prediction models used (Figure 64). Suarez and Barros (2014) conclude that, for the German and Swiss models RLS-90 and STL-86 respectively, over 94% of the measured/modelled levels showed a deviation less or equal to 3 dBA. It is relevant to note that in the RLS-90 model, over 50% of the differences between the measured and modelled values were less than 1 dBA. CORTN did not perform as well.

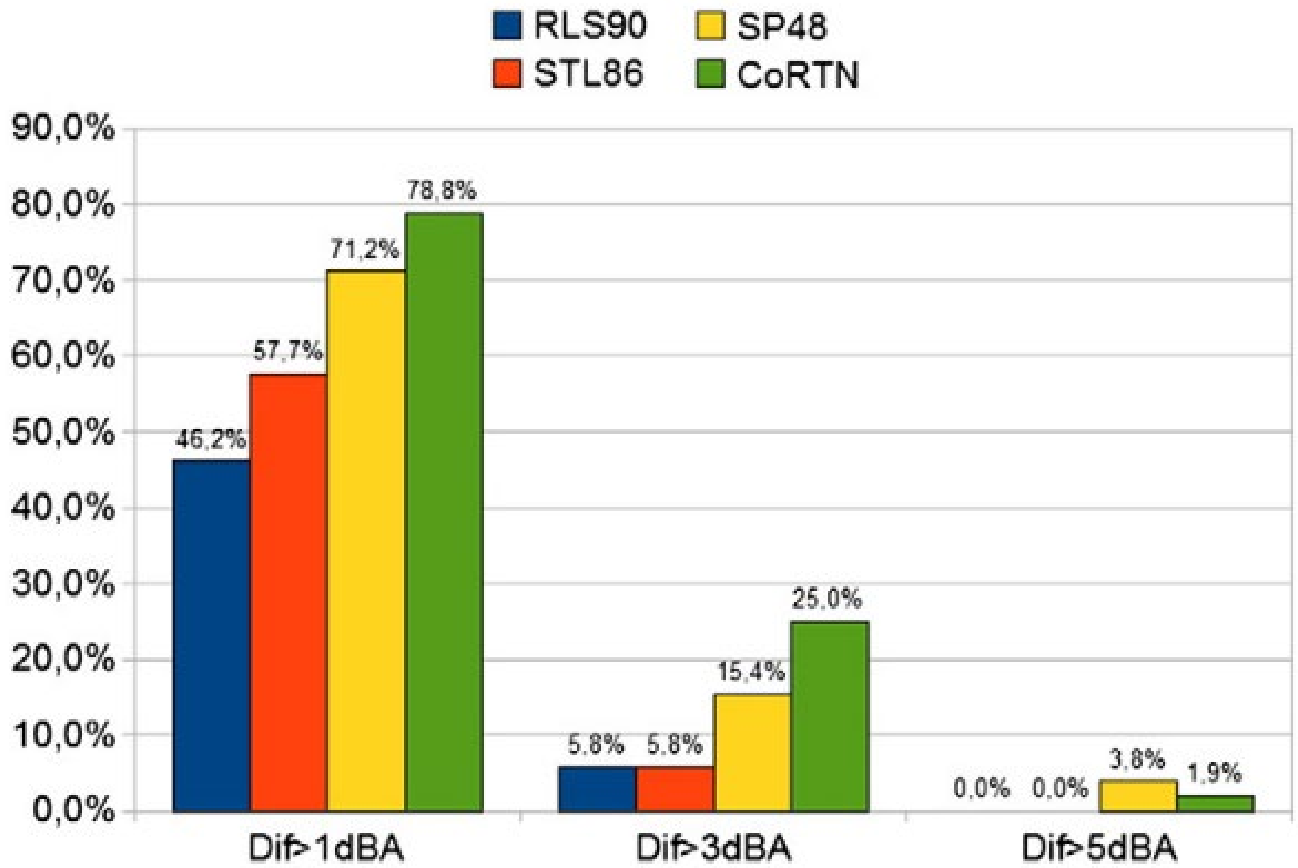



**Figure 64 The percentage of the measurement points (n=52) with absolute differences greater than 1, 3 and 5 dBA between measured and modelled values (Suarez & Barros, 2014).**

## COMMERCIAL TRAFFIC NOISE PREDICTION PACKAGES

Traffic noise predictions are usually undertaken using commercial software programs. These have one or more prediction models included in the software, each with embedded algorithms for noise sources and propagation relevant to the countries in which they are used, or some default values that can be applied. Most now also have the CNOSSOS-EU method included in software. Garg and Maji (2014) noted a list of commercial software packages available for traffic noise predictions at that time: CadnaA (Datakustik, Germany); Sound Plan (U.K.); IMMI (Wölfel); Mithra-SIG (CSTB); NovaPoint (Vianova Systems); and Predictor-LIMA (B&K, Denmark). This list could presumably beextended, but this review has not conducted a specific search for additional packages. Three (arbitrarily selected[69]) commercial packages are shown in Figure 65 below. Note the ability, within the packages, to select from a wide range of different traffic noise prediction models – a function presumably of different national model preferences even when using an internationally available commercial package.

The commercial packages integrate the noise modelling with GIS inputs and outputs. Murphy and King (2010) had noted that, overall, the noise software packages do not compare positively in terms of the mapping techniques available in commercial GIS packages. In particular, the ability of GIS packages to deal with numerous types of spatial data far outweighed (or at least they did so at that time) that available within the noise prediction software. As a reflection of this, most commercial software packages offer import/export functionality to take advantage of the greater ability of GIS to manipulate spatial data in a more sophisticated and customised manner.

There is only a little published on the uncertainties associated with predictions based on noise prediction models implemented within commercial packages, and there appears to be considerable difficulty in doing so. Over a decade ago, King and Rice (2009) warned:'*…it is worth noting that although each software product available today may follow certain standards they may not consistently yield the same* results', and noted a UK paper (Hepworth, 2006) that compared CRTN-predicted levels from five different software packages with those that were manually calculated, revealing variability across the packages. The black box approach, where the manner in which standards are implemented in commercial software packages cannot explicitly be explored, rarely make it possible to determine the cause of such variation. This observation applies to the various comparison studies reported above.

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[69] While there is no intent here to promote any particular package, it is noted from *Noise News International, 21 October 2022,* that SoundPLAN, has released version 9.0 of its software, SoundPLANnoise, for assessment and reduction of noise in buildings and outdoors. SoundPLANnoise 9.0 has a range of new features to model levels and noise dispersion. Color-coded maps are created, which visually demonstrate where the noise comes from and how it spreads

## THREE ARBITRARILY SELECTED EXAMPLES OF SOFTWARE PACKAGES FOR STRATEGIC NOISE MAPPING (including road traffic noise)

*Note the ability to choose from a wide range of traffic noise prediction models within the software packages*

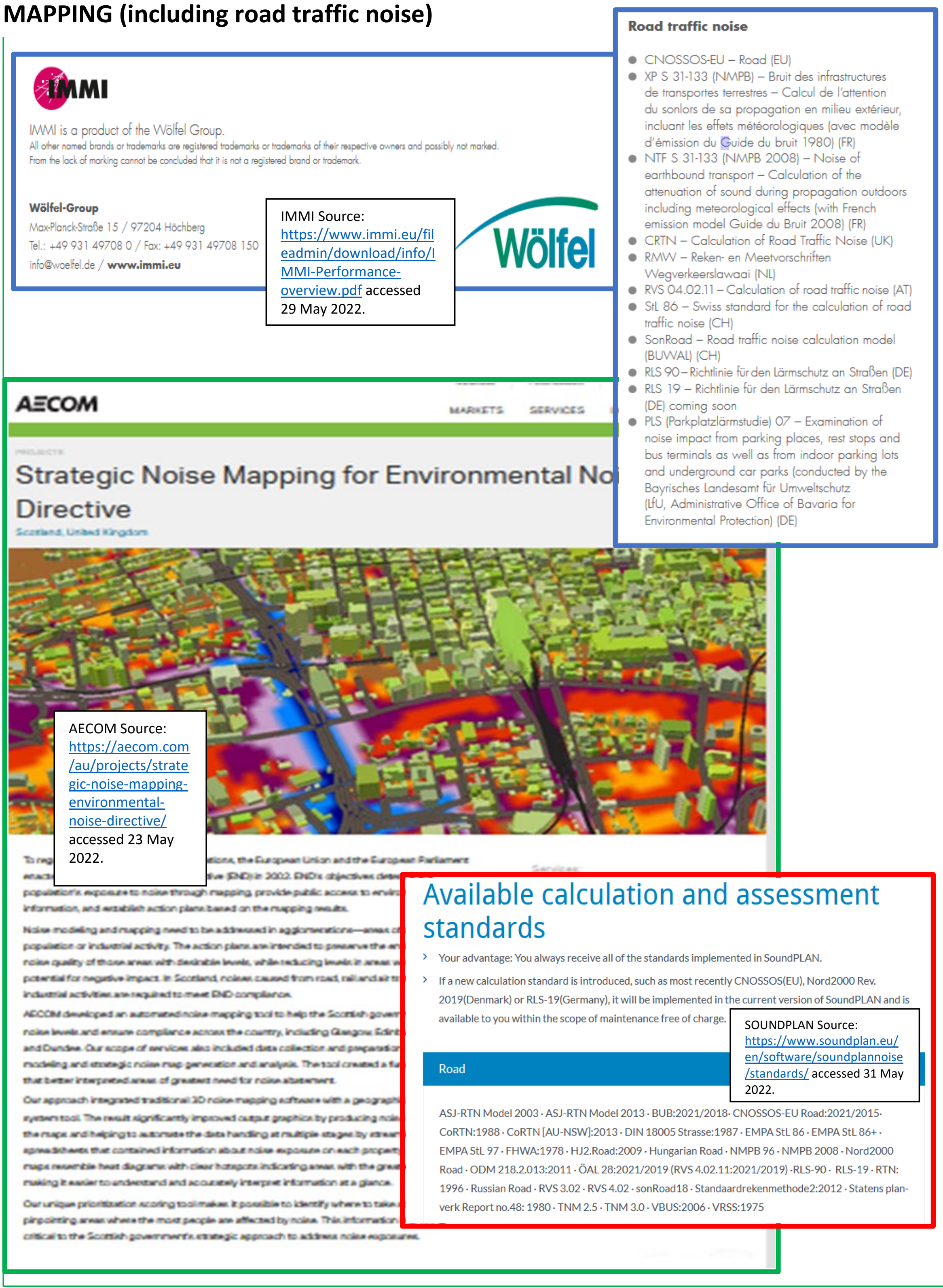


**Figure 65 Some arbitrarily selected examples of commercial software packages.**

## COMPARISONS of the SAME PREDICTION MODELS IN DIFFERENT COMMERCIAL PACKAGES

### CadnaA and SoundPLAN implementations of CORTN88

One Australian study used results from three large road projects, to conduct a comparison of the (Karantonis et al., 2010). This was a useful study comparing plots, selected point predictions, and predictions to measurements at 21 data points ($L_{Aeq,1h}$). There was generally a good correlation of the CORTN88 predictions between the CadnaA and SoundPLAN implementations (the paper noted that, at that time, SoundPLAN was used extensively in Australia). Examination of predicted noise contours showed that the level differences between the two packages tended to fall mostly within 0.5-1dB(A), with only a few isolated areas where differences reached 1.0- 2.0dB(A). Spot predictions from the two packages had an $r^2$ of 0.96 and 0.97 for $_{LAeq,1h}$ and $L_{night}$ respectively. Comparison of predicted CORTN88 levels with measured levels yielded an an $r^2$ of 0.86 and 0.90 for CadnaA and SoundPLAN respectively. The authors suggested these all fell well within the expected accuracy of the CoRTN88 noise algorithms.

### CadnaA, SoundPLAN, and FHWA (TNM software) implementations of TNM

Canadian authors, Sun et al. (2014), presented results of traffic noise modeling using the TNM algorithm method in Cadna/A (version 4.4) and SoundPLAN (version 7.2) packages and compares both to noise prediction results from FHWA TNM (version 2.5) software package. Comparisons were predictions of the exposures of the first row of buildings on three typical road projects. Their (somewhat limited) observations were that both softwares offered convenient interfaces, and that SoundPLAN predictions more closely corresponded to TNM results than did Cadna/A.

## EVALUATIONS/COMMENTARIES on CORTN in AUSTRALIA

A recent series of papers by Peng and co-workers has focussed on CORTN modelling in Australia. Their commentary (Peng et al., 2017; Peng et al., 2018) trace the origin of the CORTN model in the UK, its application and evolution in Australia and raises important issues with respect to the accuracy of the prediction model. CoRTN is used regularly in environmental assessments to identify potential noise impact from road infrastructure projects. The issues they raised include:

- CORTN was developed on $L_{A10}$ predictions, and the transformation between these and $L_{Aeq}$ is not always simple/straightforward
- the fixed correction of -1.7dB that has generally been applied to model estimates in Australia since 1983 was inappropriate
- the heavy vehicle fleet in Australia is changing, with introduction of heavier vehicles and more axles, but CORTN uses only one heavy vehicle category
- prediction performance in free-flowing conditions can be improved a little by using one light and six heavy vehicle categories for noise emission (Peng et al., 2019b) borrowing a mix of ASJ-RTN and Nord2005 source emission models. This set of emission models was tested in CORTN predictions compared to measurements at 41 locations, all with 40m uninterrupted propagation adjacent to seven roadways in NSW (four arterials, one motorway and three rural freight routes.
- CORTN does not include heavy vehicle engine braking noise for down-hill descent, nor adequately handle predictions where there is an uphill grade
- road surface corrections are dependent on traffic mix and on road temperature.
- The CORTN prediction model is unsuitable along urban and interstate freight routes with high numbers of heavy vehicles - there is observation that interstate freight routes have very different patterns and mixes of vehicle types. The prediction performance of CoRTN is highly sensitive to traffic mix and speed. See Figure 66.

The general observations in the paper appear reasonable, but there has not been direct verification of the emission models for six heavy vehicle categories – nor testing predictions against measured levels outside of the small number of roads (seven) utilised in this study. There also would have to be concern as to the widespread availability of a breakdown of traffic flows into so many vehicle classes for the application of such a modified model. However, it is a good base for further examination/improvement of this model. Peng canvasses the same work in his thesis (Peng, 2020), and additionally, using the same data set of NSW roads, suggests two supplementary noise indicators, $\Delta L_{\text{day-night}}$ and $\Delta L_{\text{emergence}}$. that potentially may reflect differences related to heavy vehicle events even where standard noise level indicators are identical. However, their testing was only in terms that they met one criterion essential for a supplementary noise indicator, viz. they were poorly correlated with conventional noise indicators.

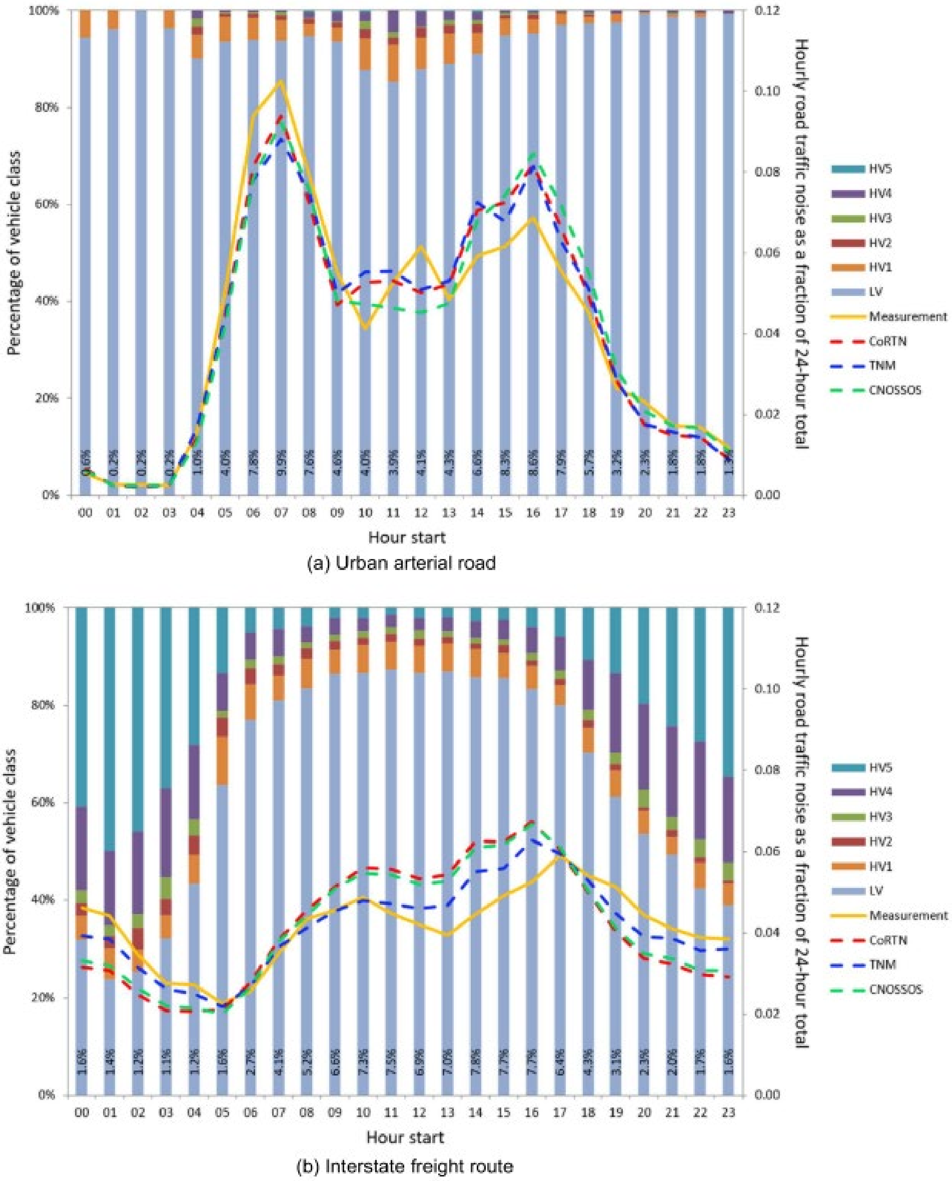


**Figure 66 Hourly traffic flow and noise contribution over a 24-hour day along an (a) urban arterial road in Western Sydney and (b) interstate freight route through the Mid North Coast of New South Wales. From Peng et al. (2019a).**

## METHODS for CALCULATING $L_{AFmax}$ USING the NORDIC PREDICTION METHOD, NORD 2000 and CNOSSOS-EU

In Sweden the use of maximum A-weighted sound level ($L_{AFmax}$), in combination with equivalent A-weighted sound level over 24h ($L_{Aeq24h}$), has been used for noise regulation for many years. The regulation was turned into legislation in 2015 for new housing development plans. The author claims the maximum level gives a good indication of sleep disturbance for residents in exposed dwellings, especially if combined with number of loud passages or events. In areas with little traffic the energy equivalent level can be very low, whereas the maximum level just depends on an individual vehicle rather than on the number of vehicles. The Swedish legislation defines maximum noise level as 'the loudest vehicle, with time weighting 'Fast', calculated as a free field value'. When calculating noise levels from road or rail traffic, there is no knowledge of individual vehicle properties. In the prediction methods used in Sweden, the calculation of maximum level is therefore based on statistical data such as the standard deviation of measured sound levels for each vehicle type at a certain speed. In the paper (by Genell & Ögren, 2019) some different methods to calculate maximum noise levels are presented, using Nordic methods as well as the common CNOSSOS-EU method, as well as their impact on results**.**

This is the only paper located that includes any estimation of maximum levels from vehicle pass bys.

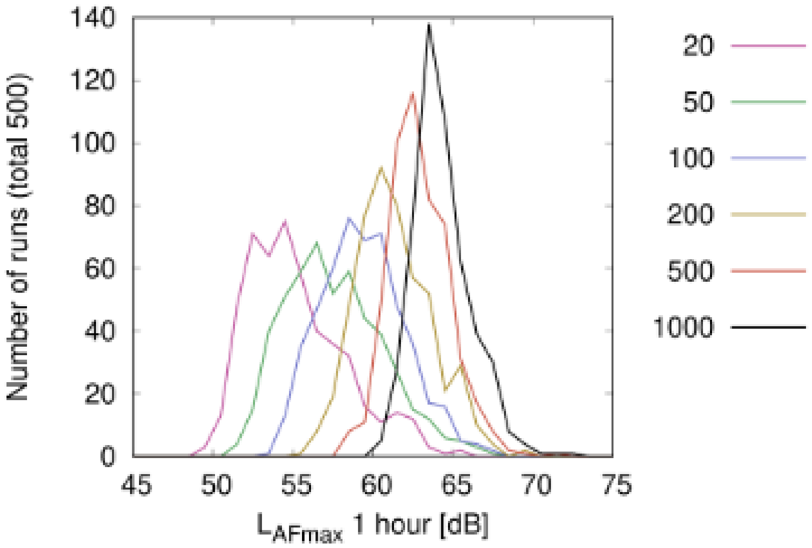


**Figure 67 Histogram of the maximum level $L_{AFmax}$ for 500 one-hour simulations. Hourly traffic was simulated from 20 to 1000 vehicles per hour. Histogram bins are 1 dB intervals (Genell & Ögren, 2019).**

## 3D NOISE MAPPING

3D noise mapping is an appropriate extension of conventional urban noise modelling where exposure estimates are required at the facades of multi-storey buildings (Figure 68). There is an excellent summary of the status of 3D noise mapping available (which would be difficult to improve upon) included as an introduction in a journal article (Stoter et al., 2020). It stands alone and is quoted in full below (Citations within the quote are not included in the Reference List for this scoping review.

> '*In recent years, various noise mapping software has been developed to simulate and visualise noise levels in 3D, e.g., CadnaA (Datakustik GmbH, 2014), Geomilieu (DGMR, 2014), SoundPlan (SoundPLAN GmbH, 2014), etc. The customised LIMA software (SoftNoise GmbH, 2011) was used in the 3D noise mapping for Hong Kong (Law, Lee, Lui, Yeung, & Lam, 2011). These noise mapping tools are commercial products and often support a limited number of input data formats. In addition, their implemented methodology is usually a 'black box' to the users and cannot be easily modified. These tools are mainly developed to simulate situations outdoors. Several open-source efforts from the academic community, on the use of 3D city models to analyse how urban citizens are harmed by noise pollution, have also been reported (Kurakula & Kuffer, 2008; Law et al., 2011; Lu, Becker, & Löwner, 2017; Pamanikabud & Tansatcha, 2009; Stoter, De Kluijver, & Kurakula, 2008; Wing, Kwan, & Kwong, 2006). Stoter et al. (2008) and de Kluijver and Stoter (2003) discussed at length the importance of using 3D noise maps for noise impact studies and produced a 3D noise map using 3D city models for obstacles in the noise propagation. Ranjbar, Gharagozlou, and Nejad (2012) used 3D city models to investigate the impact of traffic noise on high rise buildings in Tehran. They tested the model with noise barriers of different heights at different distances from the edge of the highways to determine an effective*

*approach to reduce the traffic noise in the area. Zhao et al. (2017) developed a methodology for 3D road traffic noise mapping for the city of Singapore utilising unstructured surface meshes for buildings and roads. Their work follows the UK standard CRTN (Calculation Road Traffic Noise) for noise assessment. Cai, Yao, and Wang (2018) proposed a methodology…. Details of the Dutch regulation that prescribes how noise from traffic on observation points should be calculated. This prescribed method is further explained in Section 3.2. Adapted from [http://wetten.overheid.nl/](http://wetten.overheid.nl/) to compute large scale noise maps in 3D on a supercomputer to reduce the computation time for large urban areas. Their methodology covered the noise prediction modes together with the parallel programming algorithm implemented on a supercomputer. Another recent attempt in using the 3D city models for mapping noise is the work of Bocher, Guillaume, Picaut, Petit, and Fortin (2019), which describes an open-source implementation of a 3D noise mapping tool in a GIS software OrbisGIS as a plugin: NoiseModelling. Their method is based on a profile of the French national method 'NMPB-08'. The work suggests that a similar approach can be used for other national or international noise mapping standards. At present, cities like Hong Kong (Law et al., 2011) and Paris (Butler, 2004) are using 3D city models for noise mapping. Efforts are also made in the BIM-GIS domain to integrate BIM and 3D GIS data to combine traffic noise calculations for outdoor and indoor environments (Deng, Cheng, & Anumba, 2016).*"

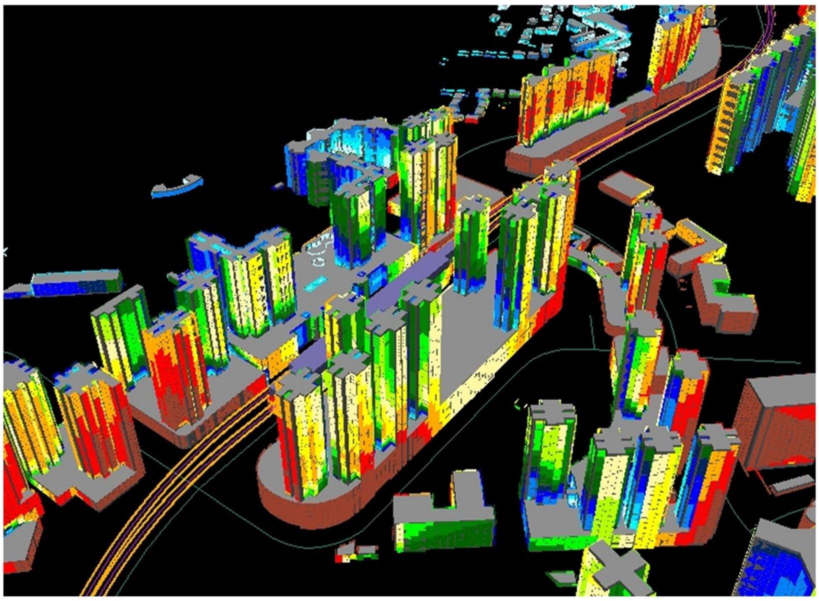

**Figure 68 An example of 3D noise mapping of high-rise residential facades in Hong Kong (copied from: Brown et al., 2015).**

## OTHER NOISE MAPPING

A different urban traffic noise mapping approach, based on road typology and noise measurements from a small number of monitoring stations representing each type of road, was reported by Zambon et al. (2018). It is based on the idea that a limited number of real-time noise measurements could be used to build up a noise map that would be representative of a large area. The idea is the basis of the Dynamap project - co-financed by the European Commission through the Life+ 2013. The method relies on identifying sets of roads that display similar temporal noise profile over an entire day – with only one monitoring station required to measure noise levels for each member of the set. The approach, designed for real-time monitoring and prediction of traffic noise across large urban areas is illustrated in Figure 69 below (from Zambon et al., 2018).

A somewhat-related street-category approach was used (Ballesteros et al., 2022) to produce a noise map for the City of Cuenca in Spain. In their method, the authors categorized all roads in the city into just six categories: from national roads through to walking streets. Mean sound power and the temporal behaviour of sound levels are allocated to each street from its characteristics and time profiles measured with semi-permanent noise monitoring systems on carefully selected examples of the six street typologies. The claim is that this reduces the measurement cost but can maintain the accuracy of urban noise maps. One category is national highways, but not included as no-one lives along these roadways, the other six are: major distributors, streets that lead to regional roads, other roadways, local streets, and walking streets (the classification is clearly particular to this type of Spanish town).

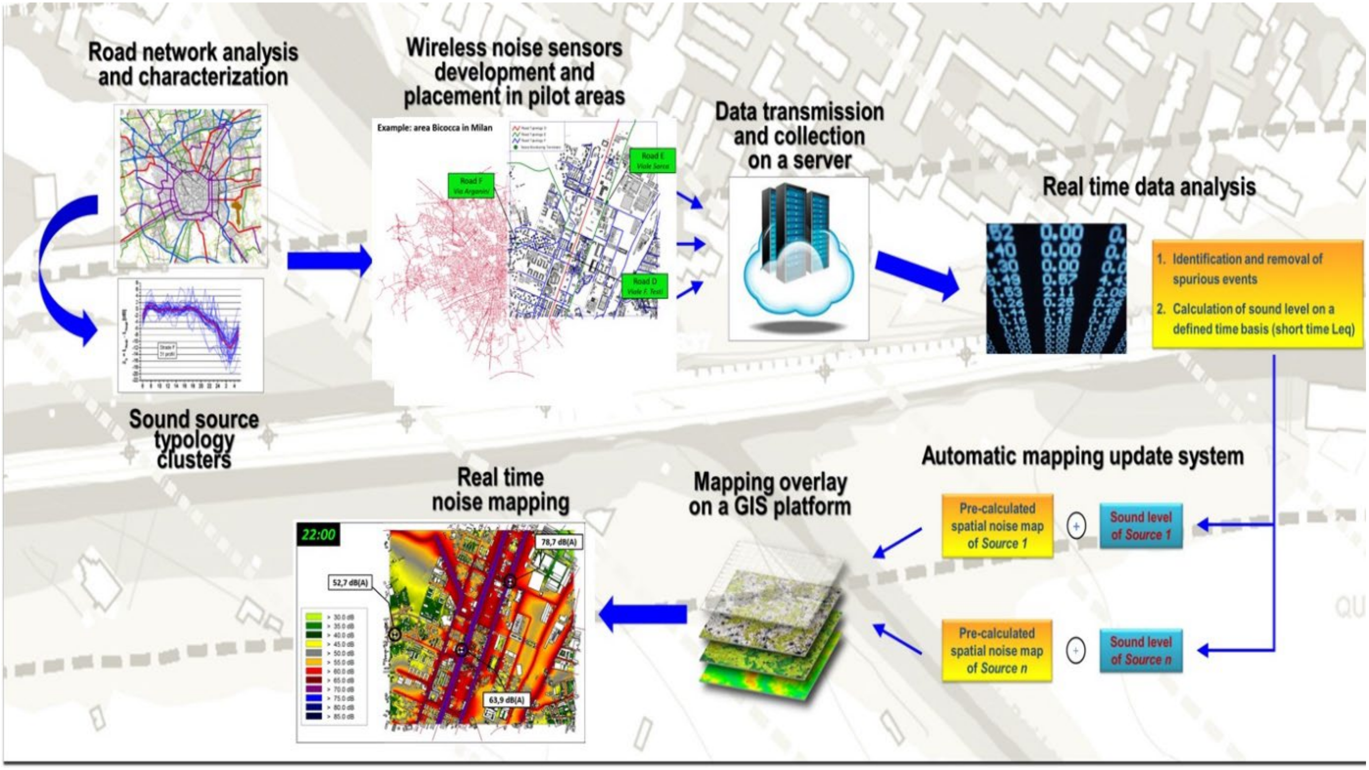


**Figure 69 Pictorial of a pilot approach towards dynamic noise mapping across part of an urban area of the City of Milan (Zambon et al., 2018), based on a limited number of road typologies.**

Apart from being of some interest as a low-cost approach to strategic noise prediction modelling, this example is mentioned here because Ballesteros et al. (2022) reported measurement results for each of the six road categories (Figure 70), and there are two observations from these that relate to material in other sections of this document

.

a) While the $L_{Aeq}$ of most of the roads track fairly closely to each other (not more than 10dB apart throughout the 24 hours) the Noise Climate ($L_{10} – L_{90}$) clearly differentiates the different categories of roadway. There are some (still to be explored) parallels between this and the suggestion in papers by Peng and others of how two supplementary noise indicators, $\Delta L_{\text{day-night}}$ and $\Delta L_{\text{emergence}}$ also differentiate roadways of different functions (see sections '*Roadways Differentiated by Functional Classification'* and '*Recent Evaluations/Commentaries on CORTN in Australia*' above.

b) The third plot in Figure 70 shows that most of the street typologies generate $L_{01}$ levels that are generally similar to each other (other than the walking street, Type 7), with only a small reduction in that metric during the night hours. It is indicative (though again this needs more exploration - see section '*Noise Events'*) of why use of a 'peak metric' such as $L_{01}$ is problematic as a limit metrics (simulation modelling presented in the *Noise Events* section had previously illustrated this for $L_{Amax}$). Simply, it tends to be similar along most roadways in urban areas, and therefore is of little use as an indicator.

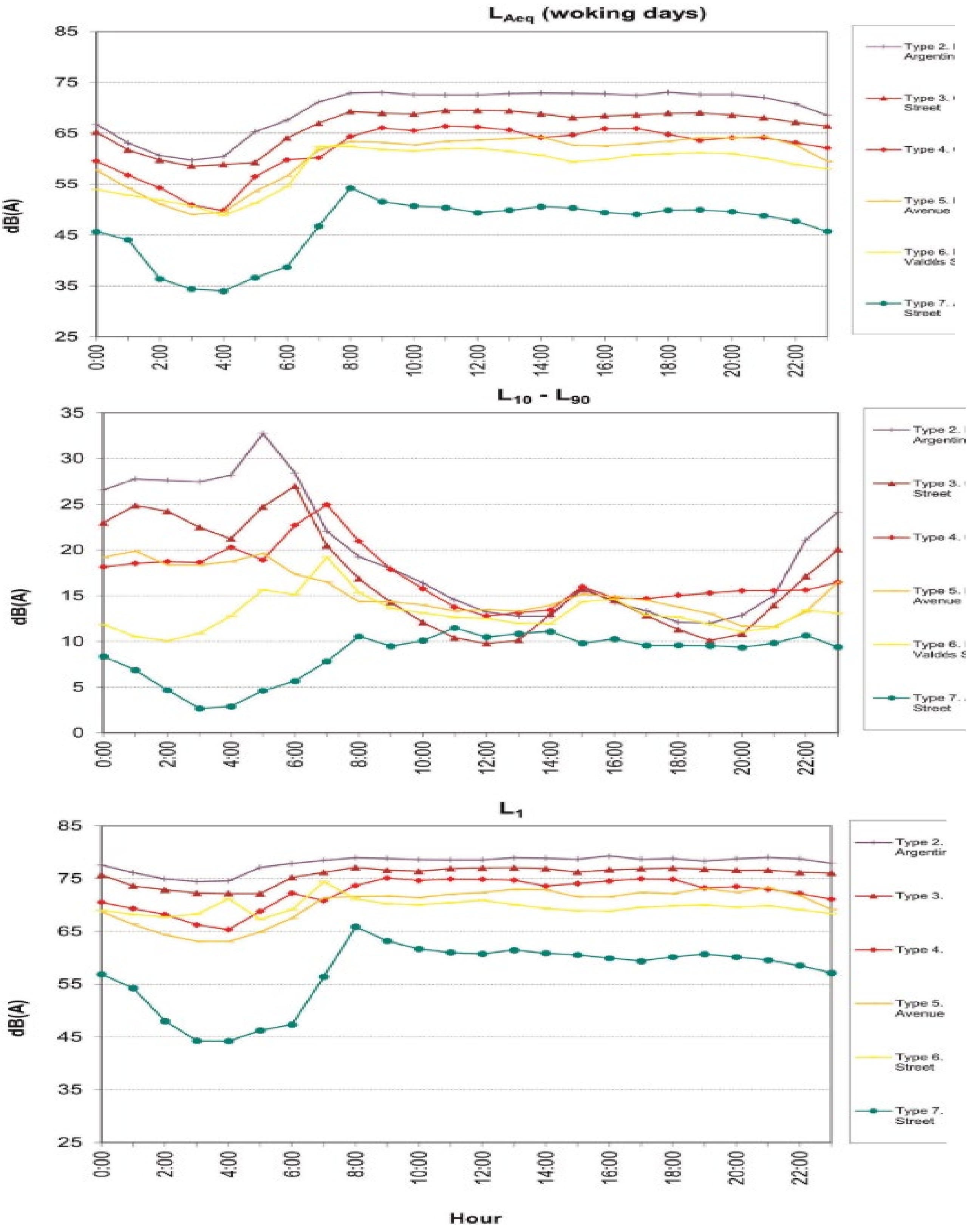


**Figure 70 Distribution throughout the day and night of three road traffic noise metrics ($L_{Aeq}$,, Noise Climate and $L_{01)}$) for each of the six typologies of road identified in Cuenca (Ballesteros et al., 2022).**

# ELECTRIC VEHICLES

Transition from internal combustion propulsion on road vehicles to electric propulsion will have effects on a range[70] of road traffic noise matters. This potential was already identified in the 2011 NSW Road Noise Policy where it was suggested, *inter alia*:

> *Gains anticipated from tougher noise emission limits from individual motor vehicles through Australian Design Rules are likely to be limited to a few decibels. More significant gains may come from:*
>
> - *reducing noise from tyres*
> - *the take-up of electric vehicles with their quieter electric motors*
> - *…*

There does not appear to be documented evidence of this in Australian as yet, presumably because of the very low rate of transition to electric vehicles (EVs) to date.

However, there is a range of literature from Europe over the review period, most concerned with the emissions from EVs, and required change to prediction modelling. Rather than reviewing each of them, the reader's attention is directed to a larger Technical Report by Pallas et al. (2020). It is most useful as it provides information on the uptake of EVs (to 2018), a succinct summary of current literature on EVs, driving behaviour of EVs, and what is needed for their incorporation into traffic noise prediction models. They describe the LIFE E-VIA project in France which aims to tackle noise pollution from road traffic noise, in a future perspective involving a consistent portion of electric and hybrid vehicles. The project involves several actions: the first focussed on noise emissions of EVs; the others on quiet pavement and tyre technologies because of the dominance of this in EV emissions, and the likely changed driving behaviours induced by EV specificities. These place a sharp focus on rolling noise of EVs in the consideration of electric vehicles in the noise prediction methods. The authors note the dynamism of the electric vehicle market is reflected both in terms of technological innovations and in the increase in the number of vehicle models available by manufacturers. In the European area, about 1.4 million light EV vehicles were in circulation at the end of 2019, with a market share of new registrations of EVs reaching 3% in 2019. This European market is by far dominated by battery-electric vehicles (BEVs) and plug-in hybrid electric vehicles (PHEVs) of the passenger car category, which is expected to stay the dominant market in the next years. The international outlook for EV fleet is estimated to reach between 15% and 30% of the global vehicle fleet by 2030 (Figure 71). The authors also provide data on electric light commercial vehicles and buses.

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[70] One of these is pedestrian safety related to interaction with the quiet electric vehicles. To address this problem, the European Commission Delegated Regulation (EU) 2017/1576 of 26 June 2017 mandated that, since 1 July 2021, all new types of electric and hybrid cars must be equipped with the acoustic vehicle alerting system (AVAS). This latter will automatically generate a sound for speeds lower than 20 km/h, and during reversing. ANNEX VIII of this regulation sets out measures concerning the Acoustic Vehicle Alerting System (AVAS) for hybrid electric and pure electric vehicles (EU, 2017)

For an internal combustion vehicle, the engine contributes significantly to the overall noise made by the vehicle. Particularly at low speed, the difference between EVs and ICEVs can be over 10 dB, whereas the noise made by the tyres on the road surface becomes dominant above 20 to 30 km/h. At the speed of 10 km/h, an electric vehicle may not be detected until it is less than 5 meters away, whereas, under the same conditions, a vehicle with an engine can be heard up to 50 meters away

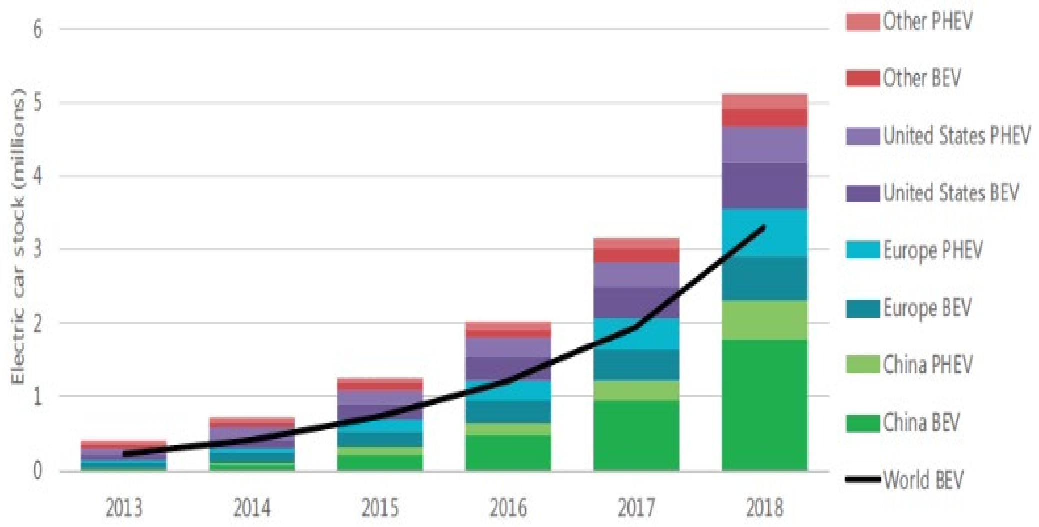



**Figure 71 Passenger electric vehicle car stock over the world from 2013 to 2018 according to IEA global EV outlook. A year-to-year growth rate of about 60% is observed since 2016. Pallas et al. (2020).**

In their literature review, Pallas et al. (2020) note that in electrically driven vehicles, several factors involve a different driving style with EVs compared to conventional ICEVs. These are the limited vehicle range, the availability of regenerative braking and different sensations (acceleration, torque, acoustical perception) arising when driving EVs. After becoming experienced, EV drivers show anticipation, use deceleration to efficiently benefit from the regenerative braking and try to drive economically by favouring a constant speed as far as possible. EV drivers have a perception of the vehicle, either from technical performance (acceleration ability and torque availability) or from acoustical feedback, which differs from conventional vehicles. This may affect their driving behaviour in different ways, often by driving more smoothly with effects on speed, acceleration/deceleration rates and lengths, but also sometimes by more aggressive driving schemes noticed with fleet users or users having powerful EVs. The ongoing traffic conditions and vehicle range certainly play a central role in the adoption of one or the other attitude. Smooth driving is favourable to propulsion noise and rolling noise reduction, while aggressive driving leads to increase of noise during accelerating and decelerating driving conditions, counteracting the potential impact of EVs on road traffic noise reduction. They consider the main EV technological characteristics that are likely to modify the driving behaviour and provide results available from scientific studies on EV driving styles that are relevant for noise emission.

Pallas et al. (2020) also note that noise source emission of EVs have been studied over several decades. They summarize the available findings as: EV powertrain noise (mainly motor noise), rolling noise and AVAS as the main specific noise contributions on EVs. Even if the propulsion noise is lower than on conventional vehicles, the motor noise has a spectrum rich in tones that make it audible, even unpleasant, at least at low speed. Existing studies have found that for an electric vehicle, tyre/road noise exceeds propulsion noise at a speed of 30 km/h and above, emphasizing the need for low noise tyres (they provide considerable experimental evidence regarding tyre noise for EVs) and quiet road surfaces for noise reduction (Figure 72).

Of available road traffic noise prediction models, the majority only refer to conventional vehicles and do not mention electric vehicles. Some of them have anticipated a specific category, but do not yet take EVs further into account in the noise prediction, as is the case with the European method CNOSSOS-EU (see Table 27 in this report). Studies have been conducted, either to define a methodology for including EVs in the models or for providing exploratory EV noise emission data. Considering the current state of EV market and the limited share of EVs in the overall fleet, these data rely on a low number of vehicles. Methodological difficulties for characterising noise emission of EVs encounter several difficulties: proximity of EV levels to background noise in some frequency bands and at low speed; inability to drive an EV in neutral for coast-by tests and hence difficulty in measurement of rolling noise.

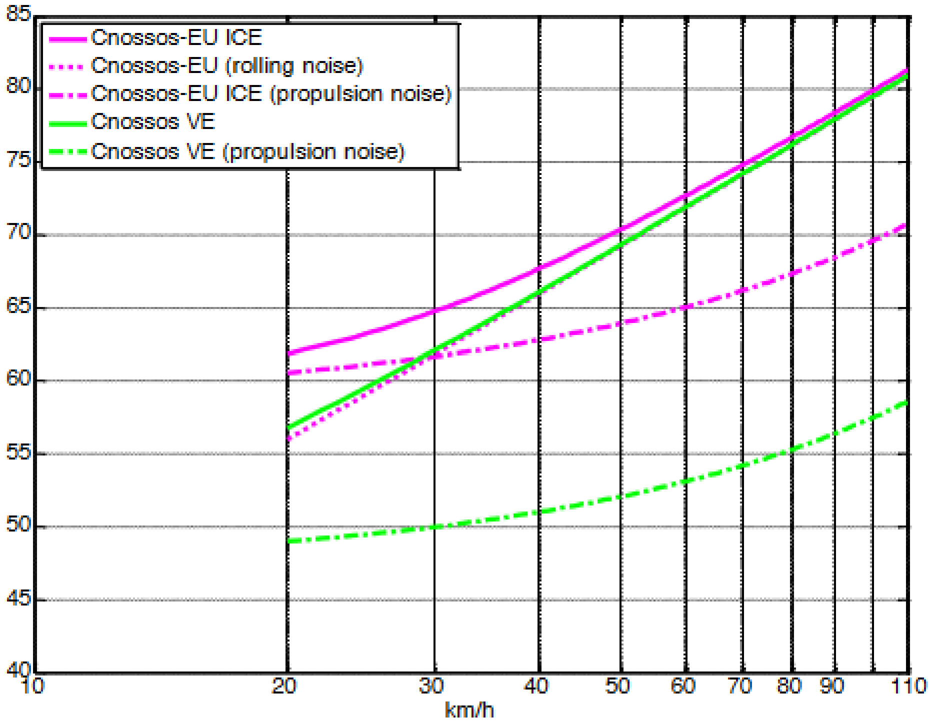



**Figure 72  Comparison of CNOSSOS-EU (for Internal Combustion Engine (ICE) light vehicles) and the model CNOSSOS-EV (from Pallas et al., 2020).**

## MITIGATION OF HEALTH RISKS BY TRANSITION TO ELECTRIC VEHICLES: A PRELIMINARY ESTIMATION

A recent Japanese study (Selamat  et al., 2020) provided a broader estimate of the noise exposure and health benefits that might be expected in an urban area as a result of transition from ICEVs (Internal Combustion Engine Vehicles) vehicles to EVs. The abstract of the paper summarizes both the approach and the result:

> *'Firstly, we calculated sound level in two areas with two different traffic flow situations: current traffic flows with ICEVs and the prospective ones with EVs. Next, we estimated the population affected with ischaemic heart disease and high sleep disturbance according to the exposure response relationships established by the World Health Organization Regional Office for Europe and the national health statistics and surveys in Japan to elucidate the contribution of the transition to EVs. While the estimated reduction in sound level was less than 4 dB (even if all vehicles were changed to the EV), the affected population were reduced by approximately several tens of percent, hence, the total health risks due to road traffic noise would significantly reduce by the shift to EVs.'*

# ROAD TRAFFIC NOISE MANAGEMENT PRACTICE

## NOISE ACTION PLANS (EUROPE)

The current situation and approach to road traffic noise in Europe has been introduced in the early section of this scoping document: '*Overview: Europe*', where Action Plans are a primary requirement of each member country for noise management. This section considers various recent literature regarding approaches to prioritization within these Action Plans. These considerations may have some relevance for Australian authorities, but it is suspected that this may be limited. Material from the sections on inequitable distribution of noise exposures and on engagement of stakeholders may hold the most interest.

## PRIORITIZATION WITHIN NOISE MANAGEMENT ACTION PLANS

How do Action Plans, and approaches to noise management, determine where action needs to be taken? The paper, '*A review and comparative analysis of European priority indices for noise action plans*' by D'Alessandro and Schiavoni (2015) notes that the most common approach to deal with noise impact at a policy, economic and strategy level is the use of priority indices focused to highlight areas or buildings where mitigation actions are more advisable or urgent. The aim of this research was to provide a review of the most used European priority indices and to test some of them in a study area. The comparative analysis demonstrated that the method chosen for the prioritisation deeply affects the ranking of the areas where noise measures need to be realized. Some methods tend to give high priority to noise sensitive locations, others to high populated buildings, and others to the areas where noise levels are high

**Table 29 Selection of Action Plans considered in the study of D'Alessandro and Schiavoni (2015).**

| Index | Reference | Short name |
|---|---|---|
| Dublin prioritisation matrix | Dublin Local Authorities (2013) | DubMat |
| Kilkenny prioritisation matrix | Kilkenny City Council (2013) | KilMat |
| Multi-annoyance Building Prioritisation Score | Licitra et al. (2011) | MABPS |
| Number of people highly annoyed | European Environmental Agency (2010) | NHA |
| Number of people annoyed | European Environmental Agency (2010) | NA |
| Qcity noise scoring proposal | Petz et al. (2007) | QCNS |
| Gden indicator | Jabben et al. (2010) | Gden |
| Italian Priority Index | Italian Government (2000) | IP |

Riedel et al. (2021) raise the issue of inequitable distribution of road traffic noise exposures and noise health effects in the community, and the need to take these inequities into account in plans for its management. The authors note that European noise policy is originally committed to the precautionary principle of environmental planning, aiming at high environmental health standards. However, recent assessments of the European Noise Directive report an increase of traffic and an under-estimation of health effects. In addition to traffic-related noise exposure, the European Union faces another serious public health problem: health inequity, with social epidemiological research providing tentative evidence on social and socio-spatial inequalities in noise exposures and health effects. In view of these two challenges, the authors set out to describe five propositions offering entry points to achieve more environmental health equity through noise action planning:

- Implementing noise action planning effectively requires noise and health in all (planning) policies. Binding standards for noise-related environmental quality and inter-sectoral collaboration across political and administrative levels help establish this requirement.

- Noise action planning should consider differences in health effects (different vulnerabilities)
- Distributional effects of noise action plans haveto be evaluated.
- The assessment of the total noise exposure is necessary to estimate the extent of inequalities in environmental exposures.
- Public information and consultation according to the END involves empowerment and innovative methods to enable effective and just civic engagement

These propositions are shown in Table 31, where the roles, responsibilities and opportunities of different government and non-government stakeholders are identified. While the authorities and responsibilities in the Table are written specific to the European/German situation, there are parallels in the Australian context, and both the issues, and approaches have relevance here. The Table may also prove a useful communication aidl for interaction with local governments and community groups.

With much of the literature focussed on noise action planning in Europe, King et al. (2011) provide a comparison/contrast (though now a decade old) of the assessment and control of road traffic noise in a European state under the END (Ireland) compared with in NSW. Both Irish and NSW jurisdictions have committed considerable resources to develop road traffic noise management measures, but the authors note that these approaches are quite different both in terms of their information requirements and required responses. These are summarized in Table 30 below.

**Table 30 Comparison of noise management tools used by Authorities in Ireland and in Australia (King et al., 2011).**

| Noise Management Tool | Ireland | NSW |
|---|---|---|
| Noise Policy | ✓ | ✓ |
| Guidelines | ✓(1 document) | ✓(2 documents) |
| Non Mandatory Criteria | ✓ | ✓ |
| Level of Mitigation based on being Rea-sonable and Feasible | ✓ | ✓ |
| Additional Consideration of Low Noise Environments | ✓ | ✓ |
| Noise Maps | ✓ | × |
| Action Plans | ✓ | × |
| Preventative Strategy | × | ✓ |
| Noise Abatement Program | × | ✓ |
| Funding Model | × | ✓ |
| Database of Impacted Residences and Measures Implemented | × | ✓ |

## A USEFUL ESSAY ON NOISE MANAGEMENT APPROACHES

In their 2022 book '*Environmental Noise Pollution*', Murphy and King provide a summary of *'Noise Mitigation Approaches' (Murphy & King, 2022c).* This covers aircraft and rail noise as well, but much is focussed on road traffic noise. The value of this chapter is not so much that it is cutting edge in terms of innovative approaches, but that it is:

- comprehensive in terms of planning for noise abatement
- identification of stakeholders
- details of source-based mitigation and
- propagation-based measures, and
- a discussion on cost efficiency in noise abatement.

It is noted here as it could be a potentially useful document to provide both structure and material for a locally produced document – should that be required for communication with other authorities and the public.

**Table 31 The five propositions for stakeholders for action on more equity in noise action planning (from Riedel et al., 2021).**

| Urban planning department | Environmental department | Health department | Local parliament with its committees | Researchers | Nongovernmental and civil society organizations | Citizens | EU |
|---|---|---|---|---|---|---|---|
| *Proposition 1) Implementing noise action planning effectively requires noise and health in all (planning) policies. Binding standards for noise-related environmental quality and inter-sectoral collaboration across political and administrative levels help establish this requirement.* | | | | | | | |
| Local: Contribute to noise action planning | - Integrate views of other departments<br>- Set own standards on noise<br>- Negotiate agreements with other departments | Contribute to local (environmental) planning | -Commit itself to noise and health in all policies<br>-Decide on local environmental standards<br>-Call for and support agreements with higher levels. | Provide evidence for standards | Expert associations: Offer trainings for practitioners to understand other stakeholders and develop argumentation lines | ---- | Set environmental standards |
| *Proposition 2) Noise action planning should consider differences in health effects (different vulnerabilities).* | | | | | | | |
| Inform environmental department about sensitive infrastructures | Integrate multiple burdens and vulnerability in analysis | Support with data on vulnerability and knowledge in this field | Provide political mandate | Provide tools for integrated multiple burden assessment and evidence on vulnerability among population groups | Expert associations: Raise public awareness through different media and trainings | ---- | Adapt END on integrating vulnerability |
| *Proposition 3) Distributional effects of noise action plans have to be evaluated.* | | | | | | | |
| ---- | Work together with neighbouring municipalities to follow 'all affected' principle<br>Use results from vulnerability assessment (see proposition 2) | ---- | Provide political mandate | Develop and help establish an indicator system for monitoring purposes | ---- | ---- | Adapt END on distributional effects |
| *Proposition 4) The assessment of the total noise exposure is necessary to estimate the extent of inequalities in environmental exposures.* | | | | | | | |
| Develop linkages to integrate noise in land use plan or urban development concepts systematically with the focus on multiple burdens | Try innovative approaches from science | Support with data on vulnerability and knowledge in this field | Provide political mandate | Develop modelling techniques and communicate their results | Expert associations: (like VDI in Germany)<br>develop guidelines on methods on multiple burdens and noise | ---- | Adapt END on methods on multiple burdens |
| *Proposition 5) Public information and consultation according to the END involves empowerment and innovative methods to enable effective and just civic engagement.* | | | | | | | |
| Provide experience in public participation to colleagues from environmental department | Run a participatory process taking vulnerability of people into account | Offer knowledge and methods on empowerment and setting related approaches | Provide political mandate | Develop innovative tools and methods for participation | - Make us of the participatory processes<br>- Encourage citizens and provide instrumental support (e.g. overcome illiteracy, language barriers, etc.) | - Make use of the participatory processes<br>- Give feedback on perceived effectiveness of measures | Develop Aarhus convention further |

*Notes*. This table was inspired by the WHO environmental health inequalities resource package (Fairburn *et al.* 2019) that provides information on what experts and stakeholders can do. We selected the categories of the environment department that is responsible for noise action planning in Germany according to the END. We included the urban planning as well as the health department. Other departments might be relevant, too. As in Germany the local level is politically strong due to their right to self-government, we included the local parliaments with its committees, mainly the one on the environment. Furthermore, we added researchers, non-governmental organisations and civic organisations, citizens and the EU as responsible institution for the legal framework. This table is much more to show relevant stakeholders named in our five propositions than being comprehensive in the sense of a policy analysis.

# LITERATURE ON INNOVATIVE TRAFFIC NOISE MITIGATION PRACTICE/STUDIES

## BARRIERS and VEGETATION

Given the extensive experience in NSW in the planning and design of noise barriers for roadways, and roadway alignments to reduce exposures, this review did not undertake a specific search for papers on this topic. However, several papers of potential interest with respect to innovative barrier design, choice of barrier material for sustainability, alternative to barriers, vegetation strips, and the effect of vegetation on perception of the noise beyond its direct reduction of noise levels, were identified during other searches and are reported below.

### Alternative Approaches to Barriers (USA)

Rochat et al. (2022) provides the most recent information on barriers in the USA, published as NCHRP Report 984: *Alternative Approaches to Avoiding and Reducing Highway Traffic Noise Impacts*. The Foreword of this document notes that: *State DOTs are required to consider highway traffic noise impacts from projects on existing and planned facilities. When these impacts exceed certain thresholds, the Code of Federal Regulations 23 CFR Part 772 specifies a limited number of approaches to mitigate highway traffic noise. The most commonly used approach is noise barriers—usually noise walls. To date, some 3,000 miles of noise walls have been constructed along U.S. highways, at an average cost of $2M per mile of wall*. Rochat et al. (2022) go on to note that 'noise walls' are not always feasible, or not always wanted by the community. The purpose of their document was to investigate the noise reduction potential of alternative strategies. These included on-road design choices such as quieter pavements, bridge decks, bridge joints and rumble strips; highway design choices such as adjustments to vertical or horizontal alignment, solid safety barriers in lieu of guardrail, and solar panels as barriers; right-of-way design choices such as earthen berms, vegetation, or sound-absorbing ground surfaces (including gravel); and traffic management strategies such as speed or truck restrictions. Sound-absorptive treatments were also examined, for walls, bridge undersides, and tunnels. The investigation was by literature search, with selected alternatives evaluated, often in combination, through virtual testing using the FHWA Traffic Noise Model (TNM v3.0). The output was a table of potential noise reductions, key considerations, relative cost estimates ($–$$$$$), and notes on context appropriateness. Highlights of the further investigations are included in the table for low-height berms, solid safety barriers, and acoustically soft ground. The project also produced a practitioner's resource, as Part II of the document, providing a procedural screening of alternative noise reduction based on roadway context, appropriateness, related costs, and estimated noise reduction.

It is unlikely that that there is much in this NCHRP document (Rochat et al., 2022) that would be regarded as novel in NSW, but its systematic approach, at least considering the possibilities of alternatives to noise barriers, may provide a useful *aide-memoire* for future considerations in NSW policy and practice. The authors identified six areas of noise mitigation for further research:

1. Effectiveness of low barriers to reduce noise generated by different types of highway vehicles;
2. Effectiveness of solar panels to reduce highway traffic noise
3. Effectiveness of gravel in right-of-way to reduce highway traffic noise
4. Effectiveness of vegetated screens to reduce highway traffic noise, and when to include effects in modelling
5. Effectiveness of in-ground treatments to reduce highway traffic noise
6. Effectiveness of absorptive treatment on a bridge under structure to reduce highway traffic noise.

## Short and long-term performance of barriers, and sustainability (Europe)

A quite different take on noise barriers is the potentially useful analysis by Ahac et al. (2021) of the long-term aspects and sustainability of road traffic noise wall design, presenting a meta-analysis of data collected during a systematic review of concrete, metal, and wood panels. Acoustic and non-acoustic performances were reviewed, together with long term performance and cradle-to-gate sustainability. The review – comprehensive, but rather long-winded – referred to the European Standards for noise barriers: *European Committee for Standardization (CEN). CEN/TC 226/WG 6—Noise Reducing Devices, Published Standards*. Their aim was to support the process of noise wall design and management by shifting the emphasis in decision making from construction costs to the long-term sustainability of the road traffic noise mitigation project. An example of their synthesis is shown in Table 32 below.

**Table 32 Overview of the minimal reported values of the specific noise wall performance measures for each analysed panel type. Numbers in square brackets are references to original sources (Ahac et al., 2021).**

| Performance Group | Noise Walls Panel Performance Measure | Concrete | Metal | | Wood | |
|---|---|---|---|---|---|---|
| | | | Aluminum | Steel | Timber | Willow |
| Acoustic performance | Single-number rating of sound absorption (dB) | 4 [73] | 8 [53] | | 8 [73] | |
| | Single-number rating of sound insulation (dB) | 24 [73] | 15 [53] | | 24 [73] | |
| | Insertion loss (dB(A)) | 18 [51] | 19 [51] | | 13 [51] | |
| Non-acoustic performance | Surface density (kg/m$^2$) | 150–250 [90] | 5–20 [90] | 5–10 [90] | <20 [90] | <5 [44] |
| | Mechanical resistance (kg/m$^2$) | >500 [53] | <400 [53] | | >20 [90] | |
| | Average construction costs (€/m$^2$) | 225 [44] | 216 [44] | | 206 [44] | |
| Long-term performance | Minimal service life (years) | 40 [44] | 30 [44] | 20–25 [44] | 20–40 [44] | 25 [44] |
| | Maintenance and replacement costs (€/m$^2$) | 40 [51] | 120 [51] | | 115 [51] | |
| | Lifecycle cost (€/m$^2$) | 305 [51] | 515 [51] | | 360 [51] | |
| Cradle-to-Gate Sustainability and Recyclability | Carbon footprint (t) | 70 [44] | 0.7 [44] | 60 [44] | 4 [44] | −17 [44] |
| | Water footprint ($10^6$ l) | 0.8 [44] | 270 [44] | 270 [44] | 0.5 [44] | 850 [44] |
| | Primary energy use ($10^9$ J) | 3.7 [44] | 1.2 [44] | 0.8 [44] | 0.1 [44] | 1.9 [44] |
| | Transportation embodied energy ($10^9$ J) | 0.4 [44] | 0.05 [44] | 0.05 [44] | 0.01 [44] | 0.1 [44] |
| | Recycling potential at end-of-life (%) | 80 [44] | 100 [44] | 100 [44] | 20 [44] | 40 [44] |

The multi-criteria analysis in this paper suggested that, when choosing a panel, preference should be given to those using lightweight concrete materials. A further comprehensive cradle-to-grave assessment of lightweight concrete panels with expanded clay and recycled tire rubber aggregates, which was performed to fill a knowledge gap observed in the literature and identify opportunities for the improvement of lightweight concrete sustainability, showed that the main environmental impacts of these panels are due to their production processes and that the way to reduce such impacts is to use panels made with aggregates from secondary raw materials.

## Sonic crystals acoustic barriers

This topic may prove to be one of novelty interest only. 'Sonic crystals' as noise barriers for road traffic are described in recent literature and it was believed appropriate that it at least be mentioned in this review. An example is the Fredianelli et al. (2019) paper, *Recent Developments in Sonic Crystals as Barriers for Road Traffic Noise Mitigation*. Sonic crystal noise barriers are non-homogeneous structures created by the arrangement of acoustic scatterers in a periodic configuration. The name apparently derives from the physics analogy of the scattering of X-rays as they pass through a solid crystal, and the interference patterns that develop. The size of the scatterers and their spacing must be of the order of the incident wavelength so that the periodic structure can interact with the incident wave. A computer impression of an example of a sonic crystal-type barrier is shown in Figure 73.

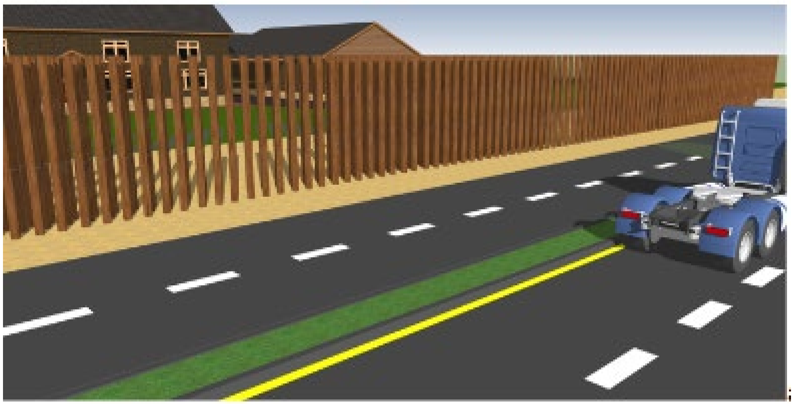


**Figure 73 Example of a sonic crystal-type barrier (Amado-Mendes et al., 2016).**

The potential advantages of such a barrier type include air flow and visual penetration, with suggestions also that the interference design may reduce diffraction over the barrier and increase overall Insertion Loss achieved – though evidence for this for road traffic noise mitigation was not obvious in the small number of papers examined on this topic. There also appears to be possibilities of frequency 'tuning' of the attenuation of the barrier, which may be of particular value in combination with certain quiet pavement surface treatments. However, there is clear acknowledgement by authors that development still has a way to go, and that such barriers may have issues with land requirements, maintenance, and accumulation of rubbish.

## Effect of the presence of vegetation on the perception of environmental noise

There is a quite widespread belief that there is an effect of vegetation screening of road traffic noise that is beyond that due to any attenuation achieved either by scattering or by ground effects associated with the vegetation. However, to date there has been little investigation or rigorous investigation of this postulated phenomenon.

Through a literature review, Van Renterghem (2019) attempts to bring together key studies, alternative observations, theories and explanatory mechanisms of how visible vegetation might mitigate negative environmental noise perception. While the material is from a diverse literature, the majority of it is related to annoyance (though annoyance was somewhat loosely associated with other dimensions such as loudness, pleasantness, relaxing potential and tranquillity) and to road traffic noise as source. He enumerates three mechanisms suggested in the literature:

(1) source screening or invisibility (two complex competing human processing mechanisms are postulated: *audio-visual congruency* and *attention focussing)*
(2) the presence of visible green in the outside view, through the window, or even just awareness of the proximity of green (with two explaining frameworks from environmental psychology: *attention restoration theory* and *stress recovery theory* – both related to the benefits of exposure to nature) and
(3) vegetation as a source of natural sounds - vegetation as habitat for organisms producing sounds, or by making sounds itself, and also the presence of water sounds. Suggested mechanisms include masking (though this is considered unlikely) or that the presence of natural sounds support the restorative action described in (2).

Van Renterghem (2019) concludes that the available but limited literature shows there is a real effect of the presence of vegetation on annoyance – and he makes a rough quantitative estimate that the equivalent level reduction of green that is visible from home, with relation to annoyance, could reach 10 dBA. Clearly, the green setting should be of sufficient quality and quantity for this. The equivalent level reduction comes on top of the physical sound pressure level reduction one might obtain behind vegetation belts – that is, the positive perception effect through visibility could easily outperform the physical sound pressure level reductions of common vegetation belts.

Of the three potential mechanisms, restorative potential of vegetation (2 above) looks to be the dominant mechanism. Visible natural features of good quality can lead to sustained attention restoration and stress relief, counteracting negative outcomes of endured environmental noise exposure. Research on the noise annoyance experienced at home over a long period shows very strong and consistently positive effects when outdoor nature is seen through the dwelling window, consistent with the restoration hypothesis. It can reasonably be expected that, in the higher sound pressure level range, vision on green has a stronger effect than at low road traffic noise levels, but significant positive effects are to be expected over a wide range of environmental sound pressure levels. (As well as an effect in the home environment, there is additional support on the importance of natural features in the window view at the workplace (higher employee productivity and well-being), in hospital environments (better recovery) and at schools (better grades)). A person's knowledge of neighbourhood or nearby green, not directly visible from home, also shows a positive effect but has a smaller impact on noise perception.

This effect of green visibility on annoyance scores is an example of the category (*Change in other physical dimensions of dwellings/neighbourhood)*, listed in European Environment Agency (2020): *Categorization of noise management and mitigation measures* (see Figure 38 above).

## SPEED REDUCTION

See the study described in the section: '*An Urban Road Traffic Speed Reduction Intervention: Zurich'* for details of the measured health effect of urban road speed reduction (Brink et al., 2022).

Another study has also been reported from Switzerland, using data from Lausanne (Rossi et al., 2020). They estimated the health benefits associated with a speed limit reduction to thirty kilometres per hour in a health impact assessment of noise and road traffic crashes. The abstract of the paper is below – with an interesting finding that the modelling showed that the speed reduction benefits in reducing noise effects was greater than in reducing road accidents.:

"*Reductions of speed limits for road traffic are effective in reducing casualties and are also increasingly promoted as an effective way to reduce noise exposure. The aim of this study was to estimate the health benefits of the implementation of 30 km/h speed limits in the city of Lausanne (136,077 inhabitants) under different scenarios addressing exposure to noise and road crashes. The study followed a standard methodology for quantitative health impact assessments to derive the number of attributable cases in relation to relevant outcomes. We compared a reference scenario (without any 30 km/h speed limits) to the current situation with partial speed limits and additional scenarios with further implementation of 30 km/h speed limits, including a whole city scenario. Compared to the reference scenario, noise reduction due to the current speed limit situation was estimated to annually prevent 1 cardiovascular death, 72 hospital admissions from cardiovascular disease, 17 incident diabetes cases, 1,127 individuals being highly annoyed and 918 individuals reporting sleep disturbances from noise. Health benefits from a reduction in road traffic crashes were less pronounced (1 severe injury and 4 minor injuries). The whole city speed reduction scenario more than doubled the annual benefits and was the only scenario that contributed to a reduction in mortality from road traffic crashes (one death per two years). Implementing 30 km/h speed limits in a city yields health benefits due to reduction in road traffic crashes and noise exposure. We found that the benefit from noise reduction was more relevant than safety benefits*."

## TYRE-PAVEMENT INTERACTION

While it is well before the time-window of this scoping review, a good starting point for Australian work on tyre-pavement noise is the work by Parnell and Samuels (2006) who summarized type-pavement noise generated by 20 pavements surfaces representing a wide range of construction materials, techniques and surface textures, in NSW. These were compared with the results of similar studies of 75 pavements constructed overseas (Figure 74). A range of more than 14 dB in pass by noise levels was reported over the pavement types investigated with the NSW data being consistent, at least at that date, with international data for similar pavement types. In general, several low-noise asphalts were found to return the lowest overall noise levels followed by concrete pavements that had minimal surface texture. Dense grade and stone mastic asphalts, and exposed aggregate concrete were found to perform better than randomly transverse tyned concrete pavements. Longitudinally tyned concrete pavements are not a design that is used in NSW however in international studies they were found to return noise levels similar to that of dense graded asphalts. Uniformly tyned pavements and those which generate acoustic energy in discrete frequency bands were found to be amongst the loudest pavements.

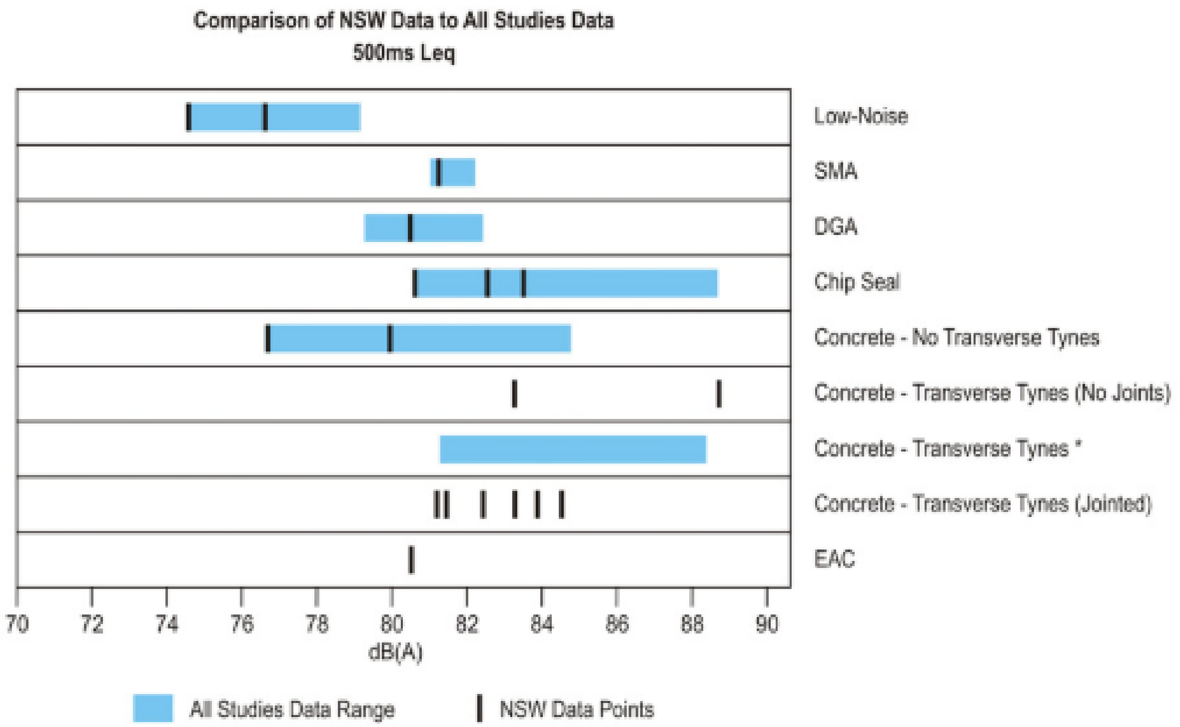



**Figure 74 Summary results from Parnell and Samuels (2006).**

VicRoads initiated an acoustic test program for low-noise road surfaces in 2013. The aim was to assess the acoustic performance and maintenance requirements for different types of pavement surfaces (described in Table 33) over a period of at least five years. Seven different products were laid on a single section of roadway on the Mornington Peninsula Freeway in Victoria, Australia. Acoustic tests were carried out at regular intervals following three different methods: Statistical Pass-By (ISO 11819-1), Close Proximity Method (ISO/DIS 11819-2) and On-Board Sound Intensity method (AASHTO TP 76-10), and results after one year are reported (Buret et al., 2014) and again after four years (Buret et al., 2019). Detailed results, including spectra, are provided in the papers, but a summary is: '*The trends observed is the early years of the trials have been confirmed and the* ***shaved*** *OGA-treated by horizontal grinding of the road surface, is found to have consistently outperformed the other asphalts tested, with noise levels approximately 3 dB lower than the standard OGA. Spectral analysis shows that the treatment appears to be particularly effective around 1 kHz, which not only is a critical frequency range for the generation of tyre/road noise, but also is the range in which the acoustic performance of the standard OGA is observed to deteriorate the most. The two configurations of double OGA tested are found to perform similarly with essentially differences in performance around 2 kHz which result in limited variations in overall noise levels. Both surfaces are found to be approximately 1 dB quieter than the standard OGA. Additional results on this study (McIntosh et al., 2019) demonstrate that grinding open graded asphalt has the potential to reduce traffic noise significantly, and this benefit lasts at least five years. This is a significant finding and suggests that the grinding process adds three years to the acoustical lifetime of OGA.*

*The additional co-benefits of improved skid resistance and reduced rolling resistance suggest that asphalt grinding warrants further investigation. The cost of asphalt grinding on a large scale is not known but may be considerable'.*

**Table 33 VicRoads trial pavements (Buret et al., 2014).**

| Section | Type of asphalt | Depth, mm | Section length, m |
|---|---|---|---|
| 1 | Standard OGA Size 10mm | 30 | 200 |
| 2 | Size 10mm OGA, treated by "shaving" | 30 | 100 |
| 3 | Double layer OGA Size 10mm and 14mm | 30 + 40 (resp.) | 150 |
| 4 | OGA Size 10mm (laid in two layers) | 2 × 35 | 150 |
| 5 | Standard SMA Size 10mm | 35 | 150 |
| 6 | Boral Noise Reducing Trial Mix2 | 35 | 150 |
| 7 | Boral Durapave Asphalt | 35 | 150 |

A review paper (Tonin, 2016) was published in Acoustics Australia: '*Quiet Road Pavements: Design and Measurement—State of the Art*'. He provided details of both the design and measurement of quiet road pavements (Table 34) of modern pavement types which shows noise reduction relative to Dense Graded Asphaltic Concrete (DGAC). The paper summarizes: '*This paper has presented an overview of state-of-the-art pavement design solutions which are being trialled to ascertain effectiveness and long-term viability. Of particular note is the development of low noise concrete solutions which potentially offer a long-term wearing surface. An important aspect of the ongoing effort in pavement noise reduction is the need to have reliable methods to measure the noise emission from those surfaces, not only to compare one with the other but also to assess the ongoing performance of a pavement surface over time. Three methods are described—the SPB method, the CPX method and the OBSI method—which each in their own way have advantages and disadvantages, but each method has a formal standard underpinning it'.* Tonin (2016) notes that the use of the SPB method is universal throughout the world, whilst the CPX method is favoured in Europe, and the OBSI method is favoured in the USA.

**Table 34 List of typical pavement surface types, physical characteristics and noise reduction properties (Tonin, 2016). Table is adapted from: Quiet Pavement Technologies. World Road Association (PIARC) (2013) http://media.cygnus.com/files/base/FCP/whitepaper/2013/09/quieter-pavements_11179329.pdf**

| Type | Acronym | Thickness (mm) | Maximum aggregate size or NMAS (mm) [a] | Texture (mm) and/or air voids content (%) | Noise reduct DGAC [b,c] |
|---|---|---|---|---|---|
| Dense-graded ssphalt concrete | DGAC 0/16 | 30 | 8/11/16 | 0.8 mm 4 % | 0 |
| | DGAC 0/11 | | | | |
| | DGAC 0/8 | | | | |
| Open-graded asphalt concrete | OGAC 0/16 | 45 | 16 | 25 % | 3 |
| | OGAC 0/11 | 45 | 11 | 25 % | 4 |
| | OGAC 0/8 | 45 | 8 | 25 | 5 |
| | OGAC (twin layer) | 25 (top) + 45 (bottom) | 8 (Top) 16 (bottom) | 20 % (top) 25 % (bottom) | 4 ~ 6 |
| Stone mastic asphalt | SMA 0/16 | 30–50 | 16 | 4 % | −2 to −1 |
| | SMA 0/11 | 30–50 | 11 | 4 % | 0 |
| | SMA 0/8 | 30–50 | 8 | 4 % | 1 |
| Poroelastic road surface | PERS | 30 | 2 (rubber) 8 (aggregates) | 20–40 % | 5–15 |
| Rubberized asphalt concrete, open | RAC(O) | 30 | 12 (as OGFC) | 14–20 % | 6 |
| Thin layer of asphalt concrete (very thin or ultra thin) | VTAC | 5–8 | 5–8 | 5–15 % | 3-7 |
| | UTAC | | | | |
| Surface dressing or bituminous surface treatment | Epoxy-bound orBST | 3–20 | 3–20 | Unknown | −3 to +2 |
| Portland cement concrete-porous | PCC (porous) | 80 | 9.5 | 20–25 % | 4–8 |
| Portland cement concrete-hessian dragged, tyned & diamond ground | PCC | Varies | Varies | 4–25 % | −3 to +8 |

[a] NMAS—nominal maximum aggregate size
[b] The figures in this column are provided as indicative values or ranges. They are either provided by manufacturers or come from specific studies found in the literature
[c] A negative value means the pavement produces higher noise levels compared with DGAC
[d] This table is adapted from reference [7]

A useful international paper in this field is published in the journal '*Measurement*' Li (2018). This is a thorough examination of the state of the art of measurement of tyre-pavement interaction noise (TPIN) techniques available. Noting that, currently, TPIN dominates for passenger vehicles above 40 km/h, and 70 km/h for trucks, the authors enumerate the different measurement techniques to quantify TPIN, including: the equipment used; measurement setup and standard if available; measured parameters and data processing; sound field category and major noise source; and advantages/disadvantages. The reviewed techniques were divided into three categories based on the test environment: roadside, on-board, and laboratory (Table 35). The measurement techniques have evolved from simple overall sound pressure level tests to more accurate on-board sound intensity tests, and then to sound field holography in the laboratory. Each technique has advantages and disadvantages. The roadside measurement is very easy and is regulation-oriented; however, it is not accurate or informative, and no steady and precise noise spectrum can be obtained. The lab measurement is very flexible, interesting, and is research-oriented; however, it is typically prohibitive in terms of equipment and sometimes fails to represent the actual driving conditions on the road. The on-board measurement is intermediate between the above two.

**Table 35 Tyre-pavement interaction noise measurement techniques reviewed by Li (2018). A detailed description of each is provided in the paper.**

| Category | Measurement Techniques |
|---|---|
| Roadside Measurement | Statistical Pass-By (SPB) |
| | Controlled Pass-By (CPB) |
| | Coast-By (CB) |
| | Statistical Pass-By and Time-Averaged Wayside (SPB/TA) |
| | Statistical Isolated Pass-by (SIP) |
| | Continuous-Flow Traffic Time-Integrated Method (CTIM) |
| | Coast-By Sound Power Level (CBSWL) |
| On-board Measurement | Behind the Tire (BTT) |
| | Trailer Coast-By (TCB) |
| | On-Board Sound Pressure (OBSP) |
| | Close Proximity Sound Intensity (CPI) |
| | Sound Intensity Field (SIF) |
| | Tire Cavity Microphone (TCM) |
| | Close-Proximity (CPX) Trailer |
| | In Vehicle (IV) |
| | On-Board Sound Intensity (OBSI) |
| Lab Measurement | Tire Vibration Measurement (TVM) |
| | Laboratory Drum (DR) |
| | Near-field Acoustic Array Technology (NAAT) |
| | Novel Pass-By with Novel Close Proximity (NPB with NCPX) |
| | Far-field Acoustic Array Technology (FAAT) |
| | Traffic Noise Auralization (TNA) |

Ling et al. (2021) undertook '*a comprehensive review of tire-pavement noise: generation mechanism, measurement methods, and quiet asphalt pavements*'. This review was conducted in China, but it is well-written and published in *Cleaner Production,* with good coverage of the international literature. Its multiple aims are to provide reference and guidance to measure tire-pavement noise, design quiet asphalt pavement, and select suitable quiet pavements. The paper summarized some common and some novel quiet pavements. Also included is current thinking regarding the different noise reduction mechanisms of quiet asphalt pavement: *viz*. porous sound absorption, resonance sound absorption, connected pore excretion, control of the sound propagation path and damping and noise reduction (see Figure 75). An additional mechanism – *damping and noise* reduction – is also postulated for porous elastic road surfaces. The paper also provides a summary of existing quiet asphalt pavements (Table 36 below), and also of the reduction results measured on these different pavement types. Figure 76 reproduces (from Ling et al., 2021) what they call '*steps for selecting a best tire-pavement noise test method and a suitable quiet pavement'* – useful to help gain a broad understanding of the topics, but not particularly detailed for design purposes.

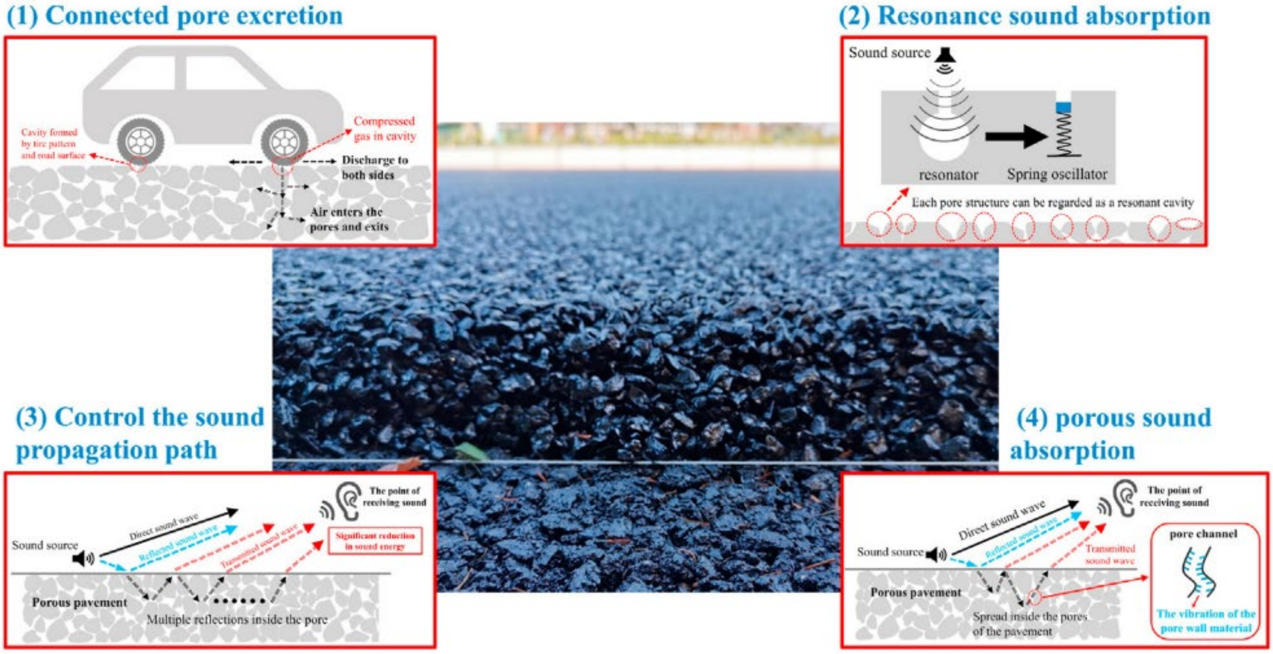



**Figure 75 Different sound reduction mechanisms of porous asphalt pavement (Ling et al., 2021).**

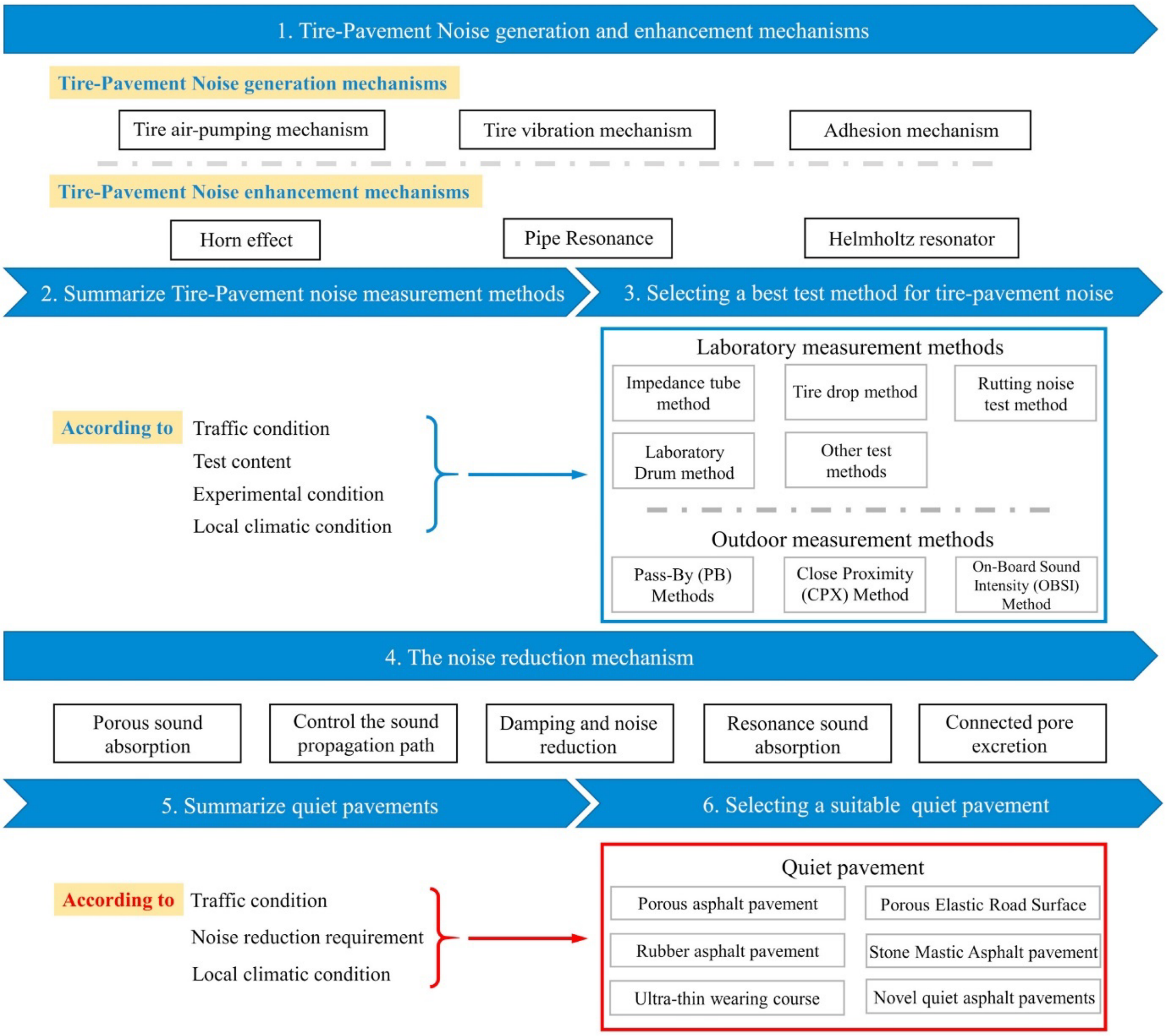
**Table 36 Summary of Existing quiet asphalt pavements (Ling et al., 2021).**

| No. | Category | | noise reduction mechanism | Advantage | Disadvantage | Scope of application |
|---|---|---|---|---|---|---|
| 1 | PAP | | ●●●● | Water drainage and noise reduction | Pores are easily blocked and have poor durability | Roads with low traffic flow, without intersections, turning lanes or narrow curves (Altreuther and Maennel., 2018; Hamzah et al., 2012; Ferguson, 2005). |
| 2 | RAP | | ● | High strength, great durability and great noise reduction performance | A higher temperature to blend | Highways with more heavy truck traffic (Guo et al., 2018) |
| 3 | UTWC | | ●● | Easy to lay and have excellent performance in skid resistance and noise reduction | Poor durability and low noise reduction efficiency | Roads with low traffic flow (Hu et al., 2019; Shen, 2015) |
| 4 | PERS | | ●●●●● | Water drainage and noise reduction | Low resistance and blocked pores | More suitable for urban roads in cold regions than PAP (Wang et al., 2017a) |
| 5 | SMAP | | ●●● | Smooth road | High cost | Highway, local and urban roads (Altreuther and Maennel., 2018). |
| 6 | Novel quiet asphalt pavements | A three-layer pavement structure | ● | great noise reduction performance | Low laying efficiency | – |
| | | Helmholtz type porous asphalt | ● | great noise reduction performance | Low laying efficiency | – |
| | | Curling Prefabricated Noise Reduction Pavement (CPNRP) | ●●●●● | High laying efficiency; great noise reduction performance | Poor durability | – |

●Porous sound absorption ; ●Resonance sound absorption ; ●Connected pore excretion ; ●Control the sound propagation path ; ●Damping and noise reduction

There appears to be ongoing interest in novel pavement types for quiet surfaces – including in China – but nothing appears to stand out. Some of it is published in 'materials-oriented' journals. There is also one paper (Praticò et al., 2021) that raises a potential issue of appropriate design of low-noise road mixtures for electric vehicles – ostensibly because of different frequency components from EVs compared to internal combustion engine vehicles (suggesting they are quieter at low frequencies, but could be noisier at high frequencies). The matter has simply been flagged as requiring a watching brief.

**Figure 76 Summary of TPIN mechanisms and design (Ling et al., 2021).**




Another recent, and extensive, literature review of low-noise pavement technologies and evaluation techniques comes from the Swiss EMPA laboratory (Mikhailenko et al., 2020). It reviews, for pavement researchers and professionals, the continuously evolving low-noise asphalt pavement technologies and the techniques which can be used to evaluate them. Test methods for determining the acoustical properties of asphalt pavements are reviewed. The Close-Proximity (CPX) method is the most used field test for pavement acoustics, followed by the Statistical Pass-By (SPB) and On-Board Sound Intensity (OBSI) methods. SPB seems is the most comprehensive methods, while the CPX is more practical. Methods for measuring the acoustical properties in the laboratory include the impedance tube for sound absorption and laboratory pavement noise simulators; with only the larger drum methods being able to produce conditions similar to in-situ. Methods for noise-relevant non-acoustical characteristics like surface texture, porosity and airflow resistivity were also reviewed. Optimizing surface texture at the macro-scale was found to be important in reducing tire/road noise. For pavement types, porous asphalt concrete (PAC) and its variants result in low-noise properties the most reliably, while having some drawbacks in durability and maintenance. Finally, various acoustical performance prediction models were discussed in both the laboratory and field environments. The principal findings (Mikhailenko et al., 2020) with regard to low-noise pavements are shown in Table 37.

**Table 37 Summary of Low-Noise Pavement Types (Mikhailenko et al., 2020).**

| Asphalt Pavement Type | Noise Mechanism Reduced | Advantages | Disadvantages | Reduction in dB(A) Rel. to Ref. at t=0 |
|---|---|---|---|---|
| Porous Asphalt Concrete (PAC) | Air pumping, Horn effect, Helmholtz resonance, Pipe effect | Very good levels of sound absorption and noise reduction<br>Good drainage properties | Durability issues with strength, moisture resistance and ravelling, lowers service life<br>Some clogging issues reducing noise improvement over time | 2–12, but normally around 3–3.5 |
| Double Layer Porous Asphalt Concrete (DPAC) | Air pumping, Horn effect, Helmholtz resonance, Pipe effect | Similar sound absorption and noise reduction to PAC<br>Less clogging issues than for PAC | More complicated placement<br>Same durability issues as PAC | Around 8 |
| Semi-Dense Asphalt (SDA) | Air pumping, Horn effect, Helmholtz resonance, Pipe effect | Improved durability properties compared to PA<br>Improved noise reduction compared to dense asphalt | Less noise reduction than PAC<br>More clogging issues compared to PAC<br>Durability issues with moisture resistance and ravelling | 5–7, but less than PAC in similar conditions |
| Crumb Rubber – Dry | Air pumping, Tread impact | Use of recycled material<br>Can improve some mechanical properties of asphalt such as resistance to fatigue | Generally has not been shown to provide noise reduction | 0–2 dB, mostly no effect |
| Crumb Rubber – Wet | Not clear | Use of recycled material<br>Improvement in rutting and fatigue resistance | Inconsistent results with noise reduction | Mostly no effect |
| Poroelastic (PERS) | Air pumping, Stick-slip, Tread impact, Stick-snap, Horn effect, Helmholtz resonance, Pipe effect | Superior noise reduction properties<br>Good skid resistance | Can have high susceptibility to rutting<br>Lack of adhesion to lower layer<br>Ravelling issues | 8–12 |
| Stone Mastic Asphalt (SMA) | Stick-snap | Good skid resistance<br>Good durability<br>Noise reduction with SMA LA | Same noise and absorption properties as dense asphalt for normal SMA | Mostly no effect, 2.5–4 dB for SMA LA |
| Thin Asphalt Layer (TAL) | Stick-snap | Good skid resistance | No significant noise improvement without higher air voids | Mostly no effect |
| Next Generation Concrete Surface (NGCS) | Air pumping, Horn effect, Helmholtz resonance, Pipe effect | Allows implementing PCC pavements with noise somewhat lower that conventional SMA. | Modest noise reduction compared to PAC<br>Low amount of research | 1.5–2 compared to SMA |

A 2006 Noise Intensity Testing in Europe (NITE Study) database provided Staino (2021) with the opportunity to explore the importance of aggregate size with respect to tyre-pavement noise. For this, the NITE pavements were grouped by type and evaluated via multiple linear regression analyses with respect to three variables: vehicle speed, v; maximum aggregate size, D; and minimum aggregate size, d. From Table 38, only the 24 pavements with reported aggregate sizes were considered. The evaluation found that tire-pavement noise variation for a specific pavement type is explained largely by aggregate size. Tire-pavement noise tended to increase with aggregate size—a behaviour consistently exhibited, for example, by SMA pavements. Porous asphalt pavements ranged from relatively quiet to relatively noisy depending upon aggregate size.

Ribeiro et al. (2021), at Euronoise 2021, reported on the long-term performance of low noise asphalt pavement (described as BBTM 0/6 and BBTM 0/4 type asphalts – see source document for further details). Monitoring of levels was not only at nearby residential facades but also very close to the vehicles (in the median strip?) and it is the long-term performance at the latter that is described. The work is of interest but is not well documented in the Euronoise paper. But Figures 77 and 78 provide some quantitative information on the temporal performance of a low-noise asphalt.

**Table 38 Pavement measured in the NITE study (Staino, 2021).**

| Pavement | Type | OBSI at 60 mph (dBA) | Description | Aggregate size* (mm) d | D |
|---|---|---|---|---|---|
| 49 | PA | 94.9 | Double layer porous asphalt 4/8 mm | 4.0 | 8.0 |
| 48 | PA | 95.1 | Porous asphalt 4/8 mm | 4.0 | 8.0 |
| CA/AZ 4 | Various | 96.6 | DGAC — fine aggregrate | N/A | N/A |
| CA/AZ 6 | Various | 96.9 | OGAC (nonporous) 75-mm thick on DGA | N/A | N/A |
| 50 | Various | 97.4 | Novachip 0/8 mm | 0.0 | 8.0 |
| 51 | Various | 98.0 | ISO 10844 | N/A | N/A |
| 10 | Various | 98.4 | Fine dense graded asphalt | N/A | N/A |
| 11 | PA | 98.5 | Porous asphalt 0/10 mm | 0.0 | 10.0 |
| 53 | SMA | 98.6 | SMA, 0/5 mm | 0.0 | 5.0 |
| 52 | SMA | 98.8 | SMA, 0/3 mm | 0.0 | 3.0 |
| 13 | Various | 99.0 | Thin layer asphalt 0/6 mm | 0.0 | 6.0 |
| 9 | CS | 99.7 | Fine surface dressing 0.8/1.5 mm | 0.8 | 1.5 |
| 7 | PCC | 99.8 | Porous cement concrete | N/A | N/A |
| 59 | Various | 99.8 | DSK 0/3 mm | 0.0 | 3.0 |
| 54 | SMA | 100.3 | SMA, 0/8 mm | 0.0 | 8.0 |
| 56 | SMA | 100.5 | SMA, 0/8 mm | 0.0 | 8.0 |
| 55 | SMA | 100.8 | SMA, 0/11 mm | 0.0 | 11.0 |
| CA/AZ 23 | Various | 101.1 | DGAC (type B) 30-mm thick on DGAC | N/A | N/A |
| 12 | Various | 101.3 | Dense graded asphalt 0/10 mm | 0.0 | 10.0 |
| 5 | PCC | 101.5 | Exposed aggregate cement concrete 0/7 mm | 0.0 | 7.0 |
| 58 | Various | 102.7 | DSK 0/5 mm | 0.0 | 5.0 |
| 2 | SMA | 103.1 | SMA, 0/14 mm | 0.0 | 14.0 |
| 60 | CS | 103.2 | Surface dressing OB 2/3 round | 2.0 | 3.0 |
| 3 | PCC | 103.4 | Porous cement concrete 0/7 mm | 0.0 | 7.0 |
| 4 | PA | 104.3 | Porous asphalt 0/14 mm | 0.0 | 14.0 |
| 63 | CS | 104.4 | Surface dressing OB 5/8 sharp | 5.0 | 8.0 |
| 57 | Various | 104.8 | Smooth surface (stone mastic 0/8 with epoxy coat) | 0.0 | 8.0 |
| 61 | CS | 104.8 | Surface dressing OB 3/5 round | 3.0 | 5.0 |
| 8 | CS | 105.7 | Surface dressing 8/10 mm | 8.0 | 10.0 |
| 62 | CS | 106.4 | Surface dressing OB 5/8 round | 5.0 | 8.0 |

*European aggregate sizes—per European Standard BS EN 13043, aggregate sizes are described in terms of a lower limiting sieve size, *d*, and an upper limiting sieve size, *D*, written as *d*/*D* (where the slash is just a separator, not *d* ÷ *D*).
†Chip seal (CS), also called surface dressing in Europe, contains the same ingredients as plant-produced, hot mix asphalt but differs in construction method: a layer of liquid asphalt is sprayed on a surface first, and then a layer of aggregate is applied.
DGAC, dense-graded asphaltic concrete; OGAC, open-graded asphaltic concrete; ISO 10844, test track specification, thin layer asphalt; DSK, cold-mix asphalt; PA, porous asphalt; CS, chip seal; PCC, Portland-cement concrete; SMA, stone mastic asphalt.

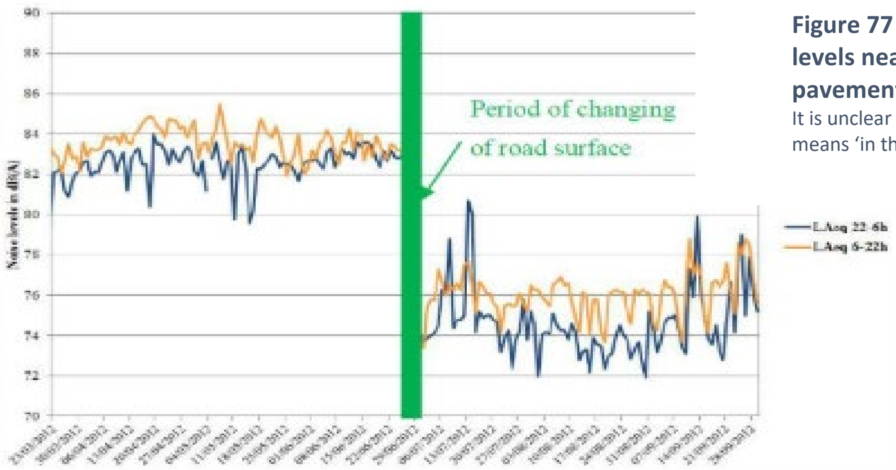



**Figure 77 Reduction of day and night noise levels near the source (median) after the pavement change (Ribeiro et al., 2021).** Note: It is unclear from the paper, but it is presumed the author means 'in the median strip'.

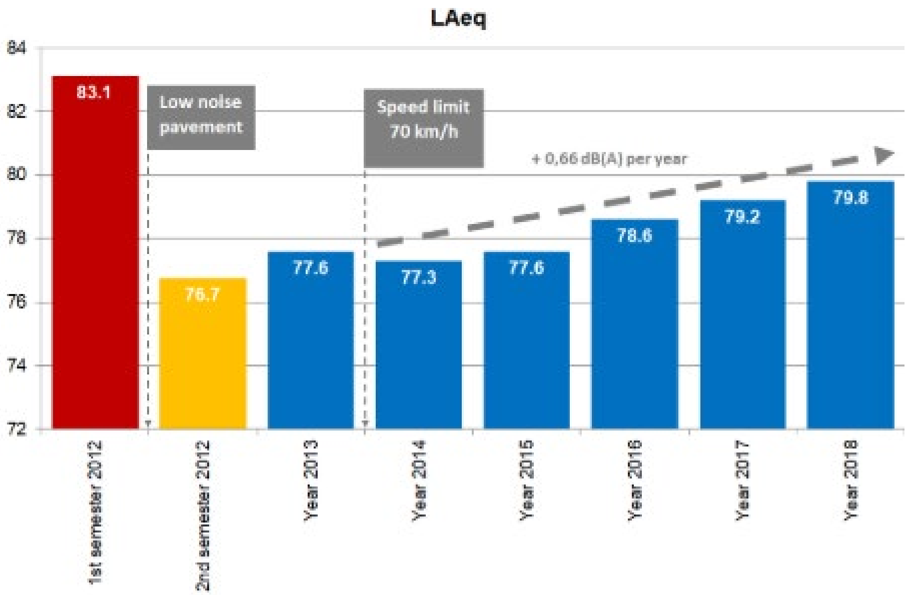



**Figure 78 Changes in levels over time (that is, 0.66dB(A) deterioration per annum, in the quieting performance of the low noise Asphalt - first measured in 2013 as in Figure 77) (Ribeiro et al., 2021).**

A recent paper by the Swiss Federal Laboratories for Material Science and Technology has undertaken a Life Cycle Assessment of low-noise pavements, considering not just the energy demand and additional climate change effects associated with low-noise pavements, but also the health benefits (as measured by Disability Adjusted Life Years – DALYs) associated with resulting reduced community exposures to road traffic noise. Piao et al. (2022) show that road traffic noise plays a significant role in the LCA of pavements. The trade-off between greenhouse gas and energy related impacts, on the one hand, and health effects, on the other hand, requires critical consideration by decision makers to take into account the health benefits when promoting low-noise pavements (Figure 79).

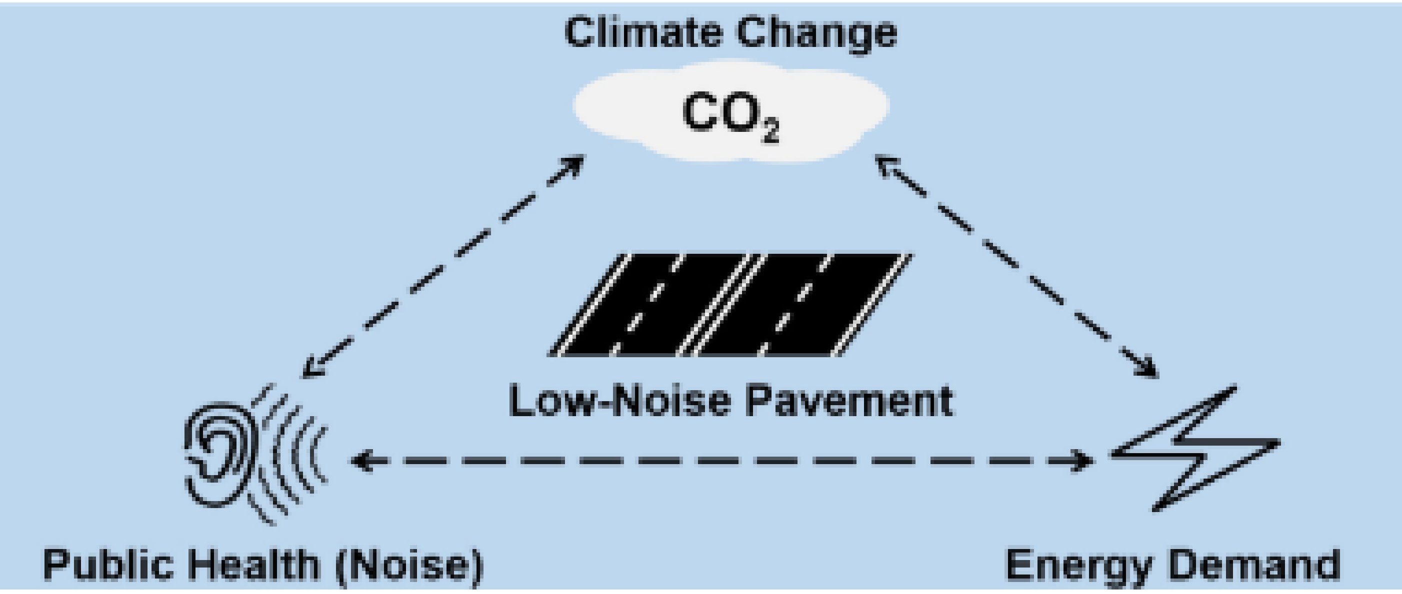



**Figure 79 Environmental trade-off with low-noise pavements: health benefits included with energy and climate change costs in a LCA (Piao et al., 2022).**

Given that noise exposure to road traffic noise is either measured, or predicted, external to dwellings, yet the effects of noise on people are presumed to be responding to are those immitted to their ears inside their dwellings, the difference between outdoor and indoor levels is of considerable importance. There are, however, not many studies that have simultaneously measured levels outdoor and indoors, also documenting details of the positioning of the windows during the measurements.

SiRENE, a major Swiss study (Locher et al., 2018), provided an opportunity to collect some good data on this topic - at least for European-constructed dwellings. Based on 102 Swiss residents exposed to road traffic noise. sound pressure level recordings were performed outdoors and indoors, in the living room and in the bedroom. Three scenarios—of open, tilted, and closed windows—were recorded for three minutes each. The median outdoor–indoor sound level differences were 10 dB(A) for open, 16 dB(A) for tilted, and 28 dB(A) for closed windows (see Figure 80). The authors (favourably) compare these attenuations (Table 39) with results from other studies (including one Australian study of open windows in ten Queensland houses (Ryan et al., 2011).

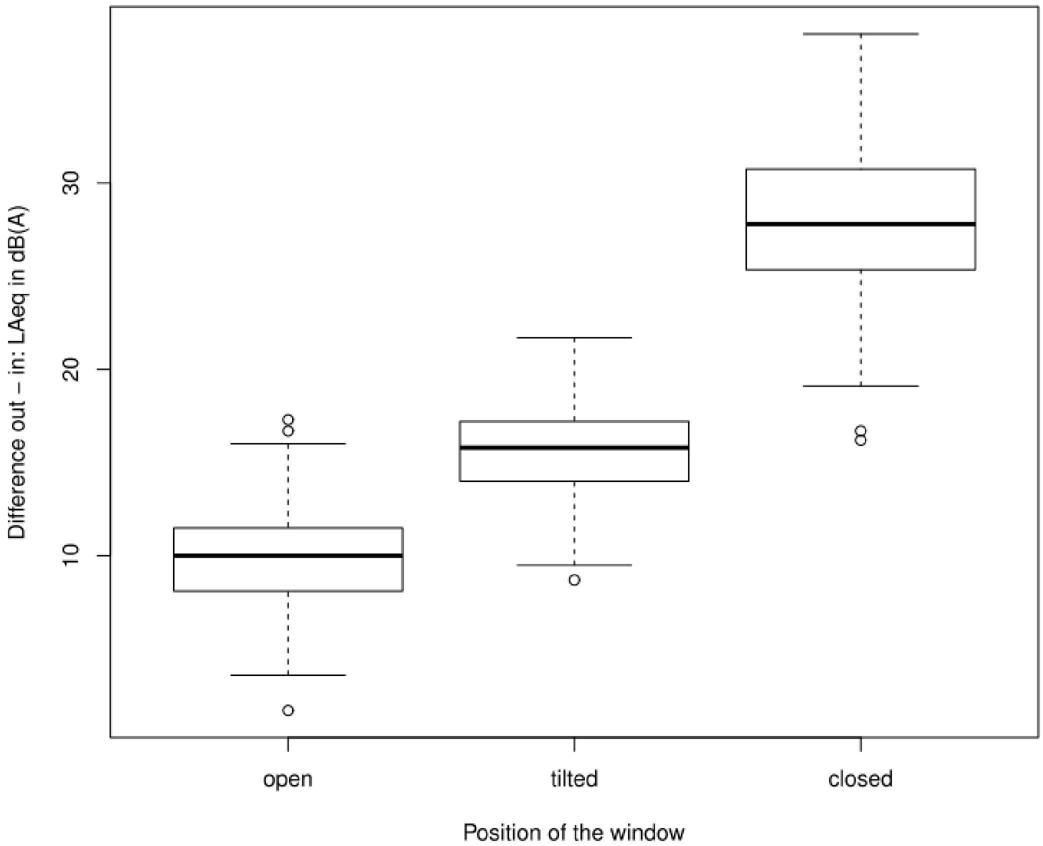



**Figure 80 Boxplots of all valid data showing the median (horizontal line in boxes), the 25% and 75% quantiles (lower and upper boundaries of boxes), the whiskers comprising the data within 1.5 times the interquartile range, and outliers outside the whiskers. Outside to inside differences – road traffic noise (Locher et al., 2018).**

For each situation, data on additional parameters such as the orientation towards the source, floor, and room, as well as sound insulation characteristics were collected. On that basis, linear regression models were established. For open and tilted windows, the most relevant parameters affecting the outdoor–indoor differences were the position of the window, the type and volume of the room, and the age of the building. For closed windows, the relevant parameters were the sound level outside, the material of the window frame, the existence of window gaskets, and the number of windows.

**Table 39 Differences in the sound levels outdoors and indoors: a comparison with other studies. The values in brackets give the number of analysed locations. (DLR: German Aerospace Center. See original paper for other references in table) (Locher et al., 2018).**

| Window Position | This Study | DLR 2010 [18] | | | DLR 2006 [16] | | Scamoni 2014 [29] | Ryan 2011 [15] | Maschke 2010 [21] | BUWAL 1998 [20] |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Freight Trains | Passenger Trains | Road | Road | Aircraft | Reference Road | Road | Aircraft | Aircraft |
| open | 10.0 (115) | 11.3 (4) | 11.9 (4) | 11.6 (4) | 13.4 (4) | 10.0 (4) | | 10.7 (11) | | |
| tilted | 15.8 (116) | 18.6 (10) | 18.0 (10) | 17.7 (10) | 13.7 (32) | 15.3 (32) | | | 12 | 15 |
| closed | 27.8 (76) | 30.1 (13) | 29.7 (13) | 30.1 (13) | 27.0 (15) | 25.6 (15) | 31.2 (334) | | | 25 |

Locher et al. (2018) also report the effect of transmission of road traffic noise through open, tilted, and closed windows on road traffic noise frequencies heard indoors (Figure 81). Of the measured $L_{eq}$ indoors in one-third octave bands from 50 to 10,000 Hz for open, tilted, and closed windows, both the open and tilted situations show maxima in the frequency spectrum around 1 kHz, with a slight decrease of levels towards lower frequencies and a more prominent decrease of levels towards higher frequencies. In contrast, for closed windows, the effect is a flatter spectrum.

The study presented physical results only, and there was no assessment of human response to the different internal spectrum. It is interesting to observe that, with closed windows, the spectrum is flat – and not necessarily dominated by the lower frequencies.

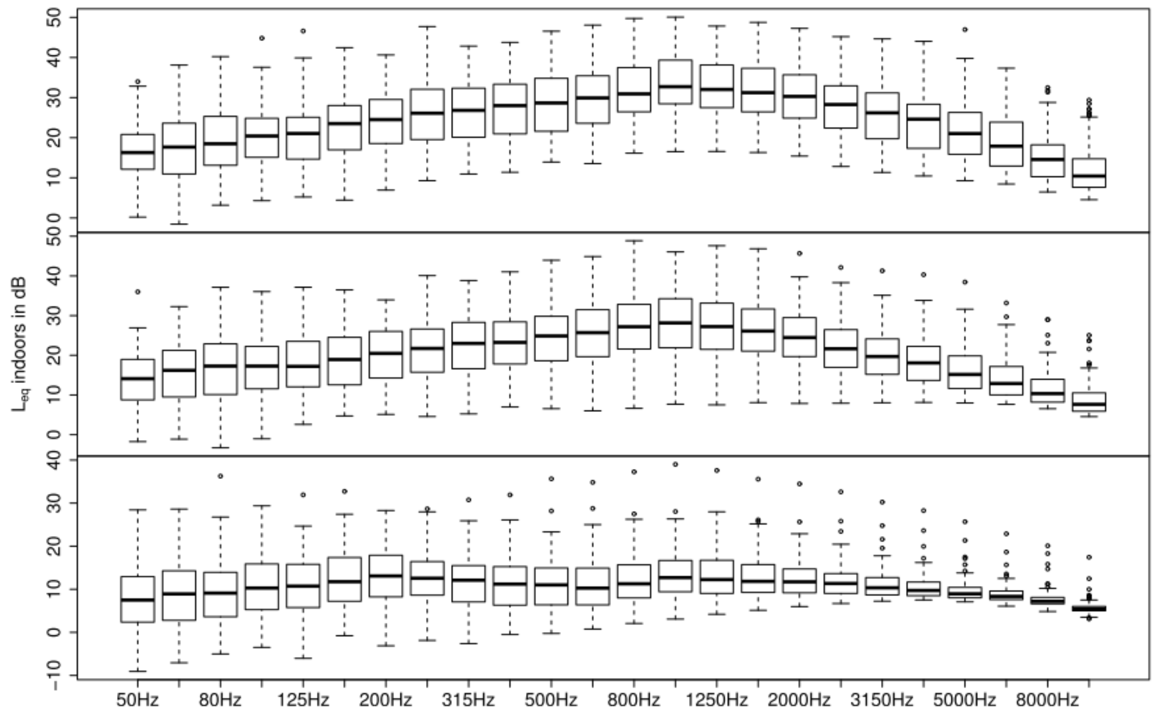



**Figure 81 $L_{eq}$ indoors in one-third octave bands for open (top), tilted (middle), and closed windows (bottom) of all valid measurements. Boxplots show the median (horizontal line in boxes), 25% and 75% quantiles (lower and upper boundaries of boxes), whiskers comprising the data within 1.5 times the interquartile range, and outliers outside the whiskers (Locher et al., 2018).**

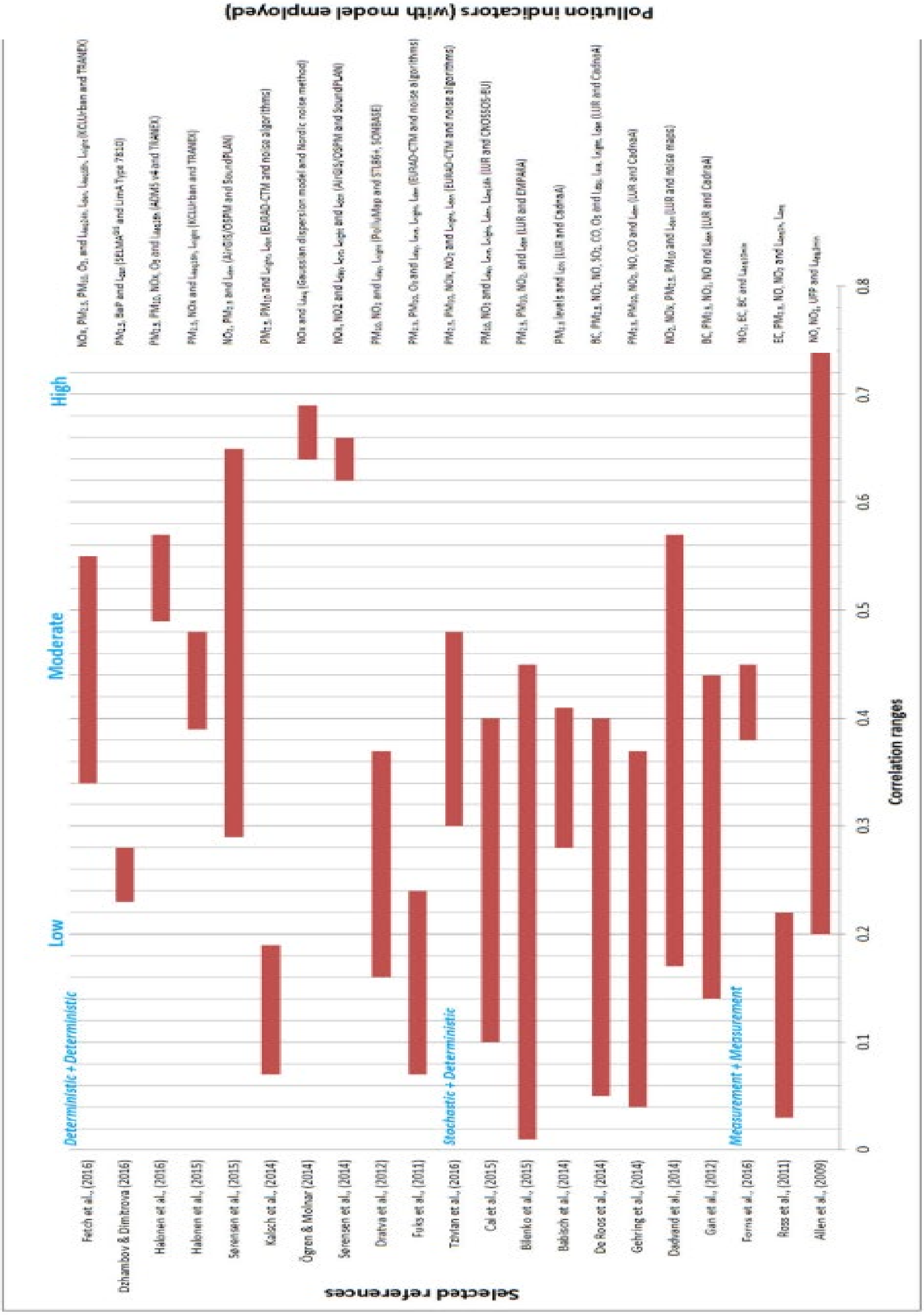
# AIR POLLUTION & NOISE: COMBINED ASSESSMENTS

Road traffic in urban areas gives rise to both air pollution and noise, thus exposure to traffic-related burdens is a multidisciplinary field. Traffic emits noise and air pollutants simultaneously. As a result, a strong correlation in the exposure is expected – but is not always found. In health impact assessments this correlation becomes a potential confounder, affecting the health impact analysis of both noise and air pollution. Khan et al. (2018) review literature about air and noise pollution from urban road traffic, including other relevant characteristics such as the employed dispersion models, Geographic Information System (GIS)-based tool, spatial scale of exposure assessment, study location, sample size, type of traffic data and building geometry information. Deterministic modelling is the most frequently used assessment technique for both air and noise pollution of short-term and long-term exposure. The authors observed a larger variety among air pollution models as compared to the applied noise models. Buildings act as screens for the dispersion of pollution, but the reduction effect is much larger for noise than for air pollution. Meteorology has a greater influence on air pollution levels as compared to noise. Khan et al. (2018) undertook a systematic literature review of studies that undertook combined road traffic noise and air pollution modelling and/or exposure assessment – reporting 57 relevant studies. They then tabulated characteristics of the studies in a large table – a portion of it is reproduced here as Table 40 (see the original paper for the full table, references, and explanations). Correlations between air and noise pollution vary significantly (0.05–0.74) and are affected by several parameters such as traffic attributes, building attributes and meteorology etc. Observed correlations between noise and air pollution are reported in Figure 82.

Table 40 Characterization of selected studies relating to air and noise pollution exposure, sorted by publication year, and grouped into general type of exposure assessment (Deterministic modelling, Stochastic modelling, Measurement). EXTRACT OF TABLE 1 (from Khan et al., 2018).

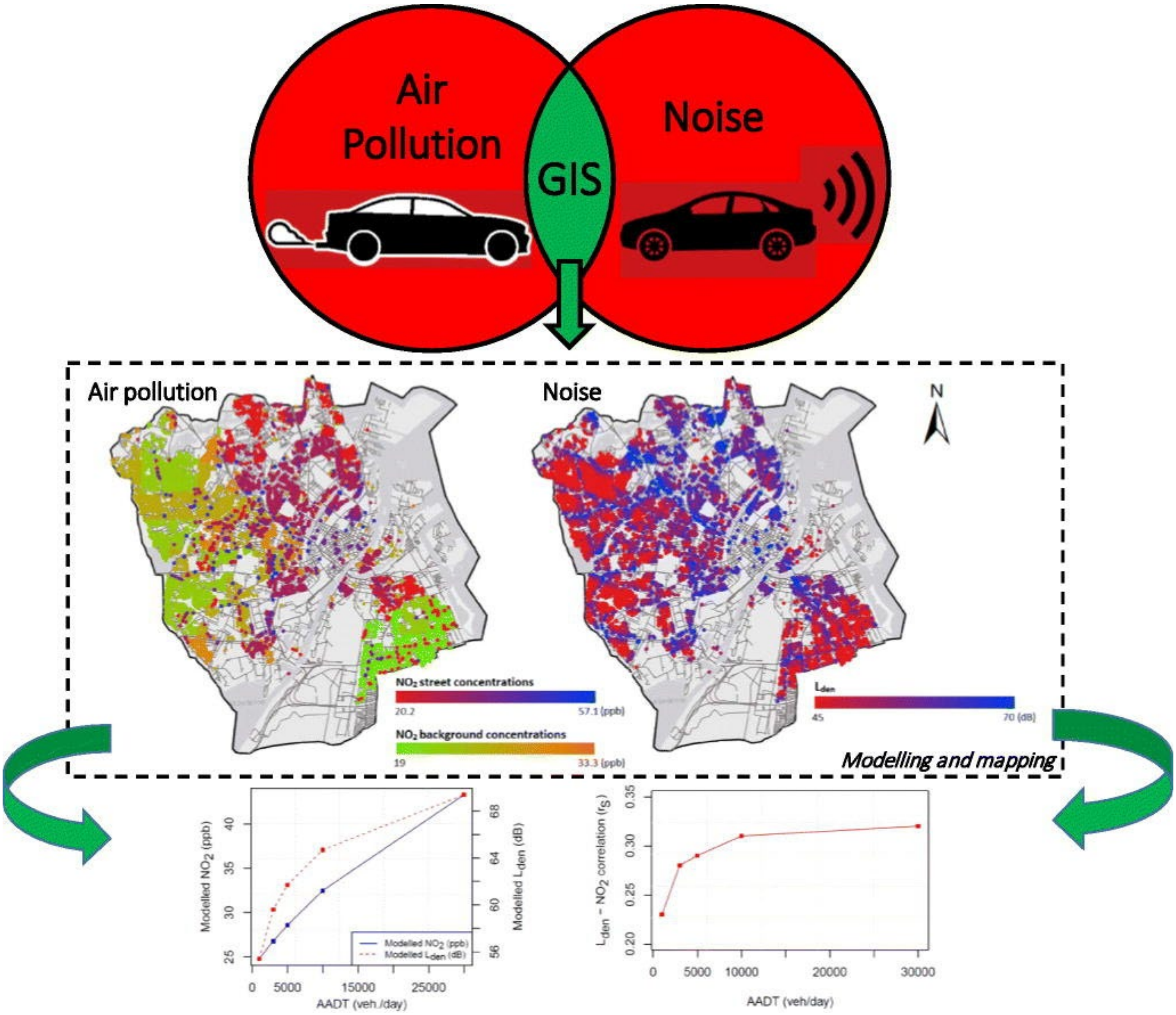
| Air pollution (Exposure indicator) | Air pollution (Dispersion model and exposure technique) | Traffic noise (Exposure indicator) | Traffic noise (Model and exposure technique) | Spatial scale | GIS tool[a] | Nature of traffic data | Study location | Sample size | Building geometry?[b] | Source |
|---|---|---|---|---|---|---|---|---|---|---|
| **Deterministic** | | **Deterministic** | | | | | | | | |
| $NO_2$ | AirGIS/OSPM | $L_{Aeq}$ | SoundPLAN | Urban | – | Avg[c] | Copenhagen/Aarhus (Denmark) | 57,053 | Yes | Roswall et al. (2017) |
| $NO_2$ | AERMOD | $L_{den}$ | SoundPLAN | Regional | ArcGIS | Avg | Skåne Region (Sweden) | 13,512 | Yes | Bodin et al. (2016) |
| $PM_{2.5}$, BaP | SELMA[GIS] | $L_{den}$ | LimA v.5 | Urban | ArcGIS | Avg | Plovdiv (Bulgaria) | 513 | Yes | Dzhambov and Dimitrova (2016) |
| NOx ($NO_2$/NO), $PM_{2.5}$, $PM_{10}$, $O_3$ | KCL-Urban, ADMS | $L_{Aeq24h}$, $L_{den}$, $L_{Aeq16h}$, $L_{night}$ | TRANEX | Regional | PostgreSQL, PostGIS | Hourly[d] | Greater London area (UK) | 9 million | Yes | Fecht et al. (2016) |
| $PM_{2.5}$, $PM_{10}$, NOx, $NO_2$, $O_3$ | KCL-Urban | $L_{Aeq16h}$ | TRANEX | Regional | PostgreSQL, PostGIS | Hourly | Greater London area (UK) | 5482 | Yes | Halonen et al. (2016) |
| $NO_2$ | ADMS-Urban | $L_{Aeq24h}$ | MITHRA-SIG | Urban | ArcGIS | Avg | Besancon (France) | 10,825 | Yes | Tenailleau et al. (2016) |
| $NO_2$, NOx, $PM_{2.5}$, $PM_{10}$, $O_3$, | KCL-Urban, Road Source Model | $L_{Aeq16h}$ | TRANEX | Regional | PostgreSQL, PostGIS | Hourly | Greater London area (UK) | 18,138 | Yes | Tonne et al. (2016) |
| $PM_{2.5}$, NOx | KCL-Urban | $L_{Aeq16h}$, $L_{night}$ | TRANEX | Regional | PostgreSQL, PostGIS | Hourly | Greater London area (UK) | 8.61 million | Yes | Halonen et al. (2015) |
| $NO_2$, $PM_{2.5}$ | AirGIS/OSPM | $L_{den}$ | SoundPLAN | Urban | ArcGIS | Avg | Copenhagen (Denmark) | 39,863 | Yes | Sørensen et al. (2015) |
| $PM_{2.5}$, $PM_{10}$ | EURAD-CTM | $L_{night}$, $L_{den}$ | END method | Urban | ArcView | Avg | Ruhr (Germany) | 4861 | Yes | Kälsch et al. (2014) |
| NOx | GDM | $L_{Aeq}$ | Nordic method | Urban | QGIS | Avg | Gothenburg (Sweden) | – | Yes | Örgen and Molnar (2014) |
| NOx, $NO_2$ | AirGIS/OSPM | $L_{day}$, $L_{eve}$, $L_{night}$, $L_{den}$ | SoundPLAN | Urban | ArcGIS | Avg | Copenhagen, Aarhus (Denmark) | 57,053 | Yes | Sørensen et al. (2014) |
| $PM_{10}$ | UFIPOLNET | $L_{day}$, $L_{eve}$, $L_{night}$, $L_{den}$ | IMMI | Urban | ArcGIS | Avg | Leipzig (Germany) | – | Yes | Weber et al. (2014) |

**Figure 82  Correlation ranges between air and noise pollution reported in the selected studies of this review, sorted by publication year and grouped into general type of exposure assessment i.e. Deterministic modelling, Stochastic modelling, Measurement. Pollution indicators and employed models methods are shown (from Khan et al., 2018).**



A related paper (Khan et al., 2020) extends these ideas and develops a tool to model exposures to air pollution and noise together - using harmonized inputs for the two pollutants based on similar geographical structure. They applied this to two different urban areas in Denmark. The paper also investigates, for the first time, the influence of traffic speed and AADT on the air-noise relationship (see Graphical Abstract from Khan et al. (2020) in Figure 83).

**Figure 83 Graphical Abstract: joint air pollution and noise assessments using harmonized inputs in a GIS (Khan et al., 2020).**




Two further studies are briefly mentioned. Both show that road traffic noise may be the more significant component in human exposures to traffic with both air pollution and noise.

Long-term residential road traffic noise and $NO_2$ exposure in relation to risk of incident myocardial infarction – A Danish cohort study: (Roswall et al., 2017). The paper notes that the two environmental hazards are both found to increase the risk of ischemic heart disease. Given the high correlation between these pollutants, it is important to investigate combined effects, in relation to myocardial infarction (MI). This study, among 50,744 middle-aged Danes enrolled into the Diet, Cancer and Health cohort from 1993 to 97, identified 2403 cases of incident MI during a median follow-up of 14.5 years. Present and historical residential addresses from 1987 to 2011 were found in national registries, and traffic noise ($L_{den}$) and air pollution ($NO_2$) were modelled for all addresses.

The result of analysis was that road traffic noise and $NO_2$ were both individually associated with a higher risk of MI, with hazard ratios of 1.14 (1.07–1.21) and 1.08 (1.03–1.12) per inter-quartile range higher 10-year mean of road traffic noise and $NO_2$, respectively. Mutual exposure adjustment reduced the association with 10-year $NO_2$ exposure (1.02 (0.96–1.08)) – that is, not significant - whereas the association with road traffic noise remained: 1.12 (1.03–1.21). In summary, in two-pollutant models, mainly noise was associated with MI. Combined exposure to both pollutants was associated with the highest risk.

The second study also focussed on correlations of noise air pollution measurements (Dekoninck & Severijnen, 2022). The paper It is a complex, but interesting, analysis of measured data. The primary argument made is that the noise-air pollution relationship is complex. For those components of air pollutants that directly relate to the combustion processes of vehicles, the spectral noise content associated with the road traffic noise predictions provides significant added value in being able to estimate those pollutants. They summarize this as "noise-as-a-proxy", with covariates providing indirectly measured attributes re vehicle flow dynamics to improve air pollution prediction estimation. This is a brief summary of the original work which should be consulted for more details. However, the "noise as proxy" concept is one that might usefully be explored further. For example, Dekoninck and Severijnen (2022) indicate that the presented results show the first direct evidence that the noise-as-a-proxy methodology will work as well for NO2, NO and NO.

## TRAFFIC NOISE MITIGATION AND PROPERTY VALUES

Examining any association between levels of road traffic noise ad property values was not in the brief for this scoping study – hence there has been no search made of the literature on this topic. However, one Swedish paper – '*A sound investment? Traffic noise mitigation and property values'* - did turn up, from the *Journal of Environmental Economics and Policy,* as a result of searching for noise interventions (Lindgren, 2021b). The study estimates the benefits of noise mitigation by its capitalization into property prices. The author evaluates a national noise mitigation programme run by the Swedish Road Administration (SRA) that built noise barriers and installed facade insulation in dwellings. Since the programme's outset in 1998, the SRA has invested more than 1 billion SEK ($100 million) in noise mitigation measures (Swedish Transport Administration 2018[71]). Consistent with programme guidelines, the measures were targeted to properties exposed to noise levels above certain limits (Swedish Road Administration 2001). The programme staff determined properties' programme eligibility by conducting noise assessments to establish whether exposure exceeded the limits and judging whether noise mitigation provision would be technologically and economically feasible.

Data were matched to register data on the attributes and sales price of the subset of properties sold from 1999 through 2017. Treated properties were those receiving noise mitigation by the SRA during the period of study and control properties as those that did not. The impact of the programme was examined in a difference-in-differences model that compared the change in sales price, before relative to after an SRA assessment, between treated and control properties. There were no meaningful differences in levels or trends in observed attributes between the treatment and control group. The abstract for the paper reads:

> *'A key policy question is how the benefit of noise mitigation compares with the cost. This study estimates the benefits of noise mitigation by its capitalization into property values. Using a dataset on properties considered for a noise mitigation programme, I estimate a difference-in-differences model that compares prices of properties receiving a measure to properties ineligible for the programme. Results show that noise mitigation raised property prices by 10–12 percent. The property price benefits exceed programme investment cost with each $1 spent on noise mitigation generating up to $1.7 in benefits'.*

NOTE:
The following paper (Lindgren, 2021a) could have been reported under various other sections of this scoping document, *viz.: 'Cardiovascular and Metabolic Effects'* or *'Interventions & Change Effects*'. It is reported here because it is the same author and uses data from the same Swedish national noise mitigation programme, as the property values paper above. The paper appears to be an internal report of the Swedish National Road and Transport Research Institute, and there is no indication as to whether the paper has been peer reviewed. The abstract of the paper reads:

> *'This study investigates the health effects of a nationwide program that provided noise mitigation to dwellings. The analysis uses hospitalization records and a difference-in-differences model that compares residents in treated homes to those with similar at-tributes in untreated homes. Results show that noise mitigation measures lower the risk of cardiovascular diseases by 10% after seven years, with effects driven by reduced risk of hypertension. Health effects are larger among the population exposed to higher baseline noise levels. These findings suggest that implementing similar noise mitigation measures will produce meaningful health benefits'..*

[71] English language versions of the Swedish Transport Administration document (and the Swedish Road Administration document) have not been sighted by the author of this scoping review.

# OTHER TOPICS

## GREEN WALLS

Yan et al. (2022) provide a recent review paper from China on the application of green wall technology in the acoustic field – including for road traffic noise reduction. It reviews some 34 papers from the last ten year on green wall technology, including green façades and living walls. Vegetation morphology and properties of substrates are the two crucial factors determining green facades and living walls' acoustic properties, respectively. High installation, operation, and maintenance costs and plant-induced problems are recognized as the main obstructions.

## ROAD TRAFFIC NOISE AND CYCLISTS/ PEDESTRIANS

This matter was addressed by several authors towards the end of the review period and may be a matter which needs to be considered in any future revision of the Road Noise Policy. It appears to be a relatively recent focus, and its small literature is not examined in full here. There are currently several papers that give a useful overview of the field (often in combination with studies of cyclist and pedestrian exposure to air pollution), either as literature review, or as comparative overview of approaches used in international studies, and some also examine cyclists and pedestrian perceptions or noise and air pollution during their commute: for example (Gelb & Apparicio, 2021, 2022; Marquart et al., 2022) (Marquart, 2022)

.

From their review (Gelb & Apparicio, 2021) and elsewhere, some conclusions are that:

- the literature identified six main subfields: the observation of cyclists' exposure, the intermodal comparison of exposures, the modelling of the exposure, the impacts on health, planning of itineraries, and the perceptions about the exposures.
- exposure to noise is rarely studied. But the few studies that do lead to belief that cyclists' exposure to noise is superior to that of other road users and has a low correlation with atmospheric pollution.
- some studies examine propose itineraries to minimise cyclists' exposure. The results may be used, either by individuals or by urban planners (to determine the best axes for the extension of a cycling network). However, planning assistance tools are still in their early stages since they do not integrate other dimensions such as connectivity to the cycling network, cyclists' actual trips, safety, and noise exposure.
- studies on cyclists' perceptions and behaviours regarding these pollutants are limited
- various studies use wearable devices to examine exposures to traffic noise and air pollution
- studies place the individual at the centre of the issue of exposure. While atmospheric pollution does not seem to be a major obstacle to cycling, it is negatively perceived by the majority of cyclists and they develop strategies to minimise their exposure (alternate routes, masks, etc.). However, the willingness to reduce their exposure conflicts with their preference for direct itineraries and cycling infrastructure that may sometimes lead to situations of overexposure.

A field study in Copenhagen, Paris and Montreal (Apparicio & Gelb, 2020) lead to the following results:

*'Earlier studies suggest that cyclists are exposed to higher noise levels than motorists. Other studies have demonstrated that cyclists' exposure to noise could vary significantly according to their routes. The aim of this study is to compare cyclists' exposure to noise and their determinants in three cities. Three participants cycled equipped with noise dosimeters and GPS watches: 1823, 967, and 1362 km in Copenhagen, Paris, and Montreal, respectively. We fitted three generalized additive mixed model with an autoregressive term model to predict the cyclists' exposure to noise according to the type of route and bicycle infrastructure after controlling for the day of the week, as well as spatial and temporal trends. The overall noise means were 73.4, 70.7, and 68.4 dB(A) in Paris, Montreal, and Copenhagen, respectively. The exposure to road traffic noise is strongly associated with the type of bicycle infrastructure used by the cyclist; riding on a cycleway significantly decreases it, while riding in a shared lane has no impact. Our findings demonstrate that it is possible to achieve a substantial reduction in cyclists' exposure by adopting new practices that include noise exposure in the planning of future cycling infrastructure.'*

In a choice based study (Bunds et al., 2019) recreational walking decisions in urban away-from-home environments, walkers were asked the relevance to them of air quality, noise, traffic, and the natural environment. In total, 501 US residents participated in an adaptive choice based conjoint study. The following seven attributes were considered (with three levels each): air pollution level, air pollution source, noise level, noise source, natural environment, traffic, as well as walking time and distance. Air pollution level was the most important attribute, followed by traffic, noise level, and the natural environment. The results are shown in Figure 84.

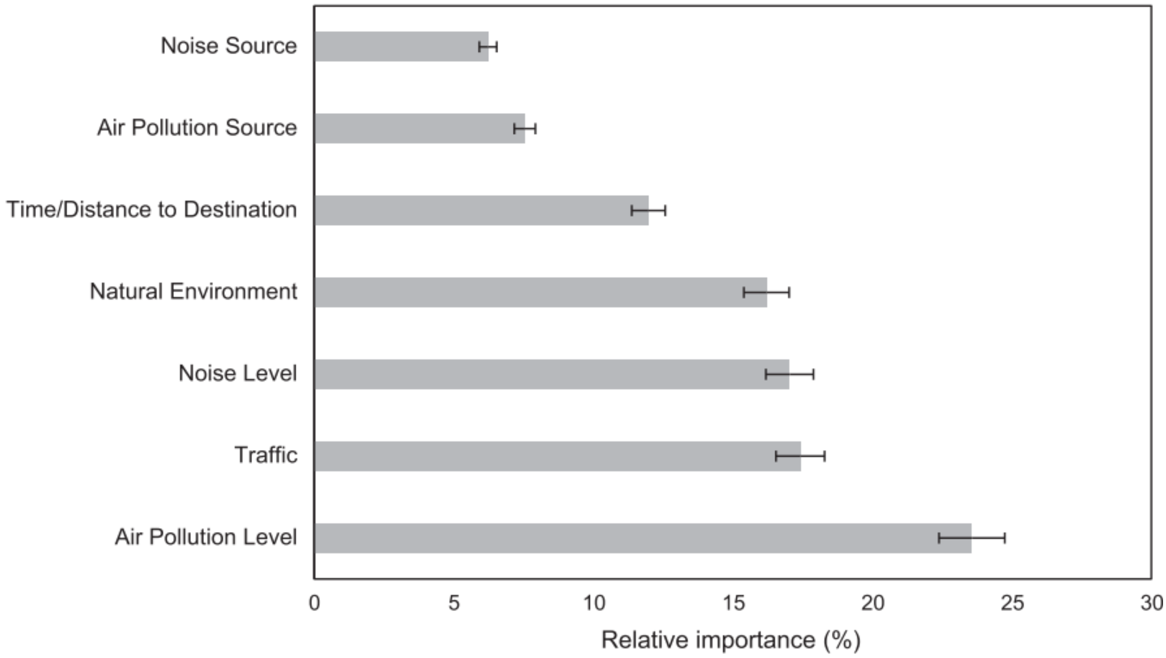



**Figure 84 Average importance of route choice attributes (Bunds et al., 2019).**

## SOME INTERNATIONAL/NATIONAL PROJECTS RELEVANT TO THIS BRIEF

A desk review of current scientific literature on road traffic noise would not be complete without identifying some major research projects/activities/consortia with respect to road traffic noise. European projects are generally identified by their acronyms. Most of these are European based – part of the European Commission framework programmes of research, *Horizon 2020* (and its predecessors, FP5, FP6, FP7) and its replacement since January 2021, *Horizon Europe,* and the *LIFE* programme. These are often multi-million Euro projects involving many institutes. Projects listed here have been identified, to 2016, by a systematic review (Alves et al., 2016) – this systematic review was of European projects addressing 'urban sound planning topics' and also identified the most relevant research results that can be applied by practitioners. After 2016, the list below is an ad-hoc compilation of international and national projects familiar to this author. They thus may not form a complete list. They are not presented in any order.

Where relevant, published information arising from the projects has been incorporated into the body of this review. Where that is not the case, this section merely reports the acronym and sometimes a brief summary of the project intent - it being useful to identify them in this way for an Australian reader. Many of the projects are multi-faceted with many proposed outputs by different teams, and more detailed information can quickly be located by an internet search on the project name.

**NEMO** Noise and Emissions Monitoring and Radical Mitigation (NEMO), https://nemo-cities.eu/ EU Horizon 2020 Remote sensing technologies for road and rail vehicle noise emission and the infrastructure related measures to reduce road traffic noise (van Blokland et al., 2020).

**NADIA** Noise Abatement Demonstrative and Innovative Actions and information to the public. The work involved Italian provinces and municipalities and went beyond the minimal mapping and reporting requirement of the END. It mapped critical areas for attention, prioritized them, conducted a cost-benefit analysis to determine the efficiency of each anti-noise measures, and undertook dissemination activities with the communities involved. The project outcomes and lessons are reported in Schiavoni et al. (2015).

**HARMONICA** HARMOnised Noise Information for Citizens and Authorities. This project is a partnership between **Acoucité** and **Bruitparif** in the context of European research program Life+. The goal of this project was to show the contribution to create a normalized noise index for the public and the public authorities. This index results from measurements and is published on a common web portal www.noiseineu.eu.Provide an easy access to information on the sound environment situation for the public and local and regional authorities to make this information understandable, and harmonize methods and means in order to compare the different territories and assess the noise mitigation actions. HARMONICA took place for 40 months, from 2011 to 2014. A conference paper describing HARMONICA is at: https://www.bruitparif.fr/pages/En-tete/300%20Publications/680%20Articles%20scientifiques/2013%20-%20HARMONICA%20project.pdf

**PHENOMENA** Assessment of Potential Health Benefits of Noise Abatement Measures in the EU. It was designed as a response to the perceived lack of effectiveness in the various rounds of Noise Action Plans– 'the objective of the study was to support the European Commission in defining noise abatement measures capable of delivering a 20–50 percent reduction of the health burden due to environmental noise'. Elements of the study included:

- A review of international and EU literature as well as EU and member state legislation
- Assessment of noise action plans (NAPs) and their implementation and enforcement
- Identification and assessment of legislative drivers of noise abatement solutions
- Listing of good practices
- Health impact assessment and cost-benefit analysis (CBA) of costs of abatement against health benefits of reduced exposure
- Assessment of available noise abatement solutions
- Scenario analysis of noise abatement solutions
- Proposals for EU and member state policies to reduce the health burden

The study is reported in full in Salomons et al. (2021) but a summary is available in *Noise/News International* (Dittrich M, 2021). For road traffic noise, the focus was on situations with long-term noise exposure levels above 53 dB $L_{den}$ - the exposure limit recommended in the Environmental Noise Guidelines (World Health Organization, 2018) – and as shown above, a much lower exposure level than currently set by most European Member State legislation (while the potential for contribution of peak levels of noise to health effects was acknowledged, they were not included within the PHENOMENA study).

Largely, the literature review in PHENOMENA provides a 'state-of-the-art' of road traffic noise management practice in Europe. This author's opinion is that (despite the Salomons et al. (2021) report being 815 pages) it was, rather disappointing that there was not much that is new in terms of management strategies…a repeat of the same practice as applied over the last 40 or more years. In summary, in terms of management practice, there is little to inform revision of the NSW RTNP. However, two matters are worthy of consideration for future Australian practice: the approach to cost benefit analysis of noise mitigation and health effects, and the attempts to address 'sub-criteria' exposures across the community – i.e. to reducing exposure over community even when these do not exceed criterion in order to reap the health benefits of doing so (in other words, not just focussing road traffic noise mitigation at lopping the tops off highest levels that exceed criterion).

**HOSANNA** Holistic and sustainable abatement of noise by optimized combinations of natural and artificial means 2009-2013. Euro 5m. *Novel solutions for quieter and greener cities*. Hosanna aims were to optimize the use of green areas, green surfaces, and other natural elements in combination with artificial elements in urban and rural environments for reducing the noise impact of road and rail traffic. The main objectives of the project are: to show by full scale evaluation that the proposed abatement methods work; to deliver noise prediction methods applicable to the proposed abatements, which can also be used in noise mapping software; to deliver assessment methods for the perceived noise environment; to deliver a good practice guide for the end-users; and to show the cost benefit, including the positive effect on urban air quality and CO2 neutrality, of the resulting noise abatement methods.

Results indicate that acoustically absorbent barriers of about 1 metre (m) lower traffic noise by 8 dBA at 1.5 m and by 6 dBA at 4 m. With the addition of lane barriers, road and tram noise can be reduced by more than 10 dBA. Furthermore, a 15 m belt of trees lowers noise levels by 3 dBA; similar reductions were obtained from typical grass covers. Other devices — including roughening elements on smooth ground and low, parallel walls — also resulted in an improvement. Modelling predicted that greening a roof would lower noise to an inner yard by 3 to 8 dBA. A low, vegetated barrier was built beside a road in Lyon, France. The effect was assessed using measurements in conjunction with responses to questionnaires. Other devices were also tested at other European sites.
The output is described in a ten-chapter handbook (Nilsson et al., 2015) and a 46 page brochure (Anon, 2013).

**SONORUS** was a training network in urban sound planning (http://www.fp7sonorus.eu/). It applied a holistic approach to the acoustic design of cities, integrating planning skills with noise prediction methods, soundscaping and noise control engineering. The focus was on planners and architects and the development of a professional – the urban sound planner. An example of output is the PhD dissertation from Chalmers University of Technology, Gothenburg (Estévez-Mauriz, 2020).

**SiRENE** http://www.sirene-studie.ch Little is known about how acute and short-term noise effects, especially those that are observed during sleep, translate into long-term health consequences. In particular, it is unclear which acoustical characteristics of noise from different traffic sources are most detrimental for human health and wellbeing. The overall goal of this (Swiss TPH) project is to investigate acute, short- and long-term effects of road, railway and aircraft noise exposure on annoyance, sleep disturbances and cardiometabolic risk. The project aims at identifying noise exposure patterns that most strongly affect individuals during sleep and thus may ultimately result in long-term health consequences. In addition, the role of individual characteristics such as age and gender, noise sensitivity and genetic predispositions will be elucidated (Röösli et al., 2017).

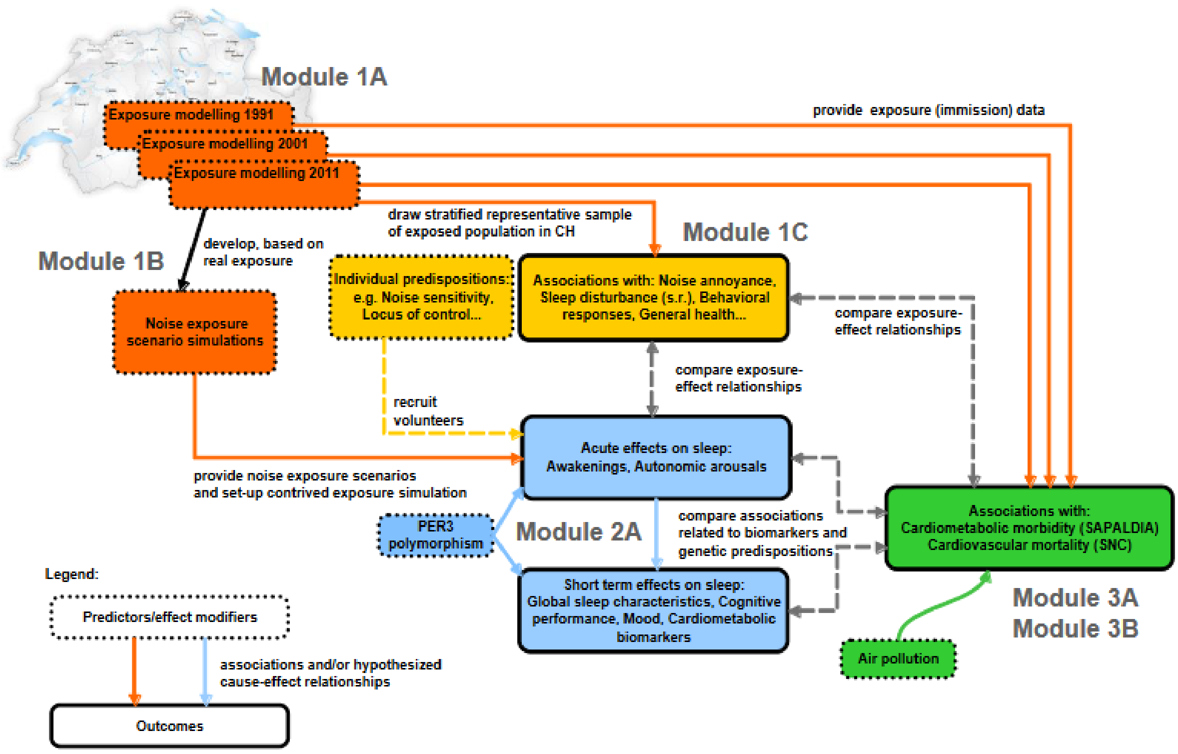



Figure 85 Overall project framework of SiRENE (Röösli et al., 2017).

**TD0804 COST ACTION – Soundscapes of European Cities and Landscapes.** The main aim of the Action was to provide the underpinning science for soundscape research and make the field go significantly beyond the current state-of-the-art, through coordinated international and interdisciplinary efforts. The Action promoted soundscape into current legislations, policies, and practice, aiming at improving/preserving our sonic environment. It involved 52 participants from 23 COST countries and 10 participants outside Europe. Much of the findings are reported in *Soundscape and the Built Environment (*Kang & Schulte-Fortkamp, editors. 2016).

**EQUAL-LIFE** Early **E**nvironmental **qual**ity and **life**-course mental health effects. Euro12m 2020-24 (van Kamp et al., 2022). Mental health is the result of the complex interplay between genetic, psychological, environmental, and other factors and experiences. The exposome concept, referring to the totality of exposures from conception onwards, is emerging as a very promising approach in studying the role of the environment in human disease. The EU-funded Equal-Life project will develop and utilise the exposome concept in an integrated study of the external exposome and its social aspects and of measurable internal physiological factors and link those to a child's development and life course mental health. This will be done using a novel approach combining exposure data to characterise, measure, model and understand influences at different developmental stages. The goal is to propose the best supportive environments for all children.

While this project is described in terms of the 'exposome' – the totality of physical and social exposures of a child – the project arose out of a consortium that was focussing on the effect of noise exposure (transport sources) on a child's development.

**PIXEL** (Horizon 2020). This project has only tangential relevance for of road traffic noise, as the immediate focus is elsewhere. It is recorded here as may prove of specific value in future issues re road traffic noise and port facilities. Its focus is on ground transportation associated with major port activities: https://cordis.europa.eu/project/id/769355 PIXEL enables a two-way collaboration of ports, multimodal transport agents and cities for optimal use of internal and external resources, sustainable economic growth and environmental impact mitigation, towards the Port of the Future.

**NORAH** Noise-Related Annoyance, Cognition, and Health (Schreckenberg et al., 2011). Norah focussed on Frankfurt (and other airports) aircraft noise – but much work was also relevant to road traffic noise, including what impact does traffic noise have on the quality of life, health and development of children. NORAH calculated the past and current, address-specific exposure to aviation, road and rail noise in a large area around the Frankfurt Airport. The researchers compared these values with data on the health, quality of life and development of a total of more than one million persons in the region. In addition to this, several thousand people in the areas around the airports Berlin Brandenburg, Cologne/Bonn and Stuttgart were surveyed. In addition to the calculation of noise data important for the overall study, NORAH was also engaged in five sub-studies on the impact of aviation, road and rail noise on people, viz:

- The Quality of Life Study carried out surveys to investigate which noise burdens have an impact on the experienced noise annoyance and quality of life.
- The Study on Health Risks examined whether disorders such as cardiovascular disorders or depression occur more frequently in people exposed to noise.
- The Blood Pressure Study examined the link between blood pressure and noise exposure.
- In the Sleep Study the scientists examined the sleep quality of the study subjects in terms of the sound pressure in the bedroom.
- The Child Study looked at the intellectual development of primary school children as well as their reading abilities and their general quality of life under noise exposure.

The NORAH study was the main focus of ICANA Health 2015 – a major conference held at Frankfurt Main airport Frankfurt, November 2015.

**FOREVER** Future Operational Impacts of Electric Vehicles on National European Roads – 2013-2014. The use of electric vehicles (hybrid or fully electric) in European road networks is increasing. While the focus of noise related research has been on the use of these vehicles in low-speed urban areas, in particular with regard to the safety risks for vulnerable road users, so far, few studies have been carried out on the potential noise exposure of electric vehicles on roads under the responsibility of National Road Authorities (NRAs), namely motorways and other main roads. The FOREVER project aimed to address acoustic issues related to e-mobility in the higher-level road network by providing data and information on three topics:

- first the identification of the noise emission levels from electric vehicles (powertrain and rolling noise components) at speeds representative of NRA roads, including the impacts of added alert sounds and the development of input data for the CNOSSOS-EU noise model,
- second the noise emission from low-noise tires, and
- third an estimation of the noise impacts of electric vehicles and low-noise tires on NRA roads, based on different fleet compositions and different take-up rates of electric vehicles.

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