ETC HE Report 2023/11

Environmental noise health risk assessment: methodology for assessing health risks using data reported under the Environmental Noise Directive

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European Environment Agency
European Topic Centre
Human health and the environment

Cover design: EEA
Cover image © Jana Sönsken, Swiss TPH
Layout: EEA / ETC HE

Publication Date: July 2024
ISBN 978-82-93970-35-4

How to cite this report:
Engelmann, N., Blanes Guàrdia, N., Fons Esteve, J., Vienneau, D., Röösli M. (2023). *Environmental noise health risk assessment: methodology for assessing health risks using data reported under the Environmental Noise Directive* (Eionet Report – ETC HE 2023/11). European Topic Centre on Human Health and the Environment.

The report is available from https://www.eionet.europa.eu/etcs/all-etc-reports and https://zenodo.org/communities/eea-etc/?page=1&size=20.

EEA strategic no. 3 (2021-2030) Human health and the environment

ETC HE coordinator: Stiftelsen NILU, Kjeller, Norway (https://nilu.com)

ETC HE consortium partners: Federal Environment Agency/Umweltbundesamt (UBA), Aether Limited,
Czech Hydrometeorological Institute (CHMI), Institut National de l'Environnement Industriel et des Risques (INERIS),
Swiss Tropical and Public Health Institute (Swiss TPH), Universitat Autònoma de Barcelona (UAB),
Vlaamse Instelling voor Technologisch Onderzoek (VITO), 4sfera Innova S.L.U., klarFAKT e.U

European Topic Centre on
Human Health and the Environment (ETC HE)
https://www.eionet.europa.eu/etcs/etc-he

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Acknowledgements

The EEA task manager was Eulalia Peris. Many thanks for the inspiring discussions and helpful feedback to draft reports. Development of the health risk assessment methods described in this report have been started with a consulting project for WHO Regional Office Europe entitled "Recommendations for future health risk assessments of environmental noise". In this context we obtained feedback from the WHO steering group (Dorota Iwona Jarosinka, Pilmu Ryu, Roman Perez Verlasco), our project partner Jördis Wothge (Umweltbundesamt, Germany) and several reviewers.

Executive Summary

Since the last health risk assessment (HRA) of the European Environmental Agency (EEA, 2020a), a growing body of evidence on the effects of environmental noise on health has been published. This has substantial implication for up to date HRA studies in Europe. The aim of this report is to critically evaluate the existing HRA method of the EEA for Europe and propose adaptations to the previously used methodology where necessary. In particular, a method to assess noise exposure with a finer resolution of 1 dB and suitable for exposure assessment below the Environmental Noise Directive (END) thresholds has been developed (summarized in PART I of the report). Further, an Umbrella+ review and meta-analyses were conducted to determine critical health outcomes and derive latest exposure-response functions (ERF) including effect threshold (PART II). In PART III disability weights (DW) for calculating disability adjusted life years (DALYs) and monetary value of DALYs were evaluated.

As done in EEA (2020a), high annoyance and high sleep disturbance for road, rail and aircraft are suggested to be quantified using the ERF described in the Environmental Noise Guidelines for the European Region from the World Health Organization (WHO ENG) and for industrial noise the ERF from Miedema and Vos (2004, 2007) (Table 1.1). The previous ERF from Clark et al. (2006) and van Kempen (2008) should also not be changed for the quantification of reading impairments in children due to aircraft noise.

For other health effects, the evidence for an association with noise has substantially increased, thus more outcomes should be considered in addition to ischaemic heart disease, as done in EEA (2020a). Many recent studies show negative effects of transport noise on various cardiovascular outcomes such as ischaemic heart disease, heart failure, hypertension and stroke. It is thus suggested pooling a relationship for all types of cardiovascular outcomes using available high-quality epidemiological research. A meta-analysis of studies addressing incidence of various cardiovascular diseases yielded a relative risk increase of 1.032 (95%-CI: 1.012-1.052) per 10 dB L_{den} increase in road traffic noise (Table 1.1).

For mortality, new high-quality cohort research demonstrates consistent associations with all-cause mortality and not only cardiovascular disease mortality. This is plausible given the systemic stress effect of noise. Thus, we suggest quantifying burden from all-cause mortality instead of ischaemic heart disease mortality as done in the former EEA HRA (EEA, 2020a). Pooling seven cohort studies found a relative risk increase of 1.055 (95%-CI: 1.014-1.069) per 10 dB L_{den} increase in road traffic noise (Table 1.1).

Eleven cohort studies investigated the association between transportation noise and diabetes. Based on high certainty evidence for an association with road traffic noise, diabetes is suggested to be included in a future HRA (relative risk of 1.062 (95%-CI: 1.036-1.088) per 10 dB road traffic noise increase).

Studies on the various health effects showed mostly higher certainty of evidence for road traffic noise than studies for railway and aircraft noise. Therefore, the relationships for road traffic noise are proposed to be used to estimate the impacts of rail and aircraft traffic noise on all-cause mortality, cardiovascular diseases and diabetes. It is plausible that biological mechanisms are similar for various transportation noise source.

Further, it is advised, that in a second step, additional health outcomes are considered in future HRAs. This includes depression, cognitive impairment in adults, dementia and behavioural problems in children and adults.

Overall, the new body of evidence shows negative effects due to transport noise at much lower levels than those captured in the END exposure assessments (i.e. 55 dB L_{den}, and 50 dB L_{night}). Therefore, health risks of noise should be quantified at levels starting at 45 dB L_{den} and 40 dB L_{night}. A method for estimating the number of people exposed to noise levels below the END thresholds is described in PART I.

The relationships and DWs used to estimate the burden of disease attributable to transportation noise for the adult population in Europe in 2022 are described in Table 4.1. The proposed source for country specific health data is the 2019 Global Burden of Disease (GBD) study. As regards to quantifying economic costs of health risks at European Union (EU) level, it is proposed to use a monetary value for a DALY of 70,000 EUR to be consistent with other EU studies on the monetisation of noise from transport.

In conclusion, the proposed changes of the methods for the HRA of environmental noise in the EU reflects recent progress in noise research. It is expected that applying the new methods will considerably increase the burden of disease attributable to transportation noise in the EU compared to previous EU-wide HRA.

Table 1.1: Overview of the proposed ERFs and outcomes to be used in an EU-wide HRA

Outcome	Source	ERF	Reference
High noise annoyance (prevalence in adults)	Road	$\%HA = 78.9270 - 3.1162 \cdot L_{den} + 0.0342 \cdot L_{den}^2$	Guski et al. (2017)
	Railway	$\%HA = 38.1596 - 2.05538 \cdot L_{den} + 0.0285 \cdot L_{den}^2$	Guski et al. (2017)
	Aircraft	$\%HA = -50.9693 + 1.0168 \cdot L_{den} + 0.0072 \cdot L_{den}^2$	Guski et al. (2017)
	Industry	$\%HA = 1\text{-normal } (72 - (-126.52 + (L_{den}) \cdot (2.49)))/\text{sqrt}(2054.43))$	Miedema and Vos (2004)
High sleep disturbance (prevalence in adults)	Road	$\%HSD = 19.4312 - 0.9336 \cdot L_{night} + 0.0126 \cdot L_{night}^2$	Basner and McGuire (2018)
	Railway	Railway: $\%HSD= 67.5406 - 3.1852 \cdot L_{night} + 0.0391 \cdot L_{night}^2$	Basner and McGuire (2018)
	Aircraft	$\%HSD=16.7885 - 0.9293 \cdot L_{night} + 0.0198 \cdot L_{night}^2$	Basner and McGuire (2018)
	Industry	$\%HSD=1\text{-normal}(72 - (-90.70 + (L_{night}) \cdot (1.80)))/\text{sqrt}(1{,}789 + 272))$	Miedema and Vos (2007)
All-cause mortality (adults)	Road, rail and aircraft	Relative risk (RR) derived from road noise RR=1.055 (95%-CI: 1.014-1.069) per 10 dB L_{den}	Meta-analyses Chapter 3.3.1
Cardiovascular disease (incidence in adults)	Road, rail and aircraft	Relative risk (RR) derived from road noise RR=1.032 (95%-CI: 1.012-1.052) per 10 dB L_{den}	Meta-analyses Chapter 3.3.2
Diabetes (incidence in adults)	Road, rail and aircraft	Relative risk (RR) derived from road noise RR=1.062 (95%-CI: 1.036-1.088) per 10 dB L_{den}	Meta-analyses Chapter 3.3.3
Reading Comprehension (prevalence in children)	Aircraft	$1/(1 + \exp(- (\ln(0.1/0.9) + (\ln(1.38)/10 \cdot (L_{den} - 50))))$ if $L_{den} \geq 50$ dB and 0.1 if $L_{den} < 50$ dB	Clark et al. (2006) and van Kempen (2008)

1. Introduction

1.1. Background

A methodology to undertake health risk assessments (HRA) on noise data collected under the Environmental Noise Directive (END) was developed by the European Topic Centre on Air pollution and Climate change mitigation (ETC/ACM) in 2018 (ETC/ACM, 2018). This HRA methodology was used to assess the health risks in the report *Environmental Noise in Europe 2020* (EEA, 2020a), the briefing *Health risks caused by environmental noise in Europe* (EEA, 2020b) and in other noise-related products developed by the European Environment Agency (EEA) between the period 2018-2022. Main results were that long-term exposure to environmental noise causes 12,000 estimated premature deaths and contributes to 48,000 new cases of ischaemic heart disease per year in the European territory (EEA, 2020a). It also estimated that 22 million people suffer chronic high noise annoyance, and 6.5 million people suffer chronic high sleep disturbance. As a result of aircraft noise, 12,500 schoolchildren are estimated to suffer learning impairment.

The European noise health risk estimates outlined above were based on exposure-response functions (ERFs) presented in the *Environmental Noise Guidelines for the European Region from the World Health Organization* (WHO ENG) (WHO Europe, 2018), and included annoyance, sleep disturbance and incidence of ischaemic heart disease. These exposure-response relationships were legally adopted in Annex III of the END in 2020 (EC, 2019, 2020). Further, other outcomes such as premature mortality due to ischaemic heart disease and the effects of aircraft noise on children's reading comprehension were included in the European health risk assessment following a recommendation made by van Kamp et al. (2018).

However, the methodology used for the HRA (EEA, 2020a) did not account for exposures in areas not covered by the END (i.e. areas below 55 dB L_{den} and 50 dB L_{night}, roads outside urban areas below 3 million passes a year, railways outside urban areas below 30,000 passes a year, airports outside urban areas below 50,000 movements a year, and agglomerations below 100,000 inhabitants). Therefore, the health impacts due to environmental noise calculated in previous EEA assessments are likely to be underestimated (EEA, 2020a).

The methodology by ETC/ACM in 2018 (ETC/ACM, 2018) is to be revised and updated based on the new 2022 strategic noise maps collected under the END and on new evidence regarding the health impacts of noise. Using the 2022 strategic noise maps, this report describes a methodology for quantifying the health burden of environmental noise in Europe from road traffic, railways, aircraft and industry. Specifically, the report describes the health outcomes to be used in the updated European HRA, the exposure-response relationships to be used for each health outcomes selected, the disability weights (DWs) to be used for the Burden of Disease (BoD) calculations in terms of disability-adjusted life years (DALYs), and the values to be used for the monetization of health impacts. In addition, the report describes an updated method for estimating population distributions below the END reporting thresholds that can be used to assess health risks more broadly.

The new methodology described here for calculating the HRA of environmental noise in Europe will feed into the forthcoming 2nd zero pollution assessment, the Noise in Europe report, State of the Environment Report (SOER) and other EEA noise related products.

1.2. Objectives

The aim of this report is to critically evaluate the methods for future HRA and propose adaptations to the previously used methodology where necessary. In particular, the current methodology is evaluated and extended to include the follow aspects:

- A method to assess noise exposure with a finer resolution of 1 dB and also suitable for exposure assessment below the END thresholds.
- Relevant health outcomes to be included in the future HRAs on the basis of up-to-date scientific evidence.
- Critical evaluation of ERFs for all selected outcomes, with updating as needed.
- Determination for each ERF the effect threshold, below which health effects are unlikely.
- Determination of the cut-off level below which no health effects occur for each ERF.
- A revisit of the DWs for calculating DALYs.
- Determination of a monetary value to noise DALYs.

The report consists of three parts, each dealing with different aspects of the updated methodology. PART I describes the assessment of noise exposure for the European population below the END threshold. PART II describes the evaluation of scientific evidence. PART III describes the calculation approach and the monetarization.

2. PART I: Assessment of noise exposure below the END thresholds

Negative health effects start to occur below the obligatory END thresholds of L_{den} 55 dB and L_{night}. The WHO recommends reducing noise levels to 53 dB L_{den} and 45 dB L_{night} for road traffic, 54 dB L_{den} and 44 dB L_{night} for rail traffic, and 45 dB L_{den} and 40 dB L_{night} for air traffic. Recent research suggests health effects even below these WHO guidelines (see Chapter 3.4.3).

In order to estimate the health risks due to environmental noise in the European area, an extrapolation of the number of people affected by noise at levels below the END thresholds is needed.

Previous EEA assessments used the methodology described in Blanes et al. (2019) to estimate the number of people exposed below the END thresholds, which was based on previous work developed by Alberts et al. (2016) and Houthuijs et al. (2018).

In Alberts et al. (2016), relative fractions depending on the number of people exposed above the END thresholds were estimated with a normal distribution for road noise exposure inside agglomerations. An exponential distribution for major road noise exposure was used outside agglomerations to account for the skewed distribution with a lower proportion of highly exposed people. Further, refinements were done by Houthuijs et al. (2018), who used more updated exposure information and extended the estimations to the other noise sources considered in the END. They found that exponential distribution would need to be applied to rail noise exposure, aircraft noise exposure and industrial noise exposure inside agglomerations, and to major sources of railway noise and aircraft noise outside agglomerations. Relative mean fractions used in the calculations for aircraft noise were extracted from ANOTEC study (2003) and updated in Houthuijs et al. (2018). Relative mean fractions used in the calculations for industrial noise exposure inside agglomerations were based on data from Netherlands Environmental Assessment Agency about exposure to industrial noise in Dutch population (Hoogervorst, 2009)).

In the case of estimating the number of people exposed below the END thresholds for road noise inside agglomerations, the method proposed in Alberts et al. (2016) was improved in Houthuijs et al. (2018) by providing exposure values discriminated at 1dB per each EU member country, resulting in the establishment of individual relative fractions to be used to estimate exposure to lower levels per each individual country. The relative fractions per country are published in Houthuijs et al. (2018) and updated in Blanes et al. (2019).

This methodology was based on population exposure data from 2012, and applied to population exposure data from 2017. Population distribution may change because of the new CNOSSOS methodology employed for the calculation of strategic noise maps (Commission Directive (EU) 2015/996 of 19 May 2015 establishing common noise assessment methods to Directive 2002/49/EC of the Parliament and of the Council). Therefore, the distributions described above by Alberts et al. (2016) and Houthuijs et al. (2018) are tested against new exposure data to validate their usability for the new HRA in Sections 2.1.1 - 2.1.7.

In addition, for the calculation of the HRA a non-uniform distribution across noise bands at 1 dB resolution is needed to more precisely calculate the health risks associated with exposure to noise (instead of using a mid-band exposure within the 5 dB bands). Therefore, Section 2.1.8 describes an improvement that is proposed to transfer exposure distribution from 5 dB to 1 dB.

2.1. Methods to estimate noise exposure below the END thresholds

The methodology implemented in previous exercises used noise exposure data from 2012 reference year (Blanes et al., 2019). No similar exercises were found in the literature to update or improve the current methodology at European scale. Some member countries were consulted to collect any case studies done at country or local level. Very few calculations considered noise levels below the END thresholds and in some cases, the exposure information to lower bands was not comparable to the data reported.

Considering the outcomes of the consultation to member states and the lack of results found in the literature, the method implemented in previous exercises is going to be used if the percentage of the exposed data above the END thresholds are equivalent comparing 2012 reference year, 2017 reference year and 2022 reference year.

The following sections detail the results encountered per each noise source in relation to the percentages of people exposed to different noise bands and the methodology that will be implemented to calculate the exposure to lower bands. A final Section (2.1.8) details the improvement that is proposed to transfer exposure distribution from 5 dB to 1 dB.

2.1.1. Agglomeration road

Comparison between the percentages used to calculate exposure values below the END threshold, using the following datasets:

- Gap filled dataset (reported and gap filled) for 2012 reference year following Alberts et al. (2016)
- Gap filled dataset (reported and gap filled) for 2012 reference year following Fons-Esteve et al. (2021)
- Gap filled dataset (reported and gap filled) for 2017 reference year following Fons-Esteve et al. (2021)
- Reported data for 2022 reference year, downloaded directly from Reportnet 3 on 26/09/2023.

The comparison of the percentage of people exposed per each L_{den} noise band in the different datasets can be seen in Table 2.1.

Table 2.1. Results of the comparison of percentages of people exposed for the different input datasets for road noise inside agglomerations

Input dataset	Percentage of people exposed					Total number of inhabitants inside agglomerations
	L_{den} 55–59 dB	L_{den} 60–64 dB	L_{den} 65–69 dB	L_{den} 70–74 dB	L_{den} ≥ 75 dB	
Gap filled dataset 2012 (Alberts et al., 2016)	15.24	11.93	8.56	4.12	0.75	187,000,000
Gap filled dataset 2012 (Fons-Esteve et al., 2021)	15.59	13.22	9.56	4.43	0.81	183,303,187
Gap filled dataset 2017 (Fons-Esteve et al., 2021)	16.90	13.57	9.34	3.96	0.64	188,779,599
Reported data for 2022 reference dataset downloaded from Reportnet 3 on 22/09/2023 (*)	20.69	14.03	9.24	3.62	0.53	41,442,365

(*) Data includes a total of 102 agglomerations distributed in a total of 9 countries: Austria, Czechia, Germany, Denmark, Estonia, France, Ireland, Latvia, Sweden. Not all the countries are complete.

According to the Kolmogorov-Smirnov test, the distributions of the share of people exposed to the 5 dB noise band were not statistically different ($p<0.001$).

Based on the above results, the methodology described in Houthuijs et al. (2018) and the relative fractions calculated at country level in Blanes et al. (2019) will be used to estimate the number of people exposed below the END thresholds for road noise inside agglomerations. Table 2.2 and Table 2.3 contain the relative fractions applied to estimate the exposure values below the END thresholds for road noise inside agglomerations for L_{den} and for L_{night}, respectively.

Table 2.2. Relative fractions per EU member country to calculate the exposure values below the END thresholds at 5 dB for L_{den} road noise inside agglomerations

ISO code	Fraction 25 – 29 dB	Fraction 30 – 34 dB	Fraction 35 – 39 dB	Fraction 40 – 44 dB	Fraction 45 – 49 dB	Fraction 50 – 54 dB
AT	0.001342	0.003916	0.015725	0.063988	0.260885	0.654145
BE	0.001432	0.004183	0.016730	0.067260	0.265214	0.645180
BG	0.001177	0.003435	0.013846	0.057102	0.244889	0.679551
CH	0.001284	0.003751	0.015140	0.061946	0.260698	0.657181
CY	0.002088	0.006159	0.025084	0.096767	0.354358	0.515545
CZ	0.001489	0.004364	0.017755	0.071739	0.292575	0.612078
DE	0.001425	0.004170	0.016857	0.068123	0.278499	0.630928
DK	0.001610	0.004728	0.019209	0.076541	0.301417	0.596494
EE	0.001332	0.003898	0.015814	0.064491	0.271124	0.643341
EL	0.001251	0.003645	0.014566	0.059559	0.243138	0.677841
ES	0.001422	0.004146	0.016478	0.066243	0.254400	0.657311
FI	0.001468	0.004304	0.017498	0.070589	0.288869	0.617272
FR	0.001451	0.004250	0.017245	0.069800	0.284199	0.623056
HR	0.001353	0.003964	0.016147	0.065827	0.276326	0.636384
HU	0.001451	0.004252	0.017252	0.069745	0.286228	0.621071
IE	0.001334	0.003900	0.015760	0.064279	0.268310	0.646416
IS	0.001387	0.004071	0.016716	0.068391	0.293790	0.615645
IT	0.001478	0.004323	0.017381	0.069610	0.273934	0.633275
LT	0.001468	0.004301	0.017443	0.070241	0.282463	0.624084
LU	0.00115	0.003354	0.013553	0.056359	0.248357	0.677227
LV	0.001514	0.004442	0.018108	0.072884	0.295542	0.607511
MT	0.001261	0.003679	0.014758	0.060384	0.252273	0.667645
NL	0.001337	0.003908	0.015815	0.064549	0.270775	0.643617
NO	0.001536	0.004508	0.018349	0.073722	0.297547	0.604338
PL	0.001419	0.004156	0.016877	0.068344	0.281698	0.627507
PT	0.001344	0.003916	0.015544	0.062804	0.246713	0.669679
RO	0.001350	0.003941	0.015778	0.063832	0.256685	0.658413
SE	0.001656	0.004866	0.019833	0.078786	0.306402	0.588457
SI	0.001416	0.004148	0.016798	0.067737	0.277322	0.632579
SK	0.001584	0.004652	0.019014	0.076450	0.306047	0.592253

Table 2.3. Relative fractions per EU member country to calculate the exposure values below the END thresholds at 5 dB for L_{night} road noise inside agglomerations

ISO code	Fraction 25 – 29 dB	Fraction 30 – 34 dB	Fraction 35 – 39 dB	Fraction 40 – 44 dB	Fraction 45 – 49 dB
AT	0.007424	0.027485	0.127368	0.350639	0.487085
BE	0.008036	0.029548	0.132636	0.354183	0.475597
BG	0.006215	0.023104	0.111736	0.340635	0.518310
CH	0.007253	0.026912	0.129207	0.360050	0.476579
CY	0.015229	0.056268	0.243336	0.406928	0.278240
CZ	0.009319	0.034735	0.164259	0.384039	0.407648
DE	0.008599	0.031837	0.149939	0.377951	0.431674
DK	0.010259	0.038105	0.174140	0.385200	0.392296
EE	0.007814	0.029077	0.140375	0.369646	0.453088
EL	0.006275	0.023244	0.106243	0.320486	0.543753
ES	0.007467	0.027440	0.118613	0.332258	0.514222
FI	0.009168	0.034092	0.161490	0.385600	0.409651
FR	0.008680	0.032350	0.151849	0.368954	0.438168
HR	0.008055	0.030103	0.145390	0.372088	0.444364
HU	0.008970	0.033317	0.158219	0.383204	0.416291
IE	0.007889	0.029262	0.139985	0.375211	0.447653
IS	0.008913	0.033479	0.167651	0.395313	0.394645
IT	0.008556	0.031561	0.142035	0.361768	0.456080
LT	0.008704	0.032408	0.149727	0.365750	0.443412
LU	0.006472	0.024077	0.120487	0.365855	0.483109
LV	0.009512	0.035507	0.166823	0.382995	0.405164
MT	0.007053	0.026031	0.123304	0.361260	0.482351
NL	0.007959	0.029541	0.142377	0.376853	0.443271
NO	0.009736	0.036262	0.169872	0.385903	0.398227
PL	0.008644	0.032152	0.152880	0.379392	0.426932
PT	0.007054	0.025856	0.114010	0.337499	0.515582
RO	0.007391	0.027239	0.124028	0.348914	0.492428
SE	0.010589	0.039467	0.177881	0.381533	0.390530
SI	0.008478	0.031417	0.147920	0.374699	0.437486
SK	0.010256	0.038421	0.178915	0.388492	0.383917

For those countries not included in the above list, the mean relative fraction per each noise exposure band is used. See the values in Table 2.4.

Table 2.4. Mean relative fractions to estimate the exposure values below the END thresholds for road noise inside agglomerations: for L_{den} and L_{night}

Indicator	Fraction 25-29 dB	Fraction 30 – 34 dB	Fraction 35 –39 dB	Fraction 40 – 44 dB	Fraction 45 – 49 dB	Fraction 50 - 54 dB
L_{den}	0.0014259	0.0041742	0.0168865	0.0682108	0.2771679	0.6321346
L_{night}	0.008524	0.031641	0.147617	0.36933	0.442888	-

The estimated population exposed to 50-54 dB band for L_{den} and for 45-49 dB band L_{night} is calculated using the total population exposed above 55 dB L_{den} and above 50 dB L_{night}, respectively. The following formulas are applied, with the corresponding relative fraction indicated per each indicator and per country (as included in Table 2.2 and Table 2.3 respectively, or Table 2.4 for countries not listed in the other two tables):

$$Population_{Lden\ 50-54\ dB} = \text{Country fraction 50} - \text{54 dB} * Population_{Above\ 55\ dB\ Lden}$$

$$Population_{Lnight\ 45-49\ dB} = \text{Country fraction 45} - \text{49 dB} * Population_{Above\ 50\ dB\ Lnight}$$

Subsequently, the same procedure is followed to estimate the lower noise bands, but taking as a starting point the population exposed above 50 dB for L_{den} and above 45 dB for L_{night}, by adding to the reported population the estimation done in the previous calculation. The formulas used to estimate the exposure values to END lower levels are the ones displayed below, using the corresponding relative fraction per each noise band and country as shown in Table 2.2, Table 2.3, or Table 2.4 for countries not listed in the other two tables:

$$Population_{Lden,5\ dB\ category} = Fraction_{Lden,5\ dB\ category} * Population_{Above\ 50\ dB\ Lden}$$

$$Population_{Lnight,5\ dB\ category} = Fraction_{Lnight,5dB\ category} * Population_{Above\ 45\ dB\ Lnight}$$

If exposure values below the END threshold have been reported for L_{den} and/or for L_{night}, the reported exposure values will be included in the corresponding noise band instead of the estimated population (see Annex 5).

2.1.2. Agglomeration rail

Comparison between the percentages used to calculate exposure values below the END threshold, using the following datasets:

- Gap filled dataset (reported and gap filled) for 2012 reference year following Fons-Esteve et al. (2021)
- Gap filled dataset (reported and gap filled) for 2017 reference year following Fons-Esteve et al. (2021)
- Reported data for 2022 reference year, downloaded directly from Reportnet 3 on 26/09/2023.

The comparison of the percentage of people exposed per each L_{den} noise band in the different datasets can be seen in Table 2.5.

Table 2.5. Results of the comparison of percentages of people exposed to the different input datasets for rail noise inside agglomerations

Input dataset	Percentage of people exposed					Total number of inhabitants inside agglomerations
	L_{den} 55–59 dB	L_{den} 60–64 dB	L_{den} 65–69 dB	L_{den} 70–74 dB	L_{den} ≥ 75 dB	
Gap filled dataset 2012 (Fons-Esteve et al., 2021)	3.04	1.63	0.79	0.30	0.12	183,303,187
Gap filled dataset 2017 (Fons-Esteve et al., 2021)	3.00	1.57	0.77	0.27	0.09	188,909,912
Reported data for 2022 reference dataset downloaded from Reportnet 3 on 22/09/2023 (*)	4.57	2.82	1.34	0.36	0.10	39,601,786

(*) Data includes a total of 98 agglomerations distributed in a total of 9 countries: Austria, Czechia, Germany, Denmark, Estonia, France, Ireland, Latvia, Sweden. Not all the countries are complete.

According to the Kolmogorov-Smirnov test, the distributions of the share of people exposed to the 5 dB noise band were not statistically different ($p<0.001$).

Based on the above results, the methodology described in Houthuijs et al. (2018) and the fractions at European level described in Section 2.2.3 of the same report will be used to estimate the number of people exposed below the END thresholds for rail noise inside agglomerations.

The estimated population exposed to 50-54 dB band for L_{den} and for 45-49 dB band L_{night} is calculated using the total population exposed above 55 dB L_{den} and above 50 dB L_{night}, respectively. The following formulas are applied, with the corresponding relative fraction indicated per each indicator:

$$Population_{Lden\ 50-54\ dB} = 0.71071 * Population_{Above\ 55\ dB\ Lden}$$

$$Population_{Lnight\ 45-49\ dB} = 0.77215 * Population_{Above\ 50\ dB\ Lnight}$$

Subsequently, the same procedure is followed to estimate the lower noise bands, but taking as a starting point the population exposed above 50 dB for L_{den} and above 45 dB for L_{night}, by adding to the reported population the estimation done in the previous calculation. The formulas used to estimate the exposure values to END lower levels are the ones displayed below, using the corresponding relative fraction per each noise bands as shown in Table 2.6:

$$Population_{Lden,5\ dB\ category} = Fraction_{Lden,5\ dB\ category} * Population_{Above\ 50\ dB\ Lden}$$

$$Population_{Lnight,5\ dB\ category} = Fraction_{Lnight,5dB\ category} * Population_{Above\ 45\ dB\ Lnight}$$

Table 2.6. Mean relative fractions from population above 50 dB L_{den} or above 45 dB L_{night} to estimate the population in lower 5 dB exposure categories for railway noise inside agglomerations

Exposure category	Fraction for L_{den}	Fraction for L_{night}
45 - 49 dB	0.68267	-
40 - 44 dB	1.05637	0.71071
35 – 39 dB	1.68894	1.16786
30 – 34 dB	-	1.80714
25 – 29 dB	-	2.88929

If exposure values below the END threshold have been reported for L_{den} and/or for L_{night}, the reported exposure values will be included in the corresponding noise band instead of the estimated population (see Annex 5).

2.1.3. Agglomeration air

Comparison between the percentages used to calculate exposure values below the END threshold, using the following datasets:

- Gap filled dataset (reported and gap filled) for 2012 reference year following Fons-Esteve et al. (2021)
- Gap filled dataset (reported and gap filled) for 2017 reference year following Fons-Esteve et al. (2021)
- Reported data for 2022 reference year, downloaded directly from Reportnet 3 on 26/09/2023.

The comparison of the percentage of people exposed per each L_{den} noise band in the different datasets can be seen in Table 2.7.

Table 2.7. Results of the comparison of percentages of people exposed for the different input datasets for aircraft noise inside agglomerations

Input dataset	Percentage of people exposed					Total number of inhabitants inside agglomerations
	L_{den} 55–59 dB	L_{den} 60–64 dB	L_{den} 65–69 dB	L_{den} 70–74 dB	L_{den} ≥ 75 dB	
Gap filled dataset 2012 (Fons. J. et al, 2021)	1.270	0.360	0.100	0.010	0.001	183,303,187
Gap filled dataset 2017 (Fons. J. et al, 2021)	1.460	0.390	0.080	0.008	<0.001	188,552,851
Reported data for 2022 reference dataset downloaded from Reportnet 3 on 22/09/2023	1.770	0.250	0.020	<0.001	<0.001	26,031,432*

(*) Data includes a total of 36 agglomerations distributed in a total of 8 countries: Austria, Czechia, Germany, Estonia, France, Ireland, Latvia, Sweden. Not all the countries are complete.

According to the Kolmogorov-Smirnov test, the distributions of the share of people exposed to the 5 dB noise band were not statistically different ($p<0.001$).

Based on the above results, the methodology described in Houthuijs et al. (2018) and the fractions at European level described in Section 2.2.3 of the same report will be used to estimate the number of people exposed below the END thresholds for aircraft noise inside agglomerations.

In the case of aircraft noise estimations, the exposed population used as reference to calculate the relative fractions is solely the number of inhabitants exposed from 55 to 59 dB L_{den} and from 50 to 54 dB L_{night}. This is due to the fact that restrictions in spatial planning are often in place at higher noise levels and may also differ between airports, so it was assumed that considering the total population exposed equal or above 55dB L_{den} and equal or above 50 dB L_{night} might be less representative to estimate the exposure values at lower END thresholds.

The formulas used to estimate the exposure values to END lower levels are displayed below, using the corresponding relative fraction per each noise bands as shown in Table 2.8:

$$Population_{Lden,5\ dB\ category} = Fraction_{Lden,5\ dB\ category} * Population_{55-59\ dB\ Lden}$$

$$Population_{Lnight,5\ dB\ category} = Fraction_{Lnight,5dB\ category} * Population_{50-54\ dB\ Lnight}$$

Table 2.8. Mean relative fractions from population exposed to 55-59dB L_{den} or from population exposed to 50-54 dB L_{night} to estimate the population in lower 5 dB exposure categories for aircraft noise inside agglomerations

Exposure category	Fraction for L_{den}	Fraction for L_{night}
50-54 dB	2.425775	-
45-49 dB	4.400184	2.659887
40-44 dB	6.444598	6.037403
35-39 dB	-	9.275389
30-34 dB	-	15.15587

If exposure values below the END threshold have been reported for L_{den} and/or for L_{night}, the reported exposure values will be included in the corresponding noise band instead of the estimated population (see Annex 5).

2.1.4. Agglomeration industry

Current methodology implemented to calculate exposure to lower END noise bands for agglomeration industry is described in Houthuijs et al. (2018), and the relative fractions used at European level are described in Section 2.2.6 of the same report.

Reported data for the 2022 reference year, downloaded directly from Reportnet 3 on 26/09/2023 has been used to calculate the percentage of people exposed per each L_{den} noise band.

Table 2.9. Percentages of people exposed to aircraft noise inside agglomerations (reference year 2022)

Input dataset	Percentage of people exposed					Total number of inhabitants inside agglomerations
	L_{den} 55–59 dB	L_{den} 60–64 dB	L_{den} 65–69 dB	L_{den} 70–74 dB	L_{den} ≥ 75 dB	
Reported data for 2022 reference dataset downloaded from Reportnet 3 on 22/09/2023 (*)	0.420	0.160	0.050	0.010	<0.001	34,743,478

(*) Data includes a total of 77 agglomerations distributed in a total of 8 countries: Austria, Czechia, Denmark, Germany, Estonia, France, Ireland, Latvia. Not all the countries are complete.

According to the Kolmogorov-Smirnov test, the distributions of the share of people exposed to the 5 dB noise bands obtain a result of $p=0.007$.

The results are less accurate but provided the limited amount of data when this analysis in being performed, it is considered that the relative fractions described in Houthuijs et al. (2018) can be used to estimate the number of people exposed below the END thresholds for industrial noise inside agglomerations, but a review will be needed when a complete dataset for 2022 reference year is available.

The estimated population exposed to 50-54 dB band for L_{den} and for 45-49 dB band L_{night} is calculated using the total population exposed above 55 dB L_{den} and above 50 dB L_{night}, respectively. The following formulas are applied, with the corresponding relative fraction indicated per each indicator:

$$Population_{Lden\ 50-54\ dB} = 3.191736 * Population_{Above\ 55\ dB\ Lden}$$

$$Population_{Lnight\ 45-49\ dB} = 2.217099 * Population_{Above\ 50\ dB\ Lnight}$$

Subsequently, the same procedure is followed to estimate the lower noise bands, but taking as a starting point the population exposed above 50 dB for L_{den} and above 45 dB for L_{night}, by adding to the reported population the estimation done in the previous calculation. The formulas used to estimate the exposure values to END lower levels are displayed below, using the corresponding relative fraction per each noise bands as shown in Table 2.10:

$$Population_{Lden,5\ dB\ category} = Fraction_{Lden,5\ dB\ category} * Population_{Above\ 50\ dB\ Lden}$$

$$Population_{Lnight,5\ dB\ category} = Fraction_{Lnight,5dB\ category} * Population_{Above\ 45\ dB\ Lnight}$$

Table 2.10. Mean relative fractions from population above 50 dB L_{den} or above 45 dB L_{night} to estimate the population in lower 5 dB exposure categories for industrial noise inside agglomerations

Exposure category	Fraction for L_{den}	Fraction for L_{night}
45 – 49 dB	2.207175	-
40 – 44 dB	3.468227	3.156831
35 – 39 dB	3.634048	6.505400
30 – 34 dB	-	7.570637
25 – 29 dB	-	7.314349

If exposure values below the END threshold have been reported for L_{den} and/or for L_{night}, the reported exposure values will be included in the corresponding noise band instead of the estimated population (see Annex 5).

2.1.5. Major roads

Comparison between the percentages used to calculate exposure values below the END threshold, using the following datasets:

- Gap filled dataset (reported and gap filled) for 2012 reference year following Alberts et al. (2016)
- Gap filled dataset (reported and gap filled) for 2012 reference year following Fons-Esteve et al. (2021)
- Gap filled dataset (reported and gap filled) for 2017 reference year following Fons-Esteve et al. (2021)
- Reported data for 2022 reference year, downloaded directly from Reportnet 3 on 26/09/2023.

The comparison of the percentage of people exposed per each L_{den} noise band in the different datasets can be seen in Table 2.11.

Table 2.11. Results of the comparison of percentages of people exposed for the different input datasets for major roads outside agglomerations

Input dataset	Percentage of people exposed					Total number of inhabitants outside agglomerations
	L_{den} 55–59 dB	L_{den} 60–64 dB	L_{den} 65–69 dB	L_{den} 70–74 dB	L_{den} ≥ 75 dB	
Gap filled dataset 2012 (Alberts et al., 2016)	3.62	2.34	1.45	0.71	0.18	337,000,000
Gap filled dataset 2012 (Fons-Esteve et al., 2021)	4.00	2.52	1.67	0.81	0.17	339,737,532
Gap filled dataset 2017 (Fons-Esteve et al., 2021)	3.98	2.20	1.67	0.76	0.13	344,253,152
Reported data for 2022 reference dataset downloaded from Reportnet 3 on 22/09/2023 (*)	5.34	2.96	2.07	0.95	0.13	128,637,940

(*) Data includes a total of 9 countries: Austria, Czechia, Denmark, Germany, Estonia, France, Ireland, Latvia, Sweden. There is no information in relation to completeness of exposure data reported per country.

According to the Kolmogorov-Smirnov test, the distributions of the share of people exposed to the 5 dB noise band were not statistically different ($p<0.001$).

Based on the above results, the methodology described in Houthuijs et al. (2018) and the fractions at European level described in Section 2.2.3 of the same report will be used to estimate the number of people exposed below the END thresholds for major roads outside agglomerations.

The estimated population exposed to 50-54 dB band for L_{den} and for 45-49 dB band L_{night} is calculated using the total population exposed above 55 dB L_{den} and above 50 dB L_{night}, respectively. The following formulas are applied, with the corresponding relative fraction indicated per each indicator:

$$Population_{Lden\ 50-54\ dB} = 0.71071 * Population_{Above\ 55\ dB\ Lden}$$

$$Population_{Lnight\ 45-49\ dB} = 0.77215 * Population_{Above\ 50\ dB\ Lnight}$$

Subsequently, the same procedure is followed to estimate the lower noise bands, but taking as a starting point the population exposed above 50 dB for L_{den} and above 45 dB for L_{night}, by adding to the reported population the estimation done in the previous calculation. The formulas used to estimate the exposure values to END lower levels are displayed below, using the corresponding relative fraction per each noise bands as shown in Table 2.12:

$$Population_{Lden,5\ dB\ category} = Fraction_{Lden,5\ dB\ category} * Population_{Above\ 50\ dB\ Lden}$$

$$Population_{Lnight,5\ dB\ category} = Fraction_{Lnight,5dB\ category} * Population_{Above\ 45\ dB\ Lnight}$$

Table 2.12. Mean relative fractions from population above 50 dB L_{den} or above 45 dB L_{night} to estimate the population in lower 5 dB exposure categories for railway noise inside agglomerations

Exposure category	Fraction for L_{den}	Fraction for L_{night}
45 - 49 dB	0.68267	-
40 - 44 dB	1.05637	0.71071
35 – 39 dB	1.68894	1.16786
30 – 34 dB	-	1.80714
25 – 29 dB	-	2.88929

If exposure values below the END threshold have been reported for L_{den} and/or for L_{night}, the reported exposure values will be included in the corresponding noise band instead of the estimated population (see Annex 5).

2.1.6. Major railways

Comparison between the percentages used to calculate exposure values below the END threshold, using the following datasets:

- Gap filled dataset (reported and gap filled) for 2012 reference year following Fons-Esteve et al. (2021)
- Gap filled dataset (reported and gap filled) for 2017 reference year following Fons-Esteve et al. (2021)
- Reported data for 2022 reference year, downloaded directly from Reportnet 3 on 26/09/2023.

The comparison of the percentage of people exposed per each L_{den} noise band in the different datasets can be seen in Table 2.13.

Table 2.13. Results of the comparison of percentages of people exposed for the different input datasets for major railways outside agglomerations

Input dataset	Percentage of people exposed					Total number of inhabitants outside agglomerations
	L_{den} 55–59 dB	L_{den} 60–64 dB	L_{den} 65–69 dB	L_{den} 70–74 dB	L_{den} ≥ 75 dB	
Gap filled dataset 2012 (Fons-Esteve et al., 2021)	1.31	0.67	0.32	0.17	0.09	339,737,532
Gap filled dataset 2017 (Fons-Esteve et al., 2021)	1.61	0.83	0.41	0.19	0.10	344,122,839
Reported data for 2022 reference dataset downloaded from Reportnet 3 on 22/09/2023 (*)	1.41	0.77	0.36	0.14	0.05	127,849,323

(*) Data includes a total of 8 countries: Austria, Czechia, Denmark, Germany, France, Ireland, Latvia, Sweden. There is no information in relation to completeness of exposure data reported per country.

According to the Kolmogorov-Smirnov test, the distributions of the share of people exposed to the 5 dB noise band were not statistically different ($p<0.001$).

Based on the above results, the methodology described in Houthuijs et al. (2018) and the fractions at European level described in Section 2.2.3 of the same report will be used to estimate the number of people exposed below the END thresholds for major railways outside agglomerations.

The estimated population exposed to 50-54 dB band for L_{den} and for 45-49 dB band L_{night} is calculated using the total population exposed above 55 dB L_{den} and above 50 dB L_{night}, respectively. The following formulas are applied, with the corresponding relative fraction indicated per each indicator:

$$Population_{Lden\ 50-54\ dB} = 0.71071 * Population_{Above\ 55\ dB\ Lden}$$

$$Population_{Lnight\ 45-49\ dB} = 0.77215 * Population_{Above\ 50\ dB\ Lnight}$$

Subsequently, the same procedure is followed to estimate the lower noise bands, but taking as a starting point the population exposed above 50 dB for L_{den} and above 45 dB for L_{night}, by adding to the reported population the estimation done in the previous calculation. The formulas used to estimate the exposure values to END lower levels are displayed below, using the corresponding relative fraction per each noise bands as shown in Table 2.14:

$$Population_{Lden,5\ dB\ category} = Fraction_{Lden,5\ dB\ category} * Population_{Above\ 50\ dB\ Lden}$$

$$Population_{Lnight,5\ dB\ category} = Fraction_{Lnight,5dB\ category} * Population_{Above\ 45\ dB\ Lnight}$$

Table 2.14. Mean relative fractions from population above 50 dB L_{den} or above 45 dB L_{night} to estimate the population in lower 5 dB exposure categories for railway noise inside agglomerations

Exposure category	Fraction for L_{den}	Fraction for L_{night}
45 - 49 dB	0.68267	-
40 - 44 dB	1.05637	0.71071
35 – 39 dB	1.68894	1.16786
30 – 34 dB	-	1.80714
25 – 29 dB	-	2.88929

If exposure values below the END threshold have been reported for L_{den} and/or for L_{night}, the reported exposure values will be included in the corresponding noise band instead of the estimated population (see Annex 5).

2.1.7. Major airports

Comparison between the percentages used to calculate exposure values below the END threshold, using the following datasets:

- Gap filled dataset (reported and gap filled) for 2012 reference year following Fons-Esteve et al. (2021)
- Gap filled dataset (reported and gap filled) for 2017 reference year following Fons-Esteve et al. (2021)
- Reported data for 2022 reference year, downloaded directly from Reportnet 3 on 26/09/2023.

The comparison of the percentage of people exposed per each L_{den} noise band in the different datasets can be seen in Table 2.15.

Table 2.15. Results of the comparison of percentages of people exposed for the different input datasets for major airports outside agglomerations

Input dataset	Percentage of people exposed					Total number of inhabitants outside agglomerations
	L_{den} 55–59 dB	L_{den} 60–64 dB	L_{den} 65–69 dB	L_{den} 70–74 dB	L_{den} ≥ 75 dB	
Gap filled dataset 2012 (Fons-Esteve et al., 2021)	0.250	0.060	0.010	0.001	<0.001	339,737,532
Gap filled dataset 2017 (Fons-Esteve et al., 2021)	0.120	0.036	0.007	0.002	<0.001	344,479,900
Reported data for 2022 reference dataset downloaded from Reportnet 3 on 22/09/2023 (*)	0.140	0.020	<0.001	<0.001	<0.001	28,491,461

(*) Data includes a total of 8 major airports distributed in a total of 6 countries: Austria, Czechia, Denmark, Ireland, Latvia, Sweden. Germany and France do not provide information on all major airports, so countries are considered not complete and not included in the above calculations.

According to the Kolmogorov-Smirnov test, the distributions of the share of people exposed to the 5 dB noise band show significant differences. However, provided the limited amount of data that have been used, it is proposed to apply the relative fractions described in Houthuijs et al. (2018) to estimate the number of people exposed below the END thresholds for major airports outside agglomerations, but review this proposal when a complete dataset for 2022 reference year will be available.

In the case of aircraft noise estimations, the exposed population used as reference to calculate the relative fractions is solely the number of inhabitants exposed from 55 to 59 dB L_{den} and from 50 to 54 dB L_{night}. This is due to the fact that restrictions in spatial planning are often in place at higher noise levels and may also differ between airports, so it was assumed that considering the total population exposed equal or above 55dB L_{den} and equal or above 50 dB L_{night} might be less representative to estimate the exposure values at lower END thresholds.

The formulas used to estimate the exposure values to END lower levels are displayed below, using the corresponding relative fraction per each noise bands as shown in Table 2.16:

$$Population_{Lden,5\ dB\ category} = Fraction_{Lden,5\ dB\ category} * Population_{55-59\ dB\ Lden}$$

$$Population_{Lnight,5\ dB\ category} = Fraction_{Lnight,5dB\ category} * Population_{50-54\ dB\ Lnight}$$

Table 2.16. Mean relative fractions from population exposed to 55-59dB L_{den} or from population exposed to 50-54 dB L_{night} to estimate the population in lower 5 dB exposure categories for aircraft noise inside agglomerations

Exposure category	Fraction for L_{den}	Fraction for L_{night}
50-54 dB	2.425775	-
45-49 dB	4.400184	2.659887
40-44 dB	6.444598	6.037403
35-39 dB	-	9.275389
30-34 dB	-	15.15587

If exposure values below the END threshold have been reported for L_{den} and/or for L_{night}, the reported exposure values will be included in the corresponding noise band instead of the estimated population (see Annex 5).

2.1.8. Methodology to improve the transfer process from 5 dB to 1 dB exposure distribution

For the calculation of the HRA a non-uniform distribution across noise bands at 1 dB resolution is needed to more precisely calculate the health risks associated with exposure to noise (instead of using a mid-band exposure within the 5 dB bands).

The methodology developed is based on a previous methodology proposed by Houthuijs et al. (2018) which, in turn, is based on the methodology of van den Hout et al. (2011) with minor modifications.

In brief, the distribution consists of ten numbers, N1-N10, the percentages of inhabitants exposed to <35, 35-39, 40-44, 45-49, 50-54, 55-59, 60-64, 65-69, 70-74 and ≥75 dB L_{den}, respectively. The width of the intervals is 5 dB. It is defined that the highest interval is 75-79 dB. It is also defined that the lowest interval (<35 dB) consists of two 5 dB intervals (25-29 and 30-34 dB) with 75% of the population in the 30-34 dB and 25% of the population in the 25-29 dB category.

For each 5 dB interval the mean gradient dN/dL (in % per dB) is given by:

$$\left(\frac{\mathrm{dN}}{\mathrm{dL}}\right) = \frac{1}{2} * \left[\frac{1}{5} * \left(\mathrm{N}_{\mathrm{j}+1} - \mathrm{N}_{j}\right) + \frac{1}{5} * \left(\mathrm{N}_{j} - \mathrm{N}_{\mathrm{j}-1}\right)\right]$$

Next, the distribution is refined by replacing each 5 dB interval by five 1 dB intervals:

$$N_{j,k} = \frac{1}{5} * [N_j + \left(\frac{dN}{dL}\right)_j * (k-3)\ ]$$

Where index $k = 1, 2, \dots, 5$ runs over the five 1 dB intervals in 5 dB interval j.

Negative percentages in the refined distribution are avoided by applying an upper limit to the gradients. The procedure is tailored for each of the assessments, since the number of exposure categories can vary between sources, location and noise exposure indicator.

The result, however, provides sometimes sharp changes between dB values at the frontier of consecutive 5dB ranges. This can be observed in the selected agglomerations in Figure 2.1.

This approach is improved by further interpolating the midpoints obtained. Figure 2.2 shows how the sharp changes disappear (right, points in red).

In order to avoid such sharp changes the following steps have been done

- Take the calculated value as described above for the midpoint dB of each 5dB range. For example, 52 is the midpoint of the 50-54 dB band.
- Calculate the mean gradient between midpoints.
- Next, the gradient is applied to 1 dB above the midpoint within the same 5dB range.
- The remaining population to be allocated to 1 dB within the 5 dB range is distributed according to the corresponding gradient.

As can be seen in Figure 2.2, the proposed approach provide a more smoothed distribution of the population in 1 dB.

Figure 2.1: Number of people exposed to aircraft noise inside agglomerations as a result to distribute to 5 dB intervals to 1 dB intervals as described by Houthuijs et al. (2018). Each colour corresponds to one agglomeration

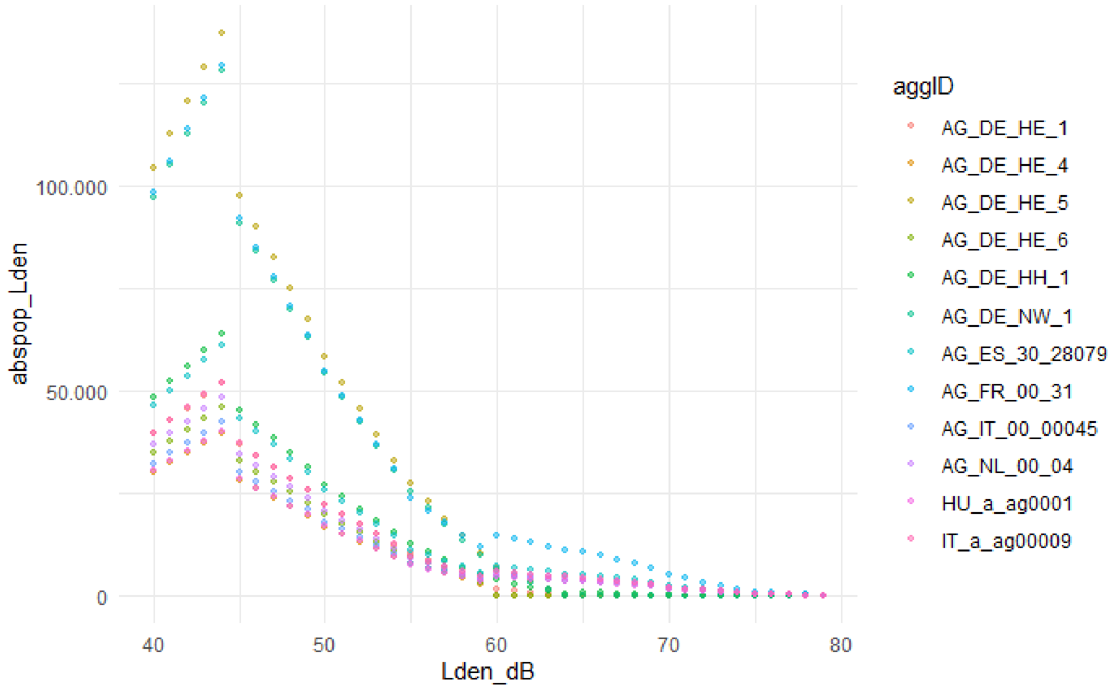
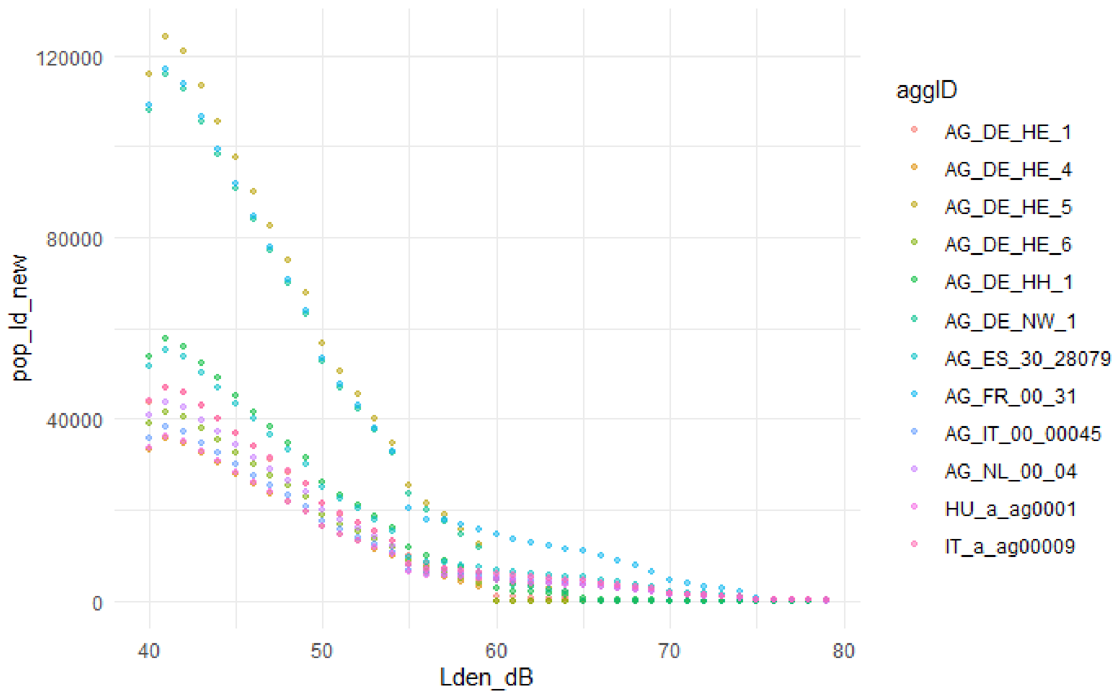


Figure 2.2: Number of people exposed to aircraft noise inside agglomerations as a result to refine the methodology described by Houthuijs et al. (2018). Each colour corresponds to one agglomeration



3. PART II: Evaluating the scientific evidence and deriving exposure-response functions

The meta-analyses commissioned by the WHO that were published as part of the WHO ENG (WHO Europe, 2018) included studies published until 2015. However, over the past years there has been a growing body of evidence on the effects of environmental noise on health. Therefore, the review in this chapter will consider how the evidence base for noise effects on health has changed based on new reviews and original studies. The evaluation will inform whether additional outcomes should be considered for the European HRA or whether exposure-response relationships should be updated.

3.1. Selection of noise sources

The environmental noise HRA at EU level is to be calculated using data collected under the Environmental Noise Directive. The END considers noise evaluated at the most exposed façade emitted by different means of transport — road traffic, rail traffic, air traffic of major infrastructures — and road, rail, aircraft and industrial activity in agglomerations of more than 100,000 inhabitants. Although there are many other sources of noise that can be harmful to health including, noise from domestic activities, neighbours, recreational venues, wind turbines, military activities and other sources, the methodology for calculating health risks due to environmental noise described in this report focusses on the health risks of noise from road traffic, rail traffic, air traffic and industrial activity.

3.2. Methods of evidence review

The HRA methodology developed by ETC/ACM in 2018 was based on the relationships and the evidence presented in the WHO Environmental Noise Guidelines for the European Region (WHO Europe, 2018). Driven by this new quantitative analysis, the EU adopted a harmonised approach to calculating the health impacts of environmental noise by updating Annex III of the Environmental Noise Directive in 2020.

3.2.1. ERFs within END Annex III

Currently, Annex III of the END considers only high levels of annoyance, high sleep disturbance, and ischaemic heart disease as relevant factors for a noise health impact assessment. However, since the publication of the WHO guidelines in 2018, the body of evidence has grown stronger regarding the health effects of noise. For instance, in the last few years, there have been a substantial number of publications linking noise to other health outcomes. In addition, higher quality studies, especially on cardiometabolic outcomes and all-cause mortality have been undertaken. Due to the publication of new high-quality evidence over the past years, the relationship between transport noise and ischaemic heart disease will be reviewed together with other emerging outcomes. Therefore, this report reviews newer evidence on the effects of noise on health in order to provide updated exposure-response relationships for the sources described in Section 3.1.

In terms of new research on annoyance, the UK's Interdepartmental Group on Costs and Benefits Noise Subject Group has reviewed studies published after the WHO ENG report, and identified 12 new studies for road and nine new studies for railway noise, published between 2014 and 2019 (Fenech et al., 2022). The group proposed to use updated aggregated ERFs for the percentage of people highly annoyed (%HA) from road and railway traffic noise by incorporating high-quality data of recent studies, published between 2014 and 2022. In general, these proposed ERFs are similar to the WHO ENG ERFs, although the proportion of %HA annoyed from road traffic is somewhat lower in the low exposure range and somewhat higher in the high exposure range. The %HA from railway traffic was observed to be slightly increased across the whole exposure range.

An update on the WHO ENG for high sleep disturbance was conducted by Smith et al. (2022) identifying eleven new studies for aircraft noise, 14 for road traffic noise and eight for railway noise, published between 2015 and 2021. The updated ERFs for the percentage of people highly sleep disturbed (%HSD) were comparable to the original ERFs from the WHO ENG for road and railway noise as well as low levels of aircraft noise, while individuals exposed to higher levels of aircraft noise might be at higher risk of sleep disturbance than previously predicted.

In conclusion, these new meta-analyses on %HA and %HSD provide very similar exposure-response functions as those adopted in Annex III of the END from the WHO guidelines (2018). The resulting changes in the HRA would be minor and thus exposure-response relationships for high annoyance and high sleep disturbance described in the WHO guidelines will not be re-evaluated in this review.

3.2.2. Scoping and Umbrella+ review to identify ERFs for additional outcomes

A scoping process, involving a literature search and expert judgement, was used to identify potentially relevant outcomes not included in the END Annex III. Next, an Umbrella+ review was conducted (e.g. used by Castro et al., 2022). In essence, an "Umbrella" review is "a review of reviews", which was used to identify the newest review of high quality. The "+" allows for the possibility to include very new, high-quality original studies in addition to the identified review.

3.2.3. Study eligibility

For this Umbrella+ review, systematic reviews and original studies in English language that were published after 2015 and provide insights into the association of at least one exposure-outcome combination were considered. Outcomes identified to be critical in the scoping process were:

- all-cause mortality,
- cardiovascular diseases (ischaemic heart disease, myocardial Infarction, stroke, hypertension, heart failure, and arrhythmia),
- mental health problems (e.g. depression, anxiety),
- cognition (e.g. reading and oral comprehension),
- behavioural problems (e.g. hyperactivity/inactivity, peer relationship problems),
- metabolic diseases including diabetes and overweight and dementia.

Noise exposures of interest were road, railway and aircraft traffic. Industry noise sources are also of interest for the noise HRA as strategic noise maps for industry inside agglomerations of more than 100,000 inhabitants are reported under the END. However, the scoping review showed that the literature on industry noise in relation to long-term health effects is very scarce. Therefore, industrial noise will only be considered in terms of high noise annoyance and was not included in the Umbrella+ review.

Study eligibility was characterized using the PECOS (Population, Exposures, Comparators, Outcomes and Study design) approach (Morgan et al., 2018). Table 3.1 shows the criteria to consider in the inclusion or exclusion of literature applying the PECOS approach.

For systematic reviews, eligible papers had to be declared systematic reviews. For original studies, we only considered high-quality studies conducted in European countries. A high-quality original study was defined as having applied reliable exposure assessment methods and accounting for most relevant confounding factors. For incident diseases like ischaemic heart disease, only cohort studies were considered to be eligible. For prevalent diseases such as hypertension, overweight, behavioural problems or cognition, case-control and cross-sectional studies were considered if they were population based, had large sample size and established methods for outcome measurements. In principle, cohort designs are considered superior to case-control and cross-sectional studies. But for prevalent disease which are partly also reversible, cross-sectional studies may be more informative for the steady state situation if people live for a long time in the same noise situation. A cohort study usually does not capture the regression of a disease if noise is reduced and may also not consider disease occurrence before the cohort was implemented.

The scoping review showed that recently several high-quality studies addressed all-cause mortality in relation to transportation noise. These studies are most reliable to assess the impact of noise on mortality given the broad systemic effects from noise (Hahad et al., 2023a, 2023b; Münzel et al., 2021). On this basis, we did not consider studies that exclusively addressed cause-specific mortality (e.g. myocardial infarction mortality) and these papers were thus excluded.

Table 3.1: Criteria for inclusion and exclusion of literature based on population, exposure, comparator, outcome and study design

PECOS	Inclusion	Exclusion
Population	General human population in Europe as well as specific, particularly vulnerable population groups	Non-human populations (in vivo, in vitro, other)
Exposure	Transportation noise exposure from road, rail, and aircraft	Occupational or leisure noise, noise annoyance
Comparator	Noise exposure (i.e., sound pressure level) as measured in decibel. Typical transportation noise levels in the range of the WHO guidelines	
Outcome	All-cause mortality, cardiovascular diseases (ischaemic heart disease, myocardial Infarction, stroke, heart failure, arrhythmia), metabolic diseases (diabetes, obesity, changes in body mass index and changes in waist circumference), mental health problems (depression, anxiety), behavioural problems, cognitive impairment (reading and oral comprehension)	Outcomes of unclear clinical health relevance, e.g., epigenetics, methylation
Study type	Key reports, systematic reviews with and without meta-analysis or major pooled analyses representative for Europe or high-quality original studies Umbrella reviews, scoping reviews, and burden of disease studies. Reviews that are published (or accepted for publication i.e., in press) between 1 January 2015 and 3 July 2023 and written in English. In case of insufficient evidence from systematic reviews, original studies from Europe of high quality not included in a review might be included.	Narrative reviews, qualitative studies, studies reporting only unadjusted results, studies with clear evidence of an analytical error, and studies using noise annoyance as a surrogate for noise exposure or no assessment of noise exposure. Intervention studies, controlled exposure studies as well as studies with focus on exposure only. Grey literature, notes, editorials, letters and unpublished data.

3.2.4. Literature search

PubMed was searched to identify peer-reviewed reviews and original studies using keywords and search terms that were defined for each element of the PECO separately. The search terms are shown in Table 7.1 in the Annex.

An initial search was performed to identify reviews including systematic reviews, meta-analyses, reviews and key reports. A second search was then conducted to identify high-quality original studies from Europe that would contribute to the evidence base. Reference lists of identified reviews were also used as a source of information. Furthermore, high-level review reports from international environmental and health agencies such as WHO, EEA and the European Environment Information and Observation Network (EIONET) were scrutinized for relevant papers. Results of the literature search and the article screening were presented in a PRISMA (Preferred Reporting Items for Systematic reviews and Meta-Analyses) flow diagram following the approach by Page et al.(2021).

3.2.5. Data extraction

Data of reviews fulfilling the eligibility criteria were extracted by one researcher (NE), subsequently checked by two other researchers (DV, MR). Extracted information includes first author, year of publication, journal, search period, considered health outcomes and noise sources, included populations, tools used to assess the quality of each study and the evidence rating for each health outcome, the number of studies identified, a summary of locations for included studies, the minimum and maximum sample size, and if a meta-analysis was conducted the pooled relative risk estimates. Further, the following information were extracted for each study that fulfilled the eligibility criteria: first author, year of publication, journal, location and cohort, follow-up period, sex, age, outcomes, number of participants, main noise source including type (modelled, measured), metric, effect estimates and increment, and adjustment for other exposures (noise and air pollution).

3.2.6. Quality Assessment Criteria for Reviews

To determine the quality of each review, a set of criteria inspired by AMSTAR 2 (Shea et al., 2017) was used with focus on three areas: literature search, risk of bias assessment, and methodology of meta-analysis. A detailed description of the criteria is shown in Table 3.2. Only reviews with high quality, i.e. those meeting all criteria, were considered for evidence rating and deriving ERFs.

Table 3.2: Criteria for quality assessment for reviews

Literature search

a) At least one relevant database (PubMed/Medline, Scopus/Embase) was considered in the literature search.

b) The author clearly and adequately defined the search terms or keywords. Thus, they are representative of the exposure as well as the health outcome considered in the review. If further inclusion criteria were defined, e.g. for the population or study design, these should also be included in the search terms.

c) The author adequately defined and explained the inclusion and exclusion criteria for all key PECOS elements, but at least for exposure and health outcome.

d) There were no critical studies missing (to our knowledge).

Risk of bias assessment

e) A risk of bias assessment was conducted using an adequate tool with adequate criteria e.g. ROBINS-E, Newcastle Ottawa Scale, or checklist of WHO.

f) If the review authors identified a risk of bias in individual studies, adequate action was taken in the review and meta-analysis. These include, for instance, conducting a sensitivity analysis, considering the risk of bias assessment in the evidence rating, or stratification of the forest plot.

Methodology of meta-analysis

g) The data extraction was adequately performed and appropriate transformation methods were used to ensure the comparability of the data between studies. Hence, the correct risk estimates were extracted, the exposure metric was coherent between studies, the same increment was used, and the conversion of categorical estimates to linear estimates was performed using a reasonable method.

h) The data pooling was done in an appropriate way, e.g. separately for noise metric or with appropriate conversions into a single noise measurement or if necessary separately for different population groups or geographical areas, and no cohort was included twice in the meta-analysis.

i) The authors used an adequate statistical method for the meta-analysis, e.g. a random effect model.

3.2.7. Evaluating the certainty of evidence for an association

Based on the compiled literature including high-quality reviews and, possibly, original study results published after the review papers, the evidence compiled for each health outcome was evaluated for an association between each source of transportation noise and physical and mental health outcomes. If a systematic evidence rating was applied in the original review, we took the corresponding evidence rating, adapting as needed to fit the terminology of the WHO ENG classification scheme: *strong, moderate, low and very low*. If such an evidence grading was missing, we applied the same criteria as WHO ENG to identify if it was *strong* or *moderate*:

- The evidence for a health effect was rated as *strong* if at least two high-quality studies were included that showed an increased risk of disease or death associated with noise and a low risk of bias.
- The evidence was classified as *moderate* if only one study of high quality has demonstrated an association.
- Studies not considered strong or moderate were excluded.

Certainty of evidence may differ for the same outcome in relation to different noise source. For some noise sources the number of studies could be very small for some outcomes. In this situation, a low certainty of evidence should not be interpreted as evidence of absence of risk. Thus, final selection of outcomes for the HRA was based on evaluation across all three noise sources for each outcome. Outcomes had to meet the following two criteria to be selected for the HRA:

- The evidence certainty for the observed association was rated to be strong or moderate for at least one transportation noise source.
- For all transportation sources together, or for at least one out of the three sources, the pooled exposure-response association had to be significantly elevated.

3.2.8. Deriving exposure-response functions

For all selected outcomes, an ERF was determined. An open question is the transferability of ERFs between different noise sources, and whether a combined ERF for transportation noise vs. individual source-specific ERFs are used. If there was an option, the latter was preferred. Thus separate meta-analyses were conducted for road traffic, railway and aircraft noise to ideally derive source specific ERFs. This is the case for %HA and %HSD, where there are numerous studies and where the source specific noise characteristics is expected to be particularly critical. However, for many other outcomes the number of studies (in particular for railway, and to a lesser extent aircraft noise) is scarce. For this reason, a pooled ERF for all transportation noise that could be applied to all three noise sources was derived.

The starting point for the meta-analysis was the WHO Environmental Noise Guidelines for the European Region (WHO Europe, 2018) or a more recent systematic review of high quality. If high-quality original studies have been published after the most recent systematic review, we incorporated the new results into the systematic review by means of a random effects meta-analysis weighted according to the inverse variance of the effect estimates. If several study results were available from the same study base, we considered the most recent and/or the most comprehensive analysis. We also gave preference to data from pooled analysis compared to single country analysis. This implied that in some cases, the effect estimate from the starting point of the meta-analysis (e.g. WHO ENG) was modified to not include studies where more recent and comprehensive updates were available. In case there were multiple studies conducted in one country it was beyond our possibility to control for potential overlap between the study populations.

ERFs were calculated as relative risk increase per 10 dB of L_{den}. If studies report noise metric other than L_{den}, we converted them to L_{den} applying the conversion factors of Brink et al. (2018). Some studies did not present relative risks per 10 dB but categorical analysis, e.g. in 5 dB exposure groups. In this case, the linear risk increase was obtained by a random effects meta-regression. The categories were weighted according to the inverse variance of the effect estimates. The weight for the reference category was estimated from the distribution of the sample size across all noise categories.

3.2.9. Effect threshold

In a HRA, the effect threshold refers to the noise exposure level below which no health effects should be quantified. Ideally, this is the level below which no health effects occur. There is no standard definition to derive this level.

In WHO ENG, the effect threshold was obtained by calculating the weighted average of the lowest noise level of the evaluated studies. For ischaemic heart disease, this level was 53 dB L_{den} in relation to road traffic and 47 dB L_{den} in relation to aircraft noise. For other outcome-exposure combinations, no lowest effect threshold was determined. This outcome of this calculation is determined by the quality of the exposure assessment. Earlier studies relied often on noise exposure models that only considered main roads and thus lowest modelled level and reference category were relatively high (e.g. 55 dB). Obviously, such studies cannot determine potential effect thresholds below this cut-off. For this reason, another approach was used to derive the effect threshold.

In an idealized form, one would assume a relative risk function that remains at the value of 1.0 until it starts to steadily increase at a certain exposure level, i.e. the lowest effect threshold. However, real data are rarely idealized and thus deviance of the relative risk from unity may also occur by chance. It is thus critical to evaluate whether the overall exposure-response pattern is indicative of threshold with subsequent consistent risk increase or not.

For all literature that we used to derive exposure-response functions, we extracted information about the lowest effect threshold, if provided, using two criteria:

1. Lowest noise exposure level where a relative risk increase in a categorical analysis or by visual inspection of a spline was above unity and remained above unity in all subsequent higher noise categories.

2. Lowest noise exposure level where a relative risk increase in a categorical analysis or by visual inspection of a spline was *significantly* above unity.

Further, it was also indicated whether any observed risk increase started at the lowest level and whether the ERF increased monotonically over the whole exposure range.

The relevant threshold was determined based on the distribution of these lowest observed effect levels. The median across all available effect thresholds was considered to be the best estimate of a true effect threshold.

3.3. Results of literature review including meta-analysis

3.3.1. All-cause mortality

Using the search terms listed in Annex 1, the literature search identified 77 potentially eligible papers. No systematic review on this topic has been identified. After reviewing title, abstract and possibly full text, 37 papers were excluded, because they did not address all-cause or all-natural cause mortality, 17 were not original research (e.g. narrative reviews, editorial, protocol papers), 14 did not provide any estimates related to noise, one was a double publication of the same cohort, and one study was not paper based. The remaining seven eligible papers (Grady et al., 2023; Sørensen et al., 2023a; Vienneau et al., 2023; Cole-Hunter et al., 2022; Hao et al., 2022b; Andersson et al., 2020; Thacher et al., 2020) plus two papers identified by manual search (Klompmaker et al., 2021; Halonen et al., 2015) are shown in Table 3.3. The PRISMA flow diagram is shown in Figure 8.1 of the Annex 2.

For the meta-analysis, Grady et al. (2023) was excluded because it is conducted outside Europe, and Halonen et al. (2015) was excluded because of the ecological study design. In total, seven studies were included in the meta-analysis. Sørensen et al. (2023a) reported separate estimates for road and railway noise (Figure 3.1). Vienneau et al. (2023) provided a relative risk for road traffic as well as for the energetic sum of all sources combined (railway, aircraft, and road traffic noise) on all-natural cause mortality. The former was included in the meta-analysis; the latter relative risk was very similar (1.044, 95%-CI: 1.039-1.048) per 10 dB after adjusting for $PM_{2.5}$, starting from 35 dB L_{den}.

Figure 3.1 shows the results of the meta-analysis. Pooled effect estimate across the five European cohort studies was 1.055 (95%-CI. 1.026-1.084) per 10 dB in road traffic noise exposure with high heterogeneity between studies (I^2=99%). For all sources combined relative risk was 1.041 (95%-CI: 1.014-1.069) per 10 dB increase in noise. For railway noise, the Danish Nurse Cohort (DNC) study did not find an association, whereas in the Dutch study (Klompmaker et al., 2021) a slightly increased risk was observed with very narrow confidence interval. Given the results from five cohort studies, the evidence certainty for an association between all-cause mortality and road traffic noise was considered to be high. For railway and aircraft noise the evidence certainty was rated to be low and very low, respectively.

Table 3.3: Characteristics of the identified original studies investigating the effect of transportation noise on all-cause mortality

Paper	Cohort [a] (Country) [b]	Cause	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c]	Comment
			N	Sex / Age	Follow-up						
Grady (2023)	NHS/NHSII (USA)	All-natural cause	117,364	Female / mean 57.3 years	1994-2014	Aircraft	Aviation Enviro Design Tool	$PM_{2.5}$	L_{dn}	Aircraft: 1.03 (0.94-1.12)	Non-European study
Sørensen (2023a)	Danish National Cohort (DK)	All-natural cause	2,600,000	Both / >50 years	2000-2017	Road Rail	Nordic Prediction Method	$PM_{2.5}$	L_{den}	Road (10-year mean): 1.091 (1.087-1.095) Rail (10-year mean): 0.997 (0.964-1.032)[$]	
Vienneau (2023)	SNC (CH)	All-natural cause	4,188,175	Both / >30 years	2000-2014	Road	SonBASE	$PM_{2.5}$	L_{den}	Road: 1.045 (1.041-1.050)	
Cole-Hunter (2022)	DNC (DK)	All-cause	22,858	Female / >44 years	1993-2014	Road	Nord2000	$PM_{2.5}$	L_{den}	Road (23-year mean): 1.08 (1.02, 1.13)	Some overlap with Sørensen (2023)
Hao (2022b)	UK Biobank	All-cause	342,566	Both / 40-69 years	2006 (+ ca. 9y)	Road	CNOSSOS-EU	-	$L_{Aeq,24h}$	Road: 1.08 (1.04-1.12)	
Klompmaker (2021)	Dutch National Cohort (NL)	All-natural cause	10,500,000	Both />30 years	2013-2018	Road	STAMINA	$PM_{2.5}$ (road only)	L_{den}	Road: 1.002 (0.999-1.006) per 7.5 dB[§] Rail: 1.004 (1.001-1.007) per 9.4 dB[§]	
Thacher (2020)	DDCH (DK)	All-natural cause	52,758	Both / 50-64 years	1993-2016	Road	SoundPLAN	$PM_{2.5}$	L_{den}	Road: 1.08 (1.05–1.11) per 10.4 dB[§]	
Andersson (2020)	PPS (SE)	All-natural cause	6,304	Male /47-55 years	1975-2011	Road	Nordic Prediction Method	NO_X	$L_{Aeq,24h}$	Road: 0.986 (0.906-1.073)[$]	
Halonen (2015)	London (UK)	All-natural cause	8,610,000	Both / ≥25 years	2003-2010	Road	TRANEX	$PM_{2.5}$	$L_{Aeq,16h}$	Road: 55-60 dB vs <55 dB: 1.03 (1.01-1.05) >60 dB vs 55 dB: 1.04 (1.00-1.07)	Small area study, separate analysis of L_{night}

[a] DDCH = Danish Diet, Cancer and Health cohort, DNC = Danish Nurse Cohort, NHS/NHSII = Nurses' Health Study, PPS = Primary Prevention Study, SNC = Swiss National Cohort

[b] CH = Switzerland, DK = Denmark, SE = Sweden, UK = United Kingdom, USA = United States of America

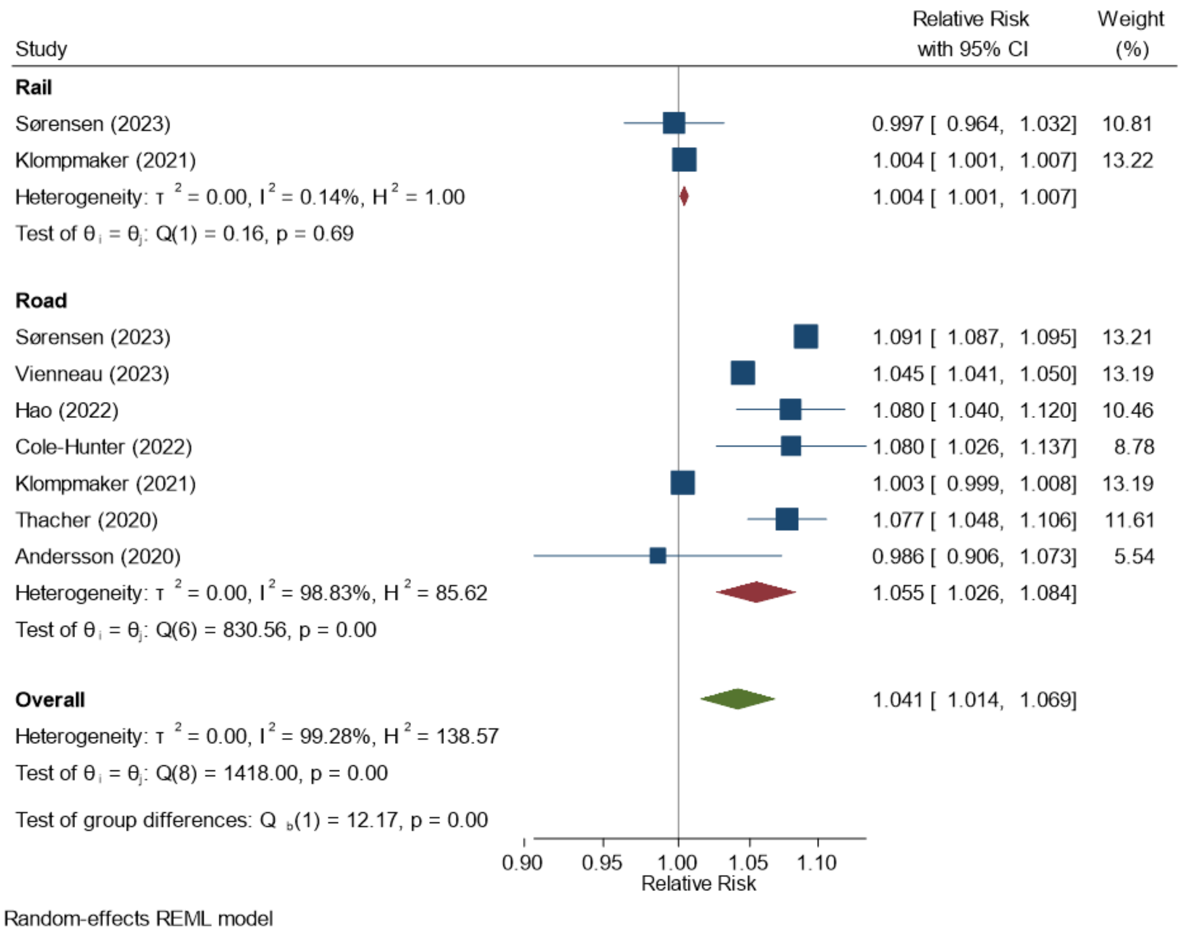
[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

[§] The relative risk has been converted to per 10 dB (based on reported effect size per increment in original study).

[$] Estimate derived from categorical results.

Abbreviations: N = Number of participants

Figure 3.1: Meta-analysis of cohort studies on all-cause mortality in relation to transportation noise, stratified by source. Relative risks refer to a 10 dB increase in L_{den}



Note: A meta-analysis of European and non-European cohort studies revealed similar estimates (Sørensen et al., 2024).

3.3.2. Cardiovascular diseases

Literature search

The literature search for cardiovascular diseases resulted in a total of 232 papers, of which 152 were excluded after scanning title and abstract. The full text of the remaining papers - 33 reviews and 48 original studies - was then reviewed. Out of the 33 reviews, 15 reviews were excluded due to being not systematic (Hahad et al., 2023a; Münzel et al., 2023; Bączalska et al., 2022; Daiber et al., 2022; Münzel et al., 2022, 2020; Argacha et al., 2019; Hahad et al., 2019a, 2019b; Halonen, 2019; Münzel et al., 2018; Sharma and Brook, 2018; Bruno et al., 2017; Münzel et al., 2017; Belojević and Paunović, 2016), three were editorials (Basner et al., 2020; Sørensen and Pershagen, 2019) or a correction of an older review (Dzhambov and Dimitrova, 2016), one only covered mortality (Cai et al., 2021), and two had literature search periods ending before or overlapping with the search period of the WHO report (Huang et al., 2015; Vienneau et al., 2015). Thus, 12 reviews (Fu et al., 2022; Hao et al., 2022b; Liu et al., 2022; Sivakumaran et al., 2022; Song et al., 2022; Zaman et al., 2022; Rabiei et al., 2021; Khosravipour and Khanlari, 2020; Weihofen et al., 2019; Van Kempen et al., 2018; Dzhambov and Dimitrova, 2018, 2017) were included for further evaluation. Of the 48 original studies six were excluded, as they only assessed mortality (Vienneau et al., 2023, 2022; Cole-Hunter et al., 2022; Thacher et al., 2020; Héritier et al., 2019, 2017) and two were published before the end of the search period of the WHO report (Bilenko et al., 2015; Hoffmann et al., 2015). Overall, 12 reviews and 40 original studies were identified for further consideration. Overviews of the original studies are presented below for each cardiovascular diagnosis separately (Table 3.6 to Table 3.10). One paper included an original analysis and a meta-analysis and was therefore counted as a review and an original study (Hao et al., 2022b). The PRISMA flow diagram is shown in Figure 8.2 of the Annex 2.

The characteristics of the 12 eligible reviews are shown in Table 3.4. Most of them addressed specific cardiovascular diagnoses and almost no studies estimated the risk for all cardiovascular diseases combined.

For all 12 reviews a quality evaluation was conducted, which is presented in Table 3.5. Based on our quality evaluation three reviews, evaluating studies published after WHO ENG, were identified to be used for the evidence rating. In one review a meta-analysis on blood pressure, hypertension and arrhythmia was conducted (Sivakumaran et al., 2022). Another eligible review focused on stroke in relation to aircraft noise (Weihofen et al., 2019) and the third review addressed blood pressure in children (Dzhambov and Dimitrova, 2017). For ischaemic heart disease and stroke no eligible review published after WHO ENG was identified and thus van Kempen et al. (2018) was selected as starting point for the meta-analysis conducted in this report meta-analysis. No review and meta-analysis was identified for heart failure and atrial fibrillation.

One review on blood pressure (Dzhambov and Dimitrova, 2018), which was of sufficient quality, was not selected, because of the newer available review on this topic (Sivakumaran et al., 2022). Two reviews focused only on myocardial infarction (MI), which is part of ischaemic heart disease (IHD), and were thus not considered. Main methodological problems of the reviews that did not pass the quality evaluation were lack of risk of bias analysis and inappropriate data extraction for meta-analysis and/or inappropriate data pooling. Data extraction was rated to be inappropriate if studies did not convert estimates to a common increment (e.g. per 10 dB change in noise exposure) but just pooled the original results as reported in the paper referring to different exposure contrasts. Inappropriate data pooling includes meta-analyses that considered several overlapping or redundant effect estimates from the same cohort without considering the clustering.

Table 3.4: Characteristics of the identified reviews investigating the effect of transportation noise on incidence of cardiovascular outcomes

Review	Outcome [a]	Noise Source [b]	End of search period	Studies				Individual study quality [d]	Evidence rating [e]
				No	N	Countries [c]	Pop		
Fu (2022)	ST	Env	Jun 2022	21	420 - 4,580,311	CH, SE, DK, NL, CA, IT, NO, UK, GR	Adults	ROBINS-E	GRADE
Song (2022)	AF	Road, Rail, Aircraft	Jan 2022	5	6,304 - 3,604,968	SE, UK, DK	Adults	NOS	-
Sivakumaran (2022)	BP, HT, HR, AR	Road, Rail, Aircraft	Dec 2021	133	15 - 502,521	IR, FI, DE, USA, RS, IT, NL, TW, CN, CH, BG, AT, SE, GR, ES, IN, UK, EG, ZA, JP, PO, KR, FR, IL, CA, SK, IN, NO	Adults, Children	RCTs, ROBINS-E	GRADE
Liu (2022)	MI	Env	Dec 2021	20	3,050 - 4,600,000	CH, SE, DE, CA, DK, LT, UK,	Adults	NOS, AHRQ	-
Hao (2022b)	CHD, ST, CVD, all-cause mortality	Road , Env	Apr 2021	16	6,304 - 8,610,000	CN, CA, DK, SE, NO, UK, CH	Adults	-	-
Zaman (2022)	IHD, MI, BP, HT, ST	Road, Rail, Aircraft, Env	2021	12	780 - 8,600,000	CA, CH, DE, DK, FR, GR, IR, NL, SE, UK	Adults, Children	-	-
Rabiei (2021)	CVDs	Env	2020	139	9 - 4,415,206	TW, USA, DE, CN, DK, IN, IR, SE, UK, NL, KR, FR, IT, ES, CA, GH, ID, PK, CH, FI, LT, others	NA	JBI	-
Khosravipour (2020)	MI	Road	Nov 2019	13	2,348 - 854,366	UK, DE, SE, LT, DK, NL	Adults	MMAT	-
Weihofen (2019)	ST	Aircraft	Aug 2017	20	420-5,523,788	USA, FR, CA, UK, CH, DE, SW, GR, IT	Adults	SIGN, CASP	-
Dzhambov (2017)	BP	Road	Jul 2016	13	115 - 1,726	SK, NL, RS, DE, IN, PK, UK, AT, USA	Children	PR	-

Review	Outcome [a]	Noise Source [b]	End of search period	Studies				Individual study quality [d]	Evidence rating [e]
				No	N	Countries [c]	Pop		
Van Kempen (2018)	HT, IHD, ST, BP (+metabolic)	Road, Rail, Aircraft	Aug 2015	113	4,721-9,619,082*	USA, FR, CA, UK, CH, DE, SW, GR, IT, others	Adults, Children	WHO	GRADE
Dzhambov (2018)	HT	Road	May 2015	9	420 - 4,415,206	DK, CA, UK, NO, SE, DE, ES, CH, GR	Adults	PR, NOS	GRADE

[a] AF = Atrial fibrillation, AR = Arrhythmia, BP = Blood pressure, CHD =Coronary heart disease, CVD = Cardiovascular disease, HF = Heart failure, HR = Heart rate, HT = Hypertension, IHD = Ischaemic heart disease, MI = Myocardial infarction, ST = Stroke, others = rarely studied cardiovascular outcomes

[b] Env = Environmental

[c] AT = Austria, AU = Australia, BG = Bulgaria, CA = Canada, CH = Switzerland, CN = China, DE = Germany, DK = Denmark, EG = Egypt, ES = Spain, FI = Finland, FR = France, GH = Ghana, GR = Greece, ID = Indonesia, IL = Israel, IN = India, IR = Iran, IT = Italy, JP = Japan, KR = South Korea, LT = Lithuania, NL = Netherlands, NO = Norway, PK = Pakistan, PO = Poland, RS = Serbia, SE = Sweden, SK = Slovakia, TW = Taiwan, UK = United Kingdom, USA = United States of America, ZA = South Africa; others = not named

[d] AHRQ = Agency for Health Care Research and Quality, Development, and Evaluation, CASP = Critical Appraisal Skills Program, JBI = Joanna Briggs Institute checklist, MMAT = Mixed Methods Appraisal Tool, NOS = Newcastle-Ottawa Scale, PR = previously used checklist, RCTs = Cochrane Risk of Bias tool for randomized controlled trials, ROBINS-E = Risk of Bias Instrument for Non-randomized Studies of Exposures, SIGN = Scottish Intercollegiate Guidelines Network, WHO = WHO checklist

[e] GRADE = Grading of Recommendations Assessment, Development and Evaluation

* Minimum and maximum number of study participants included per meta-analysis (source, outcome and study design specific)

Abbreviations: No = Number of papers (including non-cardiovascular outcomes), N = Number of participants, Pop = Populations

Table 3.5: Quality assessment of the reviews investigating the effect of transportation noise on incidence of cardiovascular outcomes

Review	Literature search				Risk of Bias		Methodology for meta-analysis			Comment	Selected for outcome
	a	b	c	D	e	f	g	h	i		
Fu (2022)	✓	✓	✓	✓	✓	✓	✗	✗	✓		
Song (2022)	✓	✓	✓	✓	✗	✗	✗	✗	✓		
Sivakumaran (2022)	✓	✓	✓	✓	✓	✓	✓	✓	✓		BP, HT, AR
Liu (2022)	✓	✓	✓	✓	✓	✗	✓	✓	✓	Only MI	
Hao (2022b)	✓	✓	✓	✓	✗	✗	✓	✗	✓		
Zaman (2022)	✓	✓	✗	✓	✗	✗	✗	✗	✗	No meta-analysis conducted	
Rabiei (2021)	✓	✓	✓	✓	✗	✗	✗	✗	✓		
Khosravipour (2020)	✓	✓	✓	✓	✓	✓	✗	✓	✓	Only MI	
Weihofen (2019)	✓	✓	✓	✓	✓	✓	✓	✓	✓	Aircraft only	ST
Van Kempen (2018)	✓	✓	✓	✓	✓	✓	✓	✓	✓	WHO ENG	IHD, ST
Dzhambov (2018)	✓	✓	✓	✓	✓	✓	✓	✓	✓	Newer review on BP available	
Dzhambov (2017)	✓	✓	✓	✓	✓	✓	✓	✓	✓	Children	

[a] Relevant data base considered
[b] Clearly and adequately defined search terms/keywords
[c] Inclusion/exclusion criteria adequately defined and explained
[d] No critical studies missed
[e] Risk of Bias conducted using adequate Tool
[f] If Risk of Bias in single studies identified, then adequate actions taken in meta-analysis
[g] Appropriateness of data extraction and transformations
[h] Data pooling done in appropriate way
[i] Adequate statistical method used

A detailed description of the criteria for the quality assessment of the reviews is provided in Table 3.2.

Ischaemic heart disease

The most recent review on the incidence of IHD, which is of good quality, was published by van Kempen et al. (2018). This meta-analysis was stratified by road, rail and aircraft noise as well as prevalence, incidence and mortality. The association between road traffic noise and IHD incidence was based on three cohort and four case-control studies and showed a relative risk of 1.08 (95%-CI: 1.01-1.15) per 10 dB L_{den} with a high certainty of evidence. The analysis of the association of aircraft noise and IHD incidence was based on two ecological studies and resulted in a relative risk of 1.09 (95%-CI: 1.04-1.15) per 10 dB L_{den}. Although significant, evidence certainty was rated as very low because of the ecological study design of the included study. For railway, no studies on IHD incidence was found, which left only four cross-sectional studies (RR: 1.18, 95%-CI: 0.82-1.68) on IHD prevalence. Evidence certainty was rated as very low.

Since the end of the search period in WHO ENG, six new studies (Pyko et al., 2023; Thacher et al., 2022a; Hao et al., 2022b; Andersson et al., 2020; Pyko et al., 2019; Carey et al., 2016) shown in Table 3.6 have been published including one pooled analysis (Pyko et al., 2023). Note that data from Andersson et al. (2020) were part of the pooled analysis. The road and railway noise analysis, but not aircraft noise, of Pyko et al. (2019) were also part of the pooled analysis.

The certainty of evidence for IHD in relation to road traffic noise was previously rated as high by the WHO ENG. The new pooled analysis from the Nordic cohorts (Pyko et al., 2023), the DNC (Thacher et al., 2022a) and UK Biobank (Hao et al., 2022b) confirmed this association, although effect estimates were somewhat lower, whereas Carey et al. (2016) did not observe any association. However, effect estimates in Carey et al. (2016) were less precise than in other papers and noise exposure was assessed on 20 m^2 grid but not at the most exposed façade, possibly introducing exposure misclassification (Vienneau et al., 2019b). In conclusion, the high evidence for a link between road traffic noise and IHD was confirmed.

For railway noise and IHD, two new papers (Pyko et al., 2023; Thacher et al., 2022a) based on cohort data have been published. Their results are inconsistent. Whereas the pooled cohort study observed a significant increased risk, the DNC reported a significant decreased risk. Since the pooled prospective cohort studies are of high quality, whereas the risk of bias from missing confounding information cannot be ruled out for the DNC cohort, the evidence certainty for an association between IHD and railway noise is rated to be moderate. For aircraft noise and IHD, two new cohort studies (Thacher et al., 2022a; Pyko et al., 2019) indicate an increased risk, although both were not significant. Thus, evidence certainty was upgraded from very low to low.

Figure 3.2 shows the result of the meta-analysis stratified by transportation noise source. For road traffic noise, relative risk to develop IHD is 1.04 (95%-CI: 1.02-1.06) per 10 dB increase in L_{den} without noticeable heterogeneity (p=0.30) between estimates. Corresponding relative risks for railway and aircraft noise is 1.00 (95%-CI: 0.93-1.06) and 1.01 (95%-CI: 0.99-1.03), respectively. The pooled exposure-response function for all transportation noise sources is 1.02 (95%-CI: 1.00-1.05) per 10 dB increase in L_{den}, although there is significant heterogeneity between the three groups (p=0.01).

Table 3.6: Characteristics of the identified original studies investigating the effect of transportation noise on ischaemic heart disease not included in WHO ENG

Paper	Cohort [a] (Country) [b]	Study type	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c]	Comment
			N	Sex / Age	Follow-up						
Pyko (2023)	DDCH, DNC, SDPP, Sixty, SNAC-K, SALT, MDC, PPS, GOT-MONICA (DK, SE)	Longitudinal (pooled analysis)	132,801	Both / mean 55.4 years	1975-2017	Road Rail	Nordic Prediction Method, Nord2000	-	L_{den}	Road: 1.03 (1.00-1.05) Rail: 1.03 (1.00-1.05)	DDCH included in review
Thacher (2022a)	Danish nationwide cohort (DK)	Longitudinal	2,538,395	Both / mean 58.5 years	2005-2017	Road Rail Aircraft	Nordic Prediction Method	-	L_{den}	Road: 1.052 (1.044-1.059) Rail: 0.964 (0.951-0.977) Aircraft: 1.008 (0.990-1.025)[$]	
Hao (2022b)	UK Biobank (UK)	Longitudinal	342,566	Both / mean 56 years	2006-2018	Road	CNOSSOS-EU	-	$L_{Aeq,, 24h}$	Road: 1.02 (0.99-1.05)	
Andersson (2020)	PPS (SE)	Longitudinal	6,304	Male / mean 58.2 years	1970-2011	Road	Nordic Prediction Method	NO_X	L_{den}	Road: 53-58 dB vs <53 dB: 0.94 (0.82-1.08) 58-63 dB vs <53 dB: 1.02 (0.88-1.18) 63+ dB vs <53 dB: 1.12 (0.97–1.29)	Cohort included in Pyko (2023)
Pyko (2019)	SDPP, Sixty, SALT, SNAC-K (SE)	Longitudinal	20,012	Both / mean 60 years	1992-2011	Road Rail Aircraft	Nordic Prediction Method	-	L_{den}	Road: 0.96 (0.90-1.03) Rail: 1.01 (0.93-1.09) Aircraft: 1.04 (0.94-1.15)	Cohort included in Pyko (2023)
Carey (2016)	CPRD (UK)	Longitudinal	211,016	Both / mean 55.4 years	2005-2011	Road	TRANEX	-	L_{night}	Road: 1.00 (0.83-1.19)[$]	

[a] CPRD = Clinical Practice Research Datalink, DDCH = Danish Diet, Cancer and Health cohort, DNC = Danish Nurse Cohort (DNC), GOT-MONICA = Gothenburg cohort of Multinational Monitoring of Trends and Determinants in cardiovascular Diseases, MDC = Malmö Diet and Cancer Study, PPS = Primary Prevention Cohort, SALT = Stockholm Screening Across the Lifespan Twin Study, SDPP = Stockholm Diabetes Prevention Program, Sixty = Stockholm 60 Years Old study, SNACK-K = Swedish National Study of Aging and Care in Kungsholmen

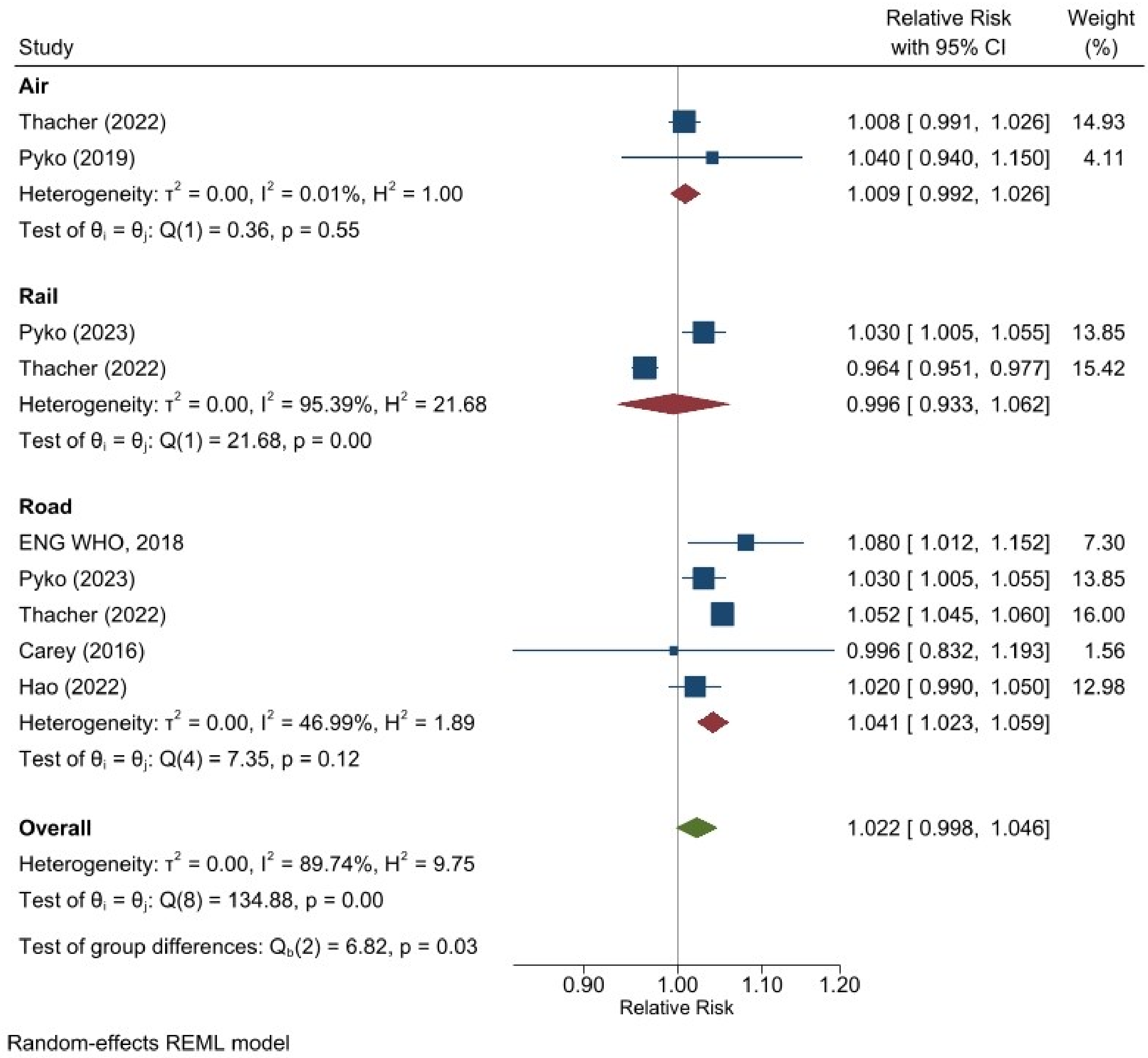
[b] DK = Denmark, SE = Sweden, UK = United Kingdom

[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

[$] Estimate derived from categorical results.

Abbreviations: N = Number of participants

Figure 3.2: Meta-analysis of the most recent systematic review on IHD (Van Kempen et al., 2018) in relation to transportation noise with subsequent cohort studies, stratified by source. Relative risks refer to a 10 dB increase in L_{den}



Blood pressure and hypertension

The most recent review investigating the association between transportation noise and blood pressure and hypertension, Sivakumaran et al. (2022), consists of multiple meta-analyses covering the association between different transportation noise sources as well as occupation noise in relation to blood pressure, hypertension and other cardiovascular outcomes that are discussed below. While most studies identified for hypertension focused on the adult population, the identified studies for blood pressure include several that examine blood pressure in children. The meta-analysis for blood pressure was thus stratified not only by noise source and study type (cross-sectional and cohort or case-control studies), but also by age, leading to one to four studies per combination. For all three noise sources - road, rail and aircraft - and the different study types, no or little effect of transport noise on systolic and diastolic blood pressure was observed. Only in preschool children was the 10 dB L_{den} associated with an increase in systolic blood pressure (MD: 4.58, 95%-CI: 3.43-5.73), while no or little effect was found in diastolic blood pressure (MD: 1.05, 95%-CI: -3.28-5.37). Evidence for changes in blood pressure related to road or railway traffic was rated by the authors to be very low.

For hypertension each noise source and study type combination included two to eight studies. The analysis for road traffic noise and cross-sectional studies indicates a risk increase of 9% (RR: 1.09, 95%-CI: 1.03-1.14) per 10 dB L_{den}, whereas the cohort and case-control studies showed no association between road traffic noise and hypertension (RR: 1.01, 95%-CI: 0.99-1.03). For aircraft noise and hypertension pooled RR for cross-sectional studies was 1.03 (95%-CI: 1.00-1.06) per 10 dB L_{den} and for cohort studies 1.10 (95%-CI: 0.95-1.27). For railway noise (RR: 0.98, 95%-CI: 0.90-1.06) no indication of an associations was found across meta-analysed cohort/case-control studies. Despite relative risks above one for road and aircraft noise, the authors rated the evidence certainty to be very low for all three sources of transportation noise.

In the review by Sivakumaran et al. (2022), the literature published up to December 2021 was considered. Since then, three studies, summarized in Table 3.7, have been published, of which two (Kim et al., 2022; Nguyen et al., 2023) were based on populations from the USA and only one cohort study (Kourieh et al., 2022) on a European population. The European study analysed the association between aircraft noise and hypertension. It showed, that a 10 dB L_{den} increase in aircraft noise levels is associated with a higher incidence of hypertension (IRR: 1.36, 95%-CI: 1.02-1.82) using a follow-up period of two to four years.

A meta-analysis (Figure 3.3) of the hypertension studies yielded non-significantly increased risk in relation to aircraft (RR: 1.08, 95%-CI: 0.98-1.20) and to road traffic noise (RR: 1.05, 95%-CI: 0.97-1.13). In contrast, the pooled estimate for transportation noise is 1.05 (95%-CI: 1.01-1.09) per 10 dB increase in L_{den} and has reached significance. Overall, with the new high-quality DEBATS study there are increasing indications for an association between transportation noise and risk of hypertension and evidence certainty was upgraded to low.

Table 3.7: Characteristics of the identified original studies investigating the effect of transportation noise on blood pressure and hypertension not included in Sivakumaran et al. (2022)

Paper	Cohort [a] (Country) [b]	Study type	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Expo-sure metric	Relative risk (95% confidence interval) [c]	Comment [d]
			N	Sex / Age	Follow-up						
Nguyen (2023)	WHI (USA)	Longitudinal	18,783	Female / mean 61.3 years	1993-2010	Aircraft	AEDT	$PM_{2.5}$	L_{dn}	Aircraft: ≥ 45 dB vs <45 dB: 1.00 (0.93-1.08)	HT Non-European study
Kourieh (2022)	DEBATS (FR)	Longitudinal	1,244	Both / mean 51 years	2013-2017	Aircraft	Maps created by Paris airports and French Civil Aviation Authority	-	L_{den}	Aircraft: 1.36 (1.02-1.82)	HT
Kim (2022)	NHS/NHSII (USA)	Longitudinal	162,183	Female / mean 59.1/40.3 years	1994-2014/ 1995-2013	Aircraft	AEDT	-	L_{dn}	Aircraft: ≥ 45 dB vs <45 dB: 1.03 (0.99-1.07) ≥ 55 dB vs <55 dB: 1.07 (0.98-1.15)	HT Non-European study

[a] DEBATS = Discussion on the health effect of aircraft noise study, NHS/NHSII = Nurses' Health Study, WHI = Women's Health Initiative

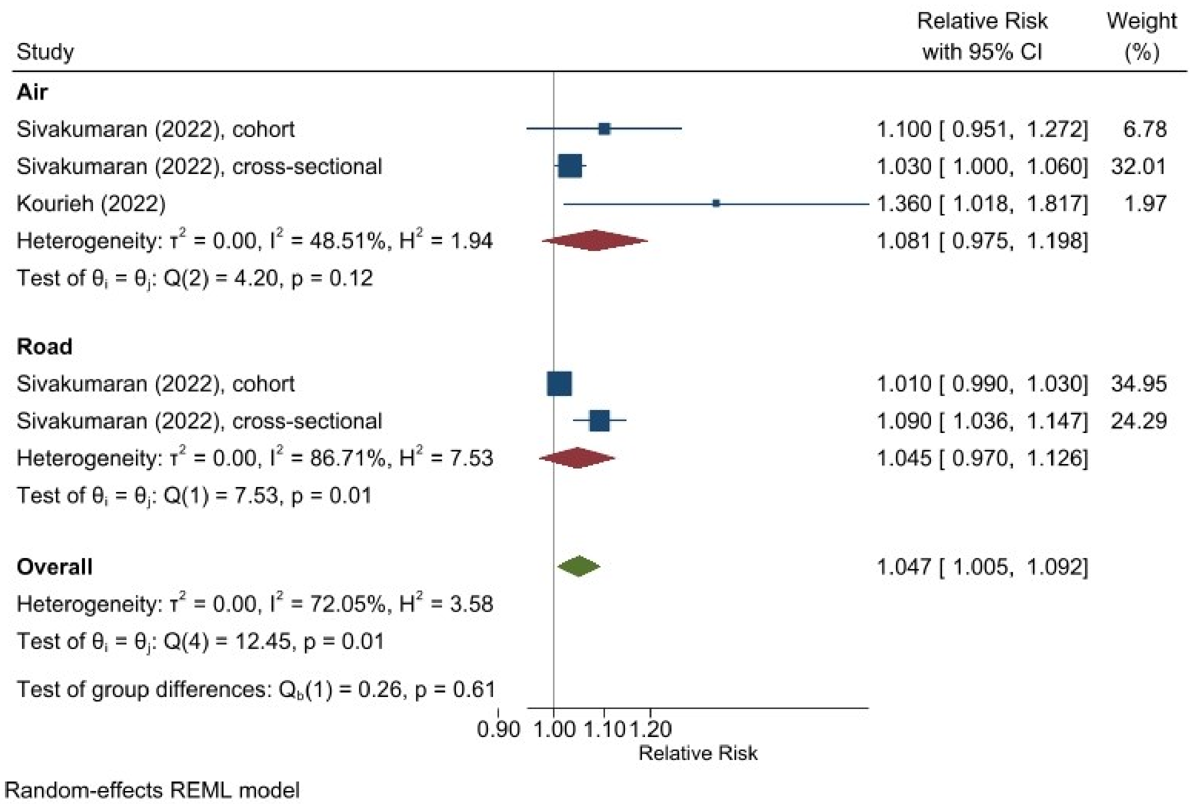
[b] FR = France, USA = United States of America

[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

[d] HT = Hypertension

Abbreviations: N = Number of participants

Figure 3.3: Meta-analysis of the most recent systematic review on hypertension (Sivakumaran et al., 2022) in relation to transportation noise a subsequent cohort study, stratified by source. Relative risks refer to a 10 dB increase in L_{den}



* "Sivakumaran (2022), cohort" includes studies with a cohort or case-control design and "Sivakumaran (2022), cross-sectional" includes only cross-sectional studies.

Arrhythmia

Sivakumaran et al. (2022) was also the most recent review covering the association between different environmental noise sources and arrhythmia. Two studies were identified, which focused on atrial fibrillation in adults. Both studies reported estimates for road traffic noise and one study for railway and aircraft noise. In the nationwide cohort study of Thacher et al. (2022b) IRR for atrial fibrillation per 10 dB increase in L_{den} were 1.000 (95%-CI: 0.995-1.005) for road and 1.012 (95%-CI: 1.003-1.022) for railway noise. In a categorical analysis for aircraft noise, IRR tended to slightly increase with increasing exposure levels resulting in an IRR of 1.055 (95%-CI: 0.996-1.116) for people exposed between 55 and 59 dB vs. <45 dB and 1.036 (95%-CI: 0.931-1.154) for those ≥60 dB. In the other study (Andersen et al., 2021), the HR was 1.02 (95%-CI: 0.95- 1.09) for female nurses per 10 dB 3-year mean L_{den}.

Sivakumaran et al. (2022) did not conduct a meta-analysis and rated the evidence to be very low for each type of transportation noise source. We have thus pooled the related effect estimates including the results from Andersson et al. (2020) and Dimakopoulou et al. (2017), which were not part of the Sivakumaran et al. (2022) review, in the meta-analysis conducted in this report, which is shown in Figure 3.4. The meta-analysis indicated a non-significant increase in risk for aircraft noise (RR: 1.21, 95%-CI: 0.7-2.08) per 10 dB L_{den} increase and a significant increase for railway and road traffic noise with an increased risk of 1.02 (95%-CI: 1.01-1.03) and 1.01 (95%-CI: 1.00-1.01) per 10 dB in L_{den}, respectively. The pooled estimate for transportation noise is 1.01 (95%-CI: 1.00-1.02) and therefore, statistically significant. The evidence certainty is rated as moderate for road traffic noise, low for railway noise and very low for aircraft noise.

Table 3.8: Characteristics of the original studies investigating the effect of transportation noise on arrhythmia

Paper	Cohort [a] (Country) [b]	Study type	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Expo-sure metric	Relative risk (95% confidence interval) [c]
			N	Sex / Age	Follow-up					
Thacher (2022b)	Danish nationwide cohort (DK)	Longitudinal	3,604,968	Both / mean 50.4 years	2000-2017	Road Rail Aircraft	Nordic Prediction Method	-	L_{den}	Road: 1.006 (1.001-1.011) Rail: 1.017 (1.007–1.026) Aircraft: 1.013 (1.00, 1.025)[$]
Andersen (2021)	DNC (DK)	Longitudinal	23,528	Female / mean 52.6 years	1993-2015	Road	Nord2000	$PM_{2.5}$	L_{den}	Road: 1.02 (0.95, 1.09)
Andersson (2020)	PPS (SE)	Longitudinal	6,304	Male / mean 58.2 years	1970-2011	Road	Nordic Prediction Method	NO_X	L_{den}	Road: 53-58 dB vs <53 dB: 0.87 (0.73-1.04) 58-63 dB vs <53 dB: 0.74 (0.60-0.91) 63+ dB vs <53 dB: 0.86 (0.68-1.07)
Dimakopoulou (2017)	HYENA (GR)	Longitudinal	420	Both / mean 58 years	2004-2013	Road Aircraft	SONDEO	-	Road: $L_{Aeq,\ 24h}$ Aircraft: L_{night}	Road: 0.96 (0.74-1.26) Aircraft: 1.88 (0.85-4.19)

[a] DNC = Danish Nurse Cohort, HYENA = Hypertension and Exposure to Noise Near Airports

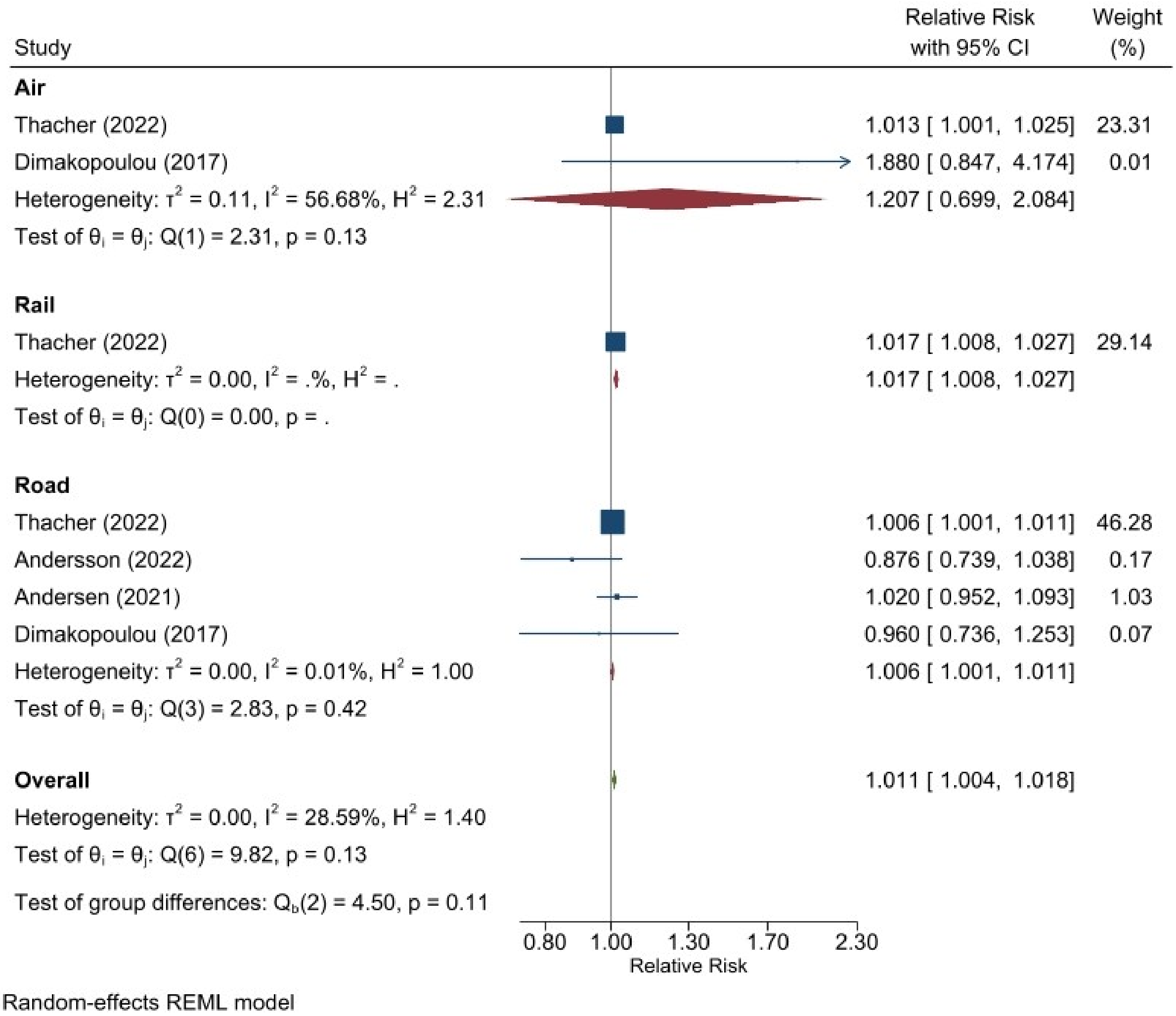
[b] DK = Denmark, GR = Greece

[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

[$] Estimate derived from categorical results.

Abbreviations: N = Number of participants

Figure 3.4: Meta-analysis of cohort studies on arrhythmia in relation to transportation noise, stratified by source. Relative risks refer to a 10 dB increase in L_{den}



Stroke

Van Kempen et al. (2018) is the most recent review on the association between stroke incidence and traffic noise. In this review the association between road and aircraft noise was investigated. For road traffic, a risk ratio of 1.14 (95%-CI: 1.03-1.25) per 10 dB L_{den} increase in noise was identified based on one cohort study and rated with a moderate certainty evidence. For railway noise only one cross-sectional study on prevalence of stroke was identified (RR=1.07, 95%-CI: 0.92-1.25) and certainty of evidence was rated to be very low (Van Kempen et al., 2018). Van Kempen et al. (2018) also evaluated the association between stroke and aircraft noise. Based on two ecological studies (Floud et al., 2013; Hansell et al., 2013) a relative risk of 1.05 (95%-CI: 0.96-1.15) per 10 dB L_{den} was found and the certainty of evidence for aircraft noise was rated as very low quality. A more recent systematic review and meta-analysis on aircraft noise (Weihofen et al., 2019) found a pooled relative risk of 1.013 (95%-CI: 0.998-1.028). In addition to the two ecological study in WHO ENG, this pooled estimate is based on another two ecological studies, two cohort studies on stroke mortality and one case-control study. The authors rated the underlying studies as being of poor to medium quality. No cohort study on stroke incidence was available for this review, which would have been the most appropriate study design. Given the limited quality of the underlying research, this pooled estimate was not considered as starting point for the meta-analysis.

Since August 2015, the end of the literature search in WHO ENG, nine new studies (Gu et al., 2023; Hao et al., 2022b; Cole-Hunter et al., 2021; Roswall et al., 2021; Sørensen et al., 2021; Andersson et al., 2020; Pyko et al., 2019; Dimakopoulou et al., 2017; Carey et al., 2016) were published (see Table 3.9). Thereof, three studies (Cole-Hunter et al., 2021; Andersson et al., 2020; Pyko et al., 2019) analysed cohorts that were also analysed by the pooled analysis by Roswall (2021) and thus not considered in the meta-analysis. Gu et al. (2023) assessed incidence of a specific stroke subtype (incident intracerebral haemorrhage) in the UK Biobank in relation to road traffic noise. All of the remaining five papers addressed the association between road traffic noise and stroke incidence, and two studies each between railway and aircraft noise and stroke incidence. These five studies include a cohort study by Sørensen et al. (2021) with a very large sample size (> 3.5 million participants) where a RR for incidence of stroke of 1.04 (95%-CI: 1.03-1.05) per 10dB increase in road traffic noise (10-year average L_{den}) and a RR of 0.97 (95%-CI: 0.96-0.99) for railway noise was observed. Further, the pooled analysis by Roswall et al. (2021) presents an increased risk per 10 dB (L_{den}) increase in road traffic noise (HR: 1.06, 95%-CI: 1.03-1.08) but not for railway noise (RR: 0.96, 95%CI: 0.91-1.01). For aircraft noise exposure (40–50 vs. ≤40 dB) RR tended to be increased (RR: 1.12; 95%-CI: 0.99-1.27), but not with higher exposure (≥50 dB) (RR: 0.94; 95%-CI: 0.79-1.11). Two other studies did not report statistical significant associations with road or railway noise.

Since WHO ENG, several high-quality cohorts reported an association between incidence of stroke and road traffic noise. Thus, previous evidence certainty from WHO ENG is upgraded from moderate to high. In terms of aircraft noise and railways noise, the new studies point mostly towards an absence of risk. Correspondingly, evidence certainty for an effect on stroke from railway and aircraft noise remains very low.

Figure 3.5 shows the meta-analysis for stroke in relation to transportation noise, stratified by the three sources road, railway and aircraft. Although we used van Kempen et al. (2018) as a starting point, we did not include their effect estimates for road traffic noise since it was based on only one cohort study (Danish Diet, Cancer and Health cohort (DDCH)), which was, as an update, also part of the new pooled analysis by Roswall et al. (2021). For road traffic noise, pooled effect estimate for stroke incidence was 1.05 (95%-CI: 1.01-1.08) per 10 dB increase in L_{den} with significant heterogeneity (p=0.04). For rail both studies found RR below unity resulting in an RR of 0.97 (95%-CI: 0.96-0.98) and for aircraft noise pooled effect estimate was close to unity (RR: 0.99, 95%-CI: 0.88-1.13). Pooled exposure-response association from all three transportation noise sources was 1.02 (95%-CI: 0.98-1.06) with significant heterogeneity between the three sources (p<0.01).

Table 3.9: Characteristics of the identified original studies investigating the effect of transportation noise on stroke not included in WHO ENG

Paper	Cohort [a] (Country) [b]	Study type	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c]	Comment
			N	Sex / Age	Follow-up						
Gu (2023)	UK Biobank (UK)	Longitudinal	402,268	Both / Mean 56.6 years	2006-2021	Road	CNOSSOS-EU	$PM_{2.5}$	L_{den}	Road: 1.16 (1.02-1.32)	Intracerebral haemorrhage
Hao (2022b)	UK Biobank (UK)	Longitudinal	342,566	Both / Mean 56 years	2006-2018	Road	CNOSSOS-EU	-	$L_{Aeq,\ 24h}$	Road: 1.07 (1.01-1.13)	
Sørensen (2021)	Danish nationwide cohort (DK)	Longitudinal	3,616,893	Both / Mean 50 years	2000-2017	Road Rail	Nordic Prediction Method	-	L_{den}	Road: 1.04 (1.03-1.05) Rail: 0.97 (0.96-0.99)	
Roswall (2021)	DDCH, DNC, SDPP, Sixty, SNAC-K, SALT, MDC, PPS, GOT-MONICA (DK, SE)	Longitudinal	135,951	Both / Mean 55.6 years	1970-2017	Road Rail Aircraft	Nordic Prediction Method, Nord2000	-	L_{den}	Road: 1.06 (1.03-1.08) Rail: 0.96 (0.91,1.01) Aircraft: 0.99 (0.86-1.13)[$]	DDCH included in WHO ENG
Cole-Hunter (2021)	DNC (DK)	Longitudinal	25,660	Both / Mean 52.9 years	1993-2014	Road	Nord2000	$PM_{2.5}$	L_{den}	Road: 1.01 (0.93-1.09)	Cohort included in Roswall (2021)
Andersson (2020)	PPS (SE)	Longitudinal	6,304	Male / Mean 58.2 years	1970–2011	Road	Nordic Prediction Method	NO_x	L_{den}	Road: 53-58 vs <53 dB: 0.98 (0.81-1.19) 58-63 vs <53 dB: 0.93 (0.75-1.15) ≥63 vs <53 dB: 1.08 (0.85-1.36)	Cohort included in Roswall (2021)
Pyko (2019)	SDPP, Sixty, SALT, SNAC-K (SE)	Longitudinal	20,012	Both / Mean 60 years	1992-2011	Road Rail Aircraft	Nordic Prediction Method	-	L_{den}	Road: 1.00 (0.92-1.09) Rail: 1.05 (0.96-1.15) Aircraft: 0.96 (0.84-1.09)	Cohort included in Roswall (2021)
Dimakopoulou (2017)	HYENA (GR)	Longitudinal	420	Both / Mean 58 years	2004-2013	Road Aircraft	SONDEO	-	Road: $L_{Aeq,\ 24h}$	Road: 1.33 (0.59-3.03)	

Paper	Cohort [a] (Country) [b]	Study type	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Expo-sure metric	Relative risk (95% confidence interval) [c]	Comment
			N	Sex / Age	Follow-up						
									Aircraft: L_{night}	Aircraft: 1.99 (0.23-17.2)	
Carey (2016)	CPRD (UK)	Longitudinal	211,016	Both / Mean 55.4years	2005-2011	Road	TRANEX	-	L_{night}	Road: 0.95 (0.88-1.02)[$]	

[a] CPRD = Clinical Practice Research Datalink, DDCH = Danish Diet, Cancer and Health cohort, DNC = Danish Nurse Cohort (DNC), GOT-MONICA = GOT-MONICA cohort, HYENA = Hypertension and Exposure to Noise Near Airports, MDC = Malmö Diet and Cancer Study, PPS = Primary Prevention Cohort, SALT = Stockholm Screening Across the Lifespan Twin Study, SDPP = Stockholm Diabetes Prevention Program, Sixty = Stockholm 60 Years Old study, SNACK-K = Swedish National Study of Aging and Care in Kungsholmen

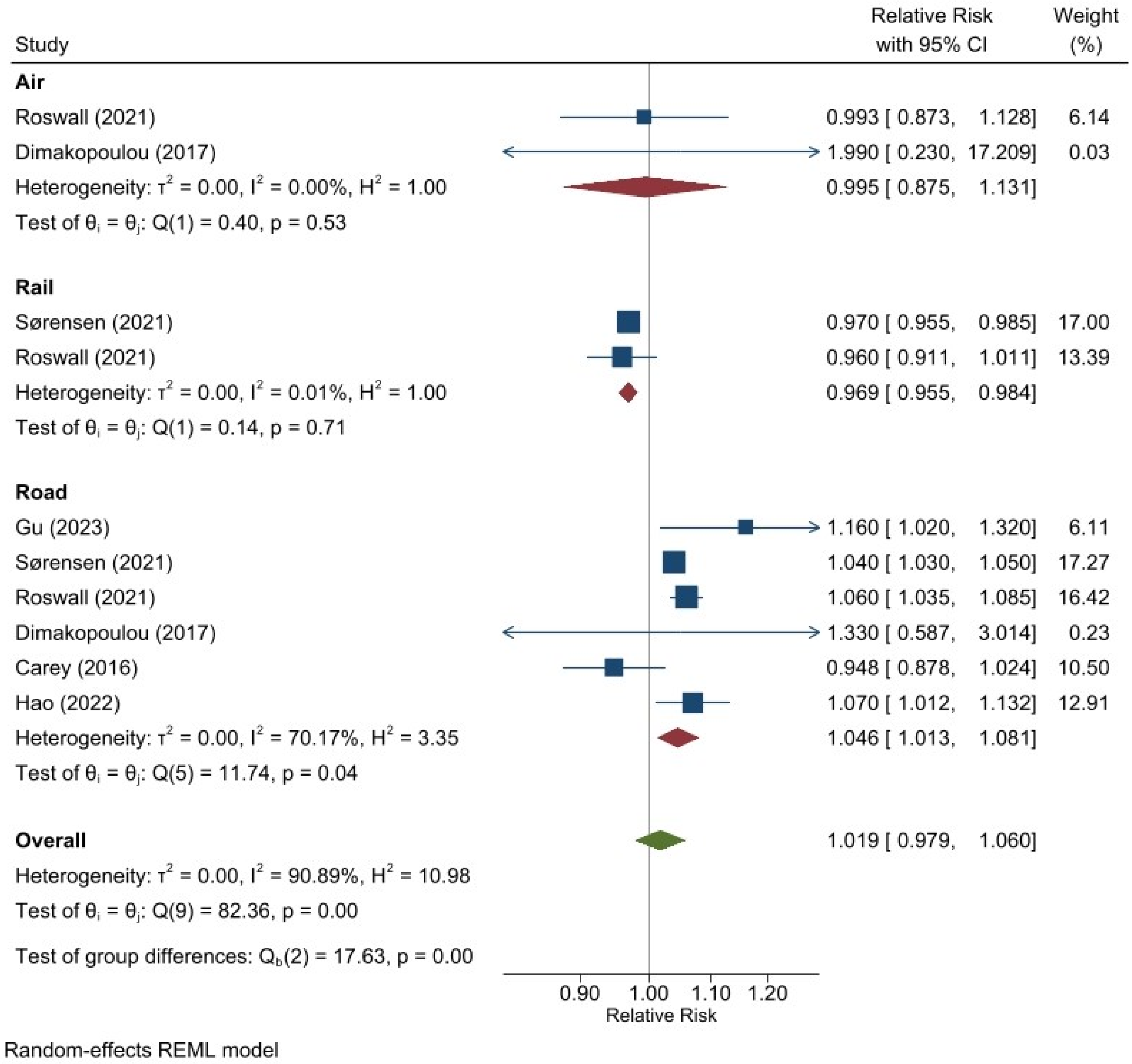
[b] FR = France, GR = Greece, SE = Sweden, UK = United Kingdom

[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

[$] Estimate derived from categorical results.

Abbreviations: N = Number of participants

Figure 3.5: Meta-analysis of cohort studies on stroke in relation to transportation noise, stratified by source. Relative risks refer to a 10 dB increase in L_{den}



Heart failure

No eligible review addressing the association between transportation noise and incidence of heart failure was identified. This association was also not addressed in the WHO ENG. Since 2015, five papers (Thacher et al., 2022a; Lim et al., 2021; Bai et al., 2020; Sørensen et al., 2017; Carey et al., 2016) have been published on this topic, as presented in Table 3.10. However, one study (Bai et al., 2020) was conducted in Canada and therefore excluded. The remaining four studies cover the association between road traffic noise and heart failure incidence and one study also includes the association with railway and aircraft noise. The three Danish studies do show an increased risk of heart failure.

Evidence certainty for heart failure and transportation noise has not been evaluated in any systematic review so far. The meta-analysis in Figure 3.6 shows for road traffic noise a RR of 1.04 (95%-CI: 1.02-1.06) with little heterogeneity across the four studies (p=0.69). This association is based on four cohort studies and thus the evidence certainty is rated to be high. For aircraft and railway only one study was available. This Danish National Cohort is very large resulting in precise effect estimates, which was significantly elevated for aircraft noise and close to unity for railways noise. Lack of lifestyle information is a limitation for this type of administrative cohort and thus the evidence certainty for the observed association with aircraft noise was rated low. For railway noise, certainty of evidence for an association is very low. For all transportation noise sources combined, RR was 1.04 (95%-CI: 1.01-1.07) with significant heterogeneity across the three groups of studies (p<0.01).

Table 3.10: Characteristics of the identified original studies investigating the effect of transportation noise on heart failure since 2015

Paper	Cohort [a] (Country) [b]	Study type	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c]	Comment
			N	Sex / Age	Follow-up						
Thacher (2022a)	Danish nationwide cohort (DK)	Longitudinal	2,538,395	Both / Mean 58.5 years	2005-2017	Road Rail Aircraft	Nordic prediction method	-	L_{den}	Road: 1.039 (1.033–1.045) Rail: 0.999 (0.988–1.011) Aircraft: 1.062 (1.040-1.085)[$]	Intracerebral haemorrhage
Lim (2021)	DNC (DK)	Longitudinal	28,189	Female / Mean 52.6 years	1993-2014	Road	Nord2000	$PM_{2.5}$	L_{den}	Road: 1.09 (0.94–1.26) per 9.3 dB[§]	
Bai (2020)	ONPHEC (CA)	Longitudinal	986,295	Both / Mean 55.6 years	2001-2015	Road	SoundPLAN	-	$L_{Aeq, 24h}$	Road: 1.07 (1.06-1.08) per 10.7dB	Non-European study
Sørensen (2017)	DDCH (DK)	Longitudinal	50,935	Both / Mean 56.2 years	1997-2011	Road	Nordic prediction method	NO_2	L_{den}	Road: 1.08 (1.00-1.16) per 9.9 dB[§]	
Carey (2016)	CPRD (UK)	Longitudinal	211,016	Both / Mean 55.4 years	2005-2011	Road	TRANEX	-	L_{night}	Road: 1.05 (0.82-1.33) [$]	

[a] CPRD = Clinical Practice Research Datalink, DDCH = Danish Diet, Cancer and Health cohort, DNC = Danish Nurse Cohort, ONPHEC = Ontario Population Health and Environment Cohort

[b] CA = Canada, DK = Denmark, UK= United Kingdom

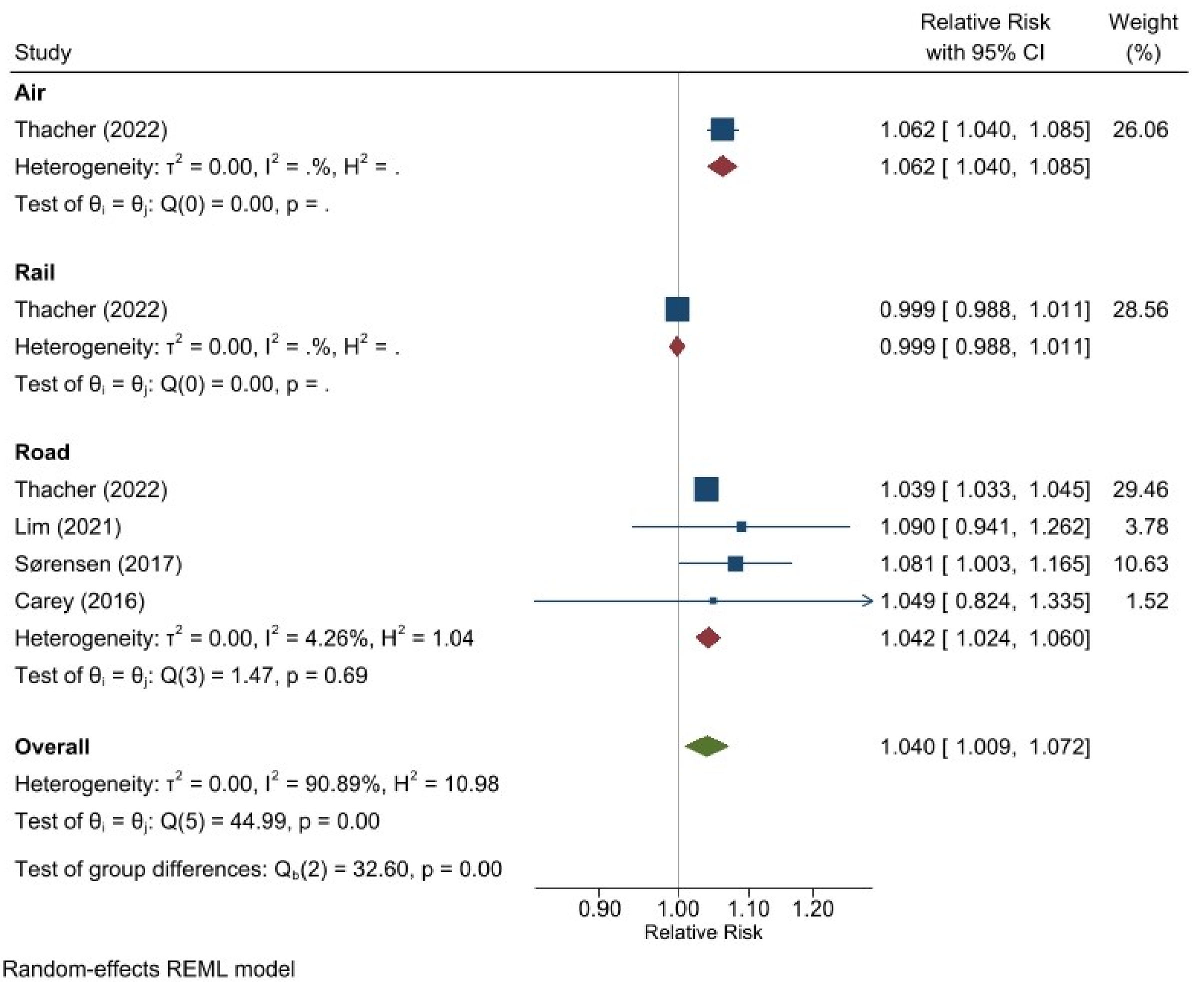
[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

[§] The relative risk has been converted to per 10 dB (based on reported effect size per increment in original study).

[$] Estimate derived from categorical results.

Abbreviations: N = Number of participants

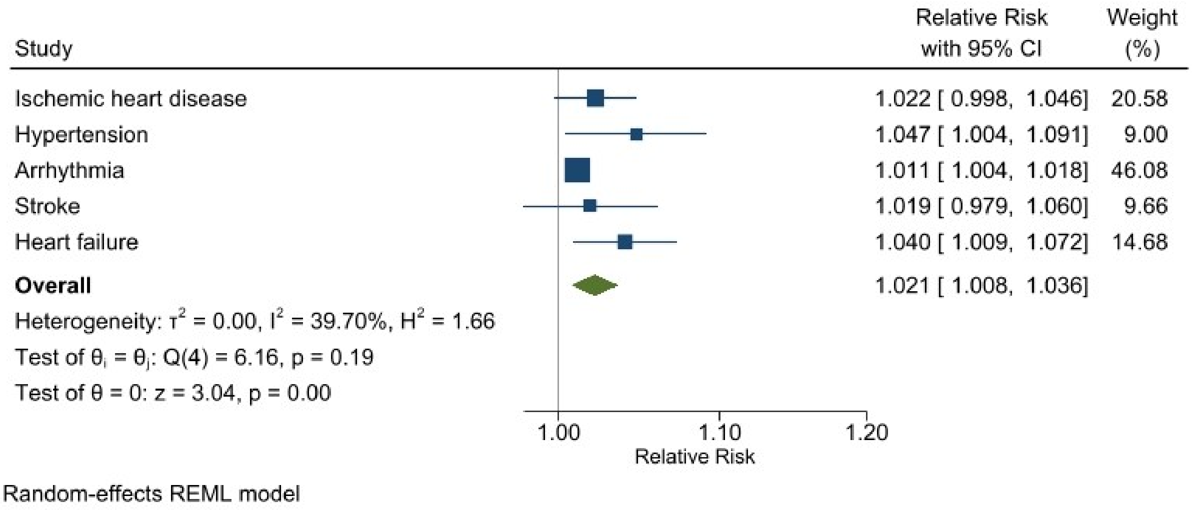
Figure 3.6: Meta-analysis of cohort studies on heart failure in relation to transportation noise, stratified by source. Relative risks refer to a 10 dB increase in L_{den}



Pooled ERF for all cardiovascular outcomes combined

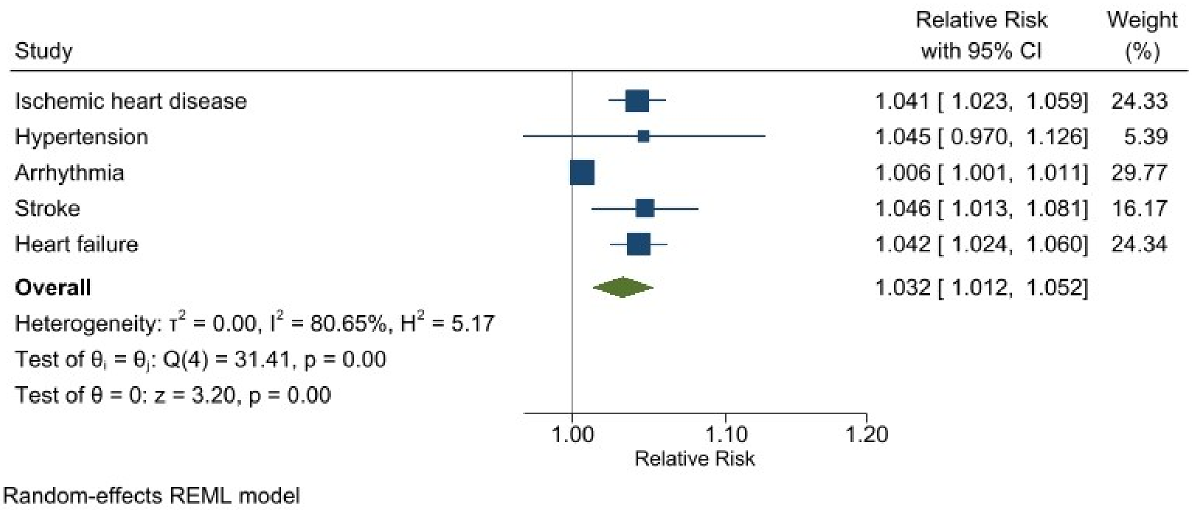
The literature review of various specific diagnostic groups of cardiovascular disease demonstrated mainly increased risk in relation to transportation noise with low to high evidence. Thus, we meta-analysed the pooled relative risks of each cardiovascular diagnostic group together to one overall relative risk. Figure 3.7 shows the overall meta-analysis for transportation noise including all three noise sources – road, rail, aircraft – and Figure 3.8 for road traffic noise only. The heterogeneity across the five diagnosis was low for transportation noise (I^2=39.70%, p=0.19) and high for road traffic noise (I^2=80.65%, p<0.01). The pooled exposure-response association from all cardiovascular outcomes for transportation noise was 1.021 (95%-CI: 1.008-1.036) per 10 dB increase in L_{den} and the pooled estimate for road traffic noise was 1.032 (95%-CI: 1.012-1.052) per 10 dB increase in L_{den}.

Figure 3.7: Meta-analysis of the overall estimates obtained for each cardiovascular outcomes in relation to all three transportation noise sources (Figure 3.2 to Figure 3.6). Relative risks refer to a 10 dB increase in L_{den}



* Confidence interval for hypertension is slightly different to the estimate shown above, due to symmetry correction of the confidence intervals during meta-analysis.

Figure 3.8: Meta-analysis of the main estimates obtained for each cardiovascular outcomes in relation to road traffic noise (Figure 3.2 to Figure 3.6). Relative risks refer to a 10 dB increase in L_{den}



Note: A meta-analysis of European and non-European studies and reviews revealed similar estimates (Münzel et al., 2024).

3.3.3. Diabetes incidence

Three systematic reviews were found for diabetes, starting with the review for the WHO END by van Kempen et al. (2018), followed by Zare Sakhvidi et al. (2018b) and a conference paper by Vienneau et al. (2019a). The latter two both included meta-analyses.

The original van Kempen et al. (2018) review found only a small number of studies on type 2 diabetes that met their inclusion criteria, and for diabetes incidence only one cohort study was available per noise source; thus, no meta-analyses were performed. The RR were 0.99 (95%-CI: 0.47-2.09) per 10 dB aircraft noise from the Stockholm study (Eriksson et al., 2014), and 1.08 (95%-CI: 1.02-1.14) per 10 dB road traffic and 0.97 (95%-CI: 0.89-1.05) per 10 dB railway noise from the DDCH cohort (Sørensen et al., 2013). Based on these individual studies, the initial certainty of evidence was rated low for aircraft and moderate for road traffic noise and railway noise in the WHO ENG review.

With the substantial increase in number of studies investigating diabetes incidence, it was deemed relevant to conduct an entirely new meta-analysis of relevant individual studies instead of abiding by the Umbrella+ protocol (that would have meant adding new studies to an existing meta-analysis). This was further justified as a way to ensure the latest and most relevant results from cohorts that had multiple publications over time were considered.

A search for new and replacement studies beyond those included in the three reviews was therefore conducted in May 2023. Duplicates were discarded, and the remaining titles and abstracts screened. Seven studies were excluded, as listed in Table 8.1 in Annex 2 along with the reason for exclusion. Also, two studies (Ohlwein et al., 2017; Sørensen et al., 2013) that were in the earlier meta-analysis were replaced with newer studies within the same cohorts (Ohlwein et al., 2019; Roswall et al., 2018).

The final selection of relevant studies are listed in Table 3.11. In total, 11 papers, with 18 effects estimates across the three noise sources were available. By source, this provided 10 estimates for road traffic noise and four each for aircraft and railway noise. With the exception of the Danish Nurses Study that only included females (Jørgensen et al., 2019), studies considered both male and female adults combined. To focus on cohort studies in the meta-analysis, one case-control study from Stockholm on aircraft noise was excluded (Eriksson et al., 2014). In the context of an European HRA, we also did not consider Canadian cohort studies (Clark et al., 2017; Shin et al., 2020). Multiple studies from Denmark, however, were allowed on the basis these covered different subsets of the population or different timeframes (Sørensen et al., 2023b; Thacher et al., 2021a; Jørgensen et al., 2019; Roswall et al., 2018). In a sensitivity analysis, only the largest Danish nationwide study was retained (Thacher et al., 2021a).

Figure 3.9 shows the meta-analysis for diabetes in relation to transportation noise, stratified by the three sources road, railway and aircraft. For road traffic noise, the pooled effect estimate for diabetes incidence was 1.06 (95%-CI: 1.04-1.09) per 10 dB increase in L_{den} with little heterogeneity (p=0.08). For rail, four studies resulted in a pooled RR of 1.02 (95%-CI: 1.00-1.03), and for aircraft noise pooled effect estimate was 1.12 (95%-CI: 0.84-1.51). Considering all sources combined in the meta-analysis, a pooled RR of 1.04 (95%-CI: 1.02, 1.06) per 10 dB transportation noise in relation to diabetes incidence was determined. In the sensitivity analysis keeping only the largest study from Denmark (to avoid double counting of individual cases), the overall pooled effect estimate considering all transportation noise sources was virtually unchanged.

The previous evidence certainty rating (Sørensen et al., 2023b) from WHO ENG for road traffic is upgraded from moderate to high, given the results from various new cohort studies pointing to an increased risk from road traffic noise. For railway and aircraft noise, relative risks from three cohorts are not significantly elevated. Thus, evidence certainty for an association is considered to be low.

Table 3.11: Characteristics of the identified original studies investigating the effect of transportation noise on diabetes incidence

Paper	Cohort [a] (Country) [b]	Study type	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c]	Comment
			N	Sex / Age	Follow-up						
Sørensen (2023b)	DNHS (DK)	Cohort	286,151	Both / mean 48.9-55.2 years depending on source	2010-2017	Road Rail	Modelled	$PM_{2.5}$	L_{den}	Road: 1.06 (1.02; 1.11)[¥] Rail: 1.02 (0.92; 1.12)[¥]	Type 2 diabetes
Zuo (2022)	UK Biobank (UK)	Cohort	305,969	Both / mean 57.1 years	2006-2010	Road	Modelled	$PM_{2.5}$	L_{den}	Road: 1.03 (1.00; 1.06)	Type 2 diabetes
Thacher (2021a)	Danish nationwide cohort (DK)	Cohort	3,563,991	Both / mean 52.7 years	2000-2017	Road Rail Aircraft	Modelled	NO_2	L_{den}	Road: 1.06 (1.05; 1.07) Rail: 1.02 (1.00; 1.03) Aircraft: 1.03 (1.01; 1.04)[$, ¥]	Type 2 diabetes
Shin (2020)	OPHEC (CA)	Cohort	914,607	Both / mean 55.3 years	2001-2015	Road	Modelled	NO_2 and UFP	$L_{Aeq, 24h}$	Road: 1.08 (1.07; 1.09)	Type not specified
Jørgensen (2019)	DNC (DK)	Cohort	23,762	Female / mean 54.0 years	1995-2012	Road	Modelled	NO_2	L_{den}	Road: 1.01 (0.91; 1.11)	Type not specified
Ohlwein (2019)	HNR (DE)	Cohort	3,396	Both / mean 58.8 years	2000-2008	Road	Modelled	NO_2	L_{den}	Road: 1.11 (0.97; 1.27)	Type 2 diabetes
Roswall (2018)	DDCH (DK)	Cohort	50,534	Both / median 56.2 years	1993-2012	Road Rail	Modelled	NO_x	L_{den}	Road: 1.12 (1.06; 1.18) Rail: 0.99 (0.94; 1.04)	Type not specified
Clark (2017)	Vancouver (CA)	Cohort	380,738	Both / mean 58 years	1999-2002	Community	Modelled	NO	L_{den}	Road: 1.03 (0.84; 1.26)	Type not specified
Dimakopoulou (2017)	HYENA (GR)	Cohort	420	Both / mean 58 years	2004-2013	Road Aircraft	Modelled	-	Road: $L_{Aeq, 24h}$ Aircraft: L_{night}	Road: 1.18 (0.85; 1.65) Aircraft: 0.92 (0.35; 2.44)	Type not specified
Eze (2017)	SALALDIA (CH)	Cohort	2,631	Both / mean 59.2 years	2002-2011	Road Rail Aircraft	Modelled	NO_2	L_{den}	Road: 1.38 (1.03; 1.83) Rail: 0.91 (0.70; 1.18)[§] Aircraft: 1.65 (0.94; 2.88)[§]	Type not specified
Eriksson (2014)	SDPP (SE)	Case-control study	5,156	Both / mean 47 years	1992-2006	Aircraft	Modelled	-	L_{den}	Aircraft: 1.03 (0.84; 1.26)[§]	Type 2 diabetes

[a] DDCH = Danish Diet, Cancer and Health cohort; DNC = Danish Nurse Cohort; DNHS = Danish National Health Survey; HNR = Heinz Nixdorf Recall; HYENA = Hypertension and Exposure to Noise near Airports; OPHEC = Ontario Population Health and Environment Cohort; SAPALDIA = Swiss Cohort Study on Air Pollution and Lung and Heart Diseases in Adults; SDPP = Stockholm Diabetes Prevention Program

[b] CA = Canada, CH = Switzerland, DE = Germany, DK = Denmark, GR = Greece, SE = Sweden, UK = United Kingdom

[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

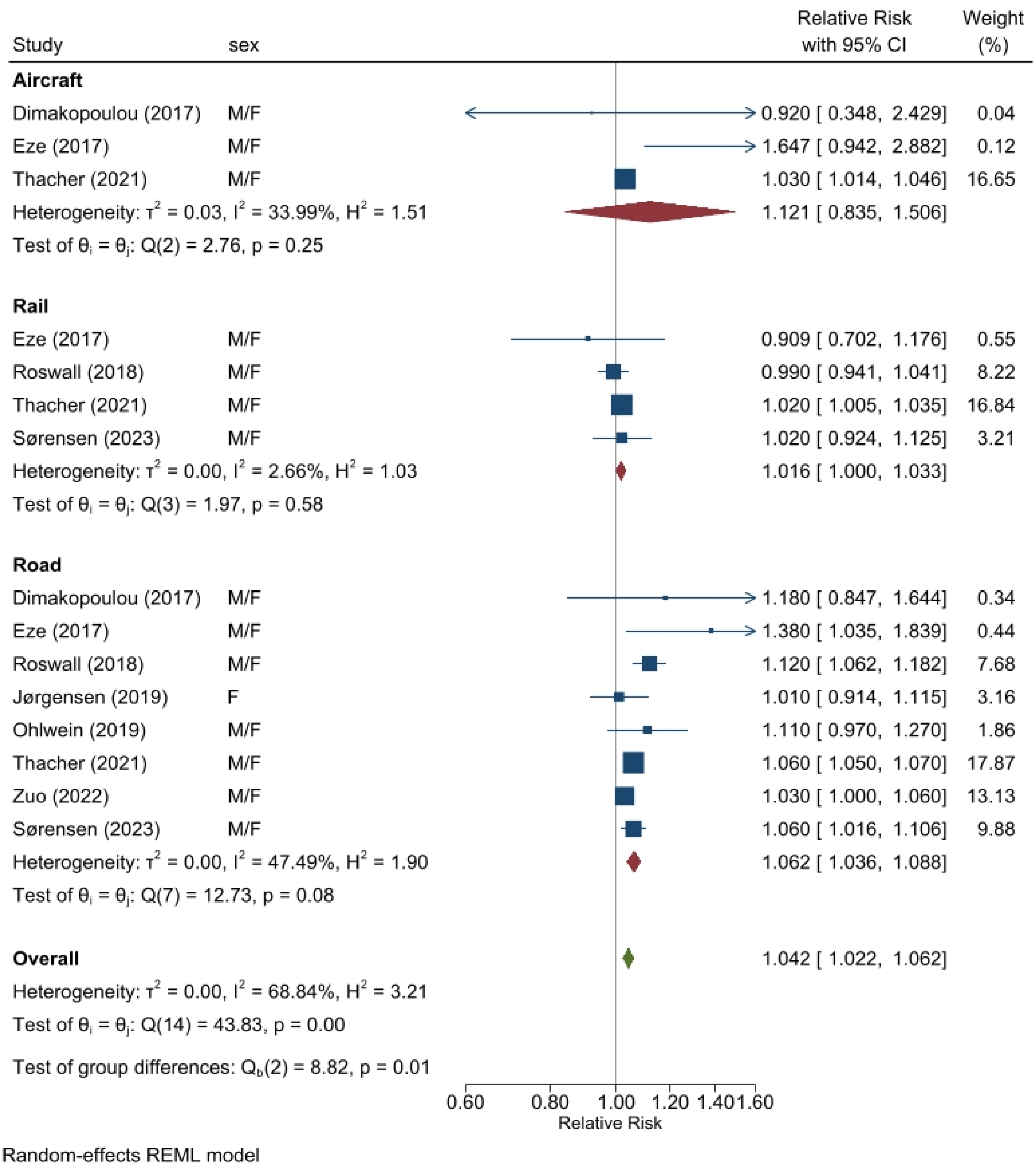
[§] The relative risk has been converted to per 10 dB (based on reported effect size per increment in original study).

[$] Estimate derived from categorical results.

[¥] Minimum exposure was taken because of importance of exposure during sleep.

Abbreviations: N = Number of participants

Figure 3.9: Meta-analysis of cohort studies on diabetes in relation to transportation noise, stratified by source. Relative risks refer to a 10 dB increase in L_{den}



Note: A meta-analysis of European and non-European studies on diabetes incidence and mortality revealed similar estimates (Vienneau et al., 2024).

3.3.4. Mental health

After identifying 212 records through the literature search, 188 papers were excluded by screening the titles and abstracts. During the full-text evaluation one review was excluded due to not being systematic (Hahad et al., 2019a), one was published before the WHO ENG report (Tzivian et al., 2015) and two original studies analysed only suicide (Wicki et al., 2023; Min and Min, 2018) thus also excluded. Therefore, eight (Tortorella et al., 2022; Zaman et al., 2022; Clark et al., 2020; Dickerson et al., 2020; Hegewald et al., 2020; Dzhambov and Lercher, 2019; Clark and Paunovic, 2018; Zare Sakhvidi et al., 2018a) reviews and 12 studies (Lin et al., 2023; Hao et al., 2022a; Leijssen et al., 2019; Klompmaker et al., 2019; Wright et al., 2018; Baudin et al., 2018; Bloemsma et al., 2022; Eze et al., 2020; Cerletti et al., 2020; Generaal et al., 2019; Orban et al., 2016; Roswall et al., 2015) were selected for further consideration. The PRISMA flow diagram is shown in Figure 8.4 of Annex 2.

The characteristics of the eight reviews and their quality assessment are shown in Table 3.12 and Table 3.13, respectively. All reviews analysed depression and anxiety and two reviews additionally considered mental health. Further, most reviews investigated all three transportation noise sources and adults as the population.

In the WHO ENG, the certainty of evidence for the association of road and aircraft noise with depression, anxiety and the use of medication against depression was rated as very low. There were no studies on railway noise. No exposure-effect relationships were calculated. A subsequent literature review, which evaluated studies published since the publication ENG WHO (mid-2015) until March 2019, concluded that the certainty of evidence for studies using interview-based depression assessment has increased somewhat to low instead of very low for all three types of traffic noise (Clark et al., 2020).

The most recent systematic review on mental health and transportation noise including a meta-analysis is from Hegewald et al. (2020). This meta-analysis included 26 studies on depression and anxiety. In five studies on aircraft noise, the RR of depression was 1.14 (95%-CI: 1.12-1.15) per 10 dB increase in noise. For road traffic noise (eleven studies) and railway noise (three studies), the risk was non-significantly increased by 2-3%. The results were strongly influenced by the German case-control study of the NORAH project, which included 77,295 patients with depression and 578,246 controls. In this study the relative risk for prescription of medication for depression was 1.17 (95%-CI: 1.10-1.25) for people with road noise ≥70 dB (L_{den}) compared to those with little exposure. Evidence for an association with aircraft and railway noise was also found in this study. Also part of the meta-analysis was a smaller prospective cohort study from Germany with 302 new cases during the study period of about 5 years. The incidence of these depressions was significantly increased (RR: 1.29, 95%-CI: 1.03-1.62) for persons with a road noise exposure of >55 dB L_{den} compared to lower exposed persons.

The characteristics of the five original studies that were published after the end of the search period of Hegewald et al. (2020) are shown in Table 3.14. However, one study (Lin et al., 2023) was conducted in Taiwan and therefore excluded. Two of the remaining studies investigated mental health as one of the health outcomes (Bloemsma et al., 2022; Cerletti et al., 2020) and did not find statistically significant association between mental health and transportation noise. Hao et al. (2022a) included symptoms of ‘nerves, anxiety, tension, or depression’, Bipolar disorder, and depressive symptoms in their analysis. They found significant associations of higher noise levels with NATD and bipolar disorder, but not with depressive symptoms. Since mental health includes multiple health outcomes, this review focuses on incidence of depression. Therefore, the only study published after the end of the search period applied in Hegewald et al. (2020) that will be included in the meta-analysis is Eze et al. (2020). The Swiss SAPALDIA study (Eze et al., 2020) included 4,581 persons who did not suffer from depression in 2001/2002 and were followed until 2010/2011. Taking into account a large number of co-factors, and using noise exposure at the place of residence the risk of disease increased non-significantly with an IRR of 1.07 (95%-CI: 0.93-1.22) per 10 dB increase in road traffic noise L_{den} and a IRR of 1.20 (95%-CI: 0.92-1.55) per 10 dB increase in aircraft noise. For study participants without a change of residence during the study period, the association with aircraft noise was statistically significant. No association was observed for railway noise.

Figure 3.10 shows pooled estimates of all available evidence on depression. For road traffic noise RR was 1.03 (95%-CI: 1.00-1.07), for railways noise 0.96 (95%-CI: 0.84-1.11) and for aircraft noise 1.14 (95%-CI: 1.13-1.16) per 10 dB increase in L_{den}. The combined estimate for transportation noise is 1.05 (95%-CI: 0.98-1.12) per 10 dB increase in L_{den}. The evidence certainty was rated to be moderate for aircraft noise, since at least one prospective cohort study found an association. For road traffic noise the certainty of evidence for an association was rated to be low and for railway noise to be very low, given the absence of association.

Table 3.12: Characteristics of the identified reviews investigating the effect of transportation noise on mental health

Review	Outcome	Noise Source [a]	End of search period	Studies				Individual study quality [c]	Evidence rating [d]
				No	N	Countries [b]	Pop		
Tortorella (2022)	Depression, Anxiety	Road, Rail, Aircraft	Dec 2021	19	339 - 3,218,500	ES, CH, NL, DK, CA, DE, CN, FI, TR	Adults	-	-
Zaman (2022)	Depression, Anxiety	Road, Rail, Aircraft	2021	6	1,244 - 23,293	NL, FI, FR, DE	Adults	-	-
Hegewald (2020)	Depression, Anxiety	Road, Rail, Aircraft	Dec 2019	31	82 - 1,026,670	FR, UK, NL, SE, IT, DE, GR, CA, FI, JP, NO	Adults	SIGN, CASP	-
Dzhambov (2019)	Depression, Anxiety	Road	Aug 2019	20	1,477 - 655,524	GR, UK, NL, SE, IT, DE, FI, CA	Adults	PR	GRADE
Clark (2020)	Depression, Anxiety, Mental health	Road, Rail, Aircraft	Mar 2019	12	399 - 387,195	BG, CA, FI, DE, IE, NL, KR	Adults, Children	Own measure	GRADE
Dickerson (2020)	Depression, Anxiety	Road, Rail, Aircraft, Env	Dec 2018	22	48 - 77,295	DE, UK, RS, IT, FI, NO, JP, KR, IN, IR, CA, USA	Adults	-	-
Zare Sakhvidi (2018a)	Anxiety, Mental health	Road, Rail, Env	Mar 2018	3	399 - 1,403	MK, BG, AT	Children	PR	GRADE
Clark (2018)	Depression, Anxiety	Road, Aircraft	Oct 2015	9	323 - 190,617	NA (Europe)	Adults	WHO	GRADE

[a] Env = Environmental

[b] AT = Austria, BG = Bulgaria, CA = Canada, CH = Switzerland, CN = China, DE = Germany, DK = Denmark, ES = Spain, FI = Finland, FR = France, GR = Greece, IE = Ireland, IN = India, IR = Iran, IT = Italy, JP = Japan, KR = South Korea, MK = Macedonia, NL = Netherlands, NO = Norway, RS = Serbia, SE = Sweden, TR = Turkey, UK = United Kingdom, USA = United States of America

[c] CASP = Critical Appraisal Skills Program, PR = previously used checklist, SIGN = Scottish Intercollegiate Guidelines Network, WHO = WHO checklist

[d] GRADE = Grading of Recommendations Assessment, Development and Evaluation

Abbreviations: No = Number of papers, N = Number of participants, Pop = Populations

Table 3.13: Quality assessment of the reviews investigating the effect of transportation noise on mental health

Review	Literature search				Risk of Bias		Methodology for Meta-analysis			Comment	Selected for outcome
	a	b	c	d	e	f	g	h	i		
Tortorella (2022)	✓	✗	✓	✓	✗	✗	✗	✗	✗	No meta-analysis conducted	
Zaman (2022)	✓	✓	✗	✓	✗	✗	✗	✗	✗	No meta-analysis conducted	
Hegewald (2020)	✓	✓	✓	✓	✓	✓	✓	✓	✓		Depression
Dzhambov (2019)	✓	✓	✓	✓	✓	✓	✓	✓	✓	Newer review on depression available	
Clark (2020)	✓	✓	✓	✓	✓	✓	✗	✗	✗	No meta-analysis conducted	
Dickerson (2020)	✓	✓	✓	✓	✗	✗	✗	✗	✗	No meta-analysis conducted	
Zare Sakhvidi (2018a)	✓	✓	✓	✓	✓	✓	✗	✗	✗	No meta-analysis conducted	
Clark (2018)	✓	✓	✓	✓	✓	✓	✗	✗	✗	No meta-analysis conducted	

[a] Relevant data base considered
[b] Clearly and adequately defined search terms/keywords
[c] Inclusion/exclusion criteria adequately defined and explained
[d] No critical studies missed
[e] Risk of Bias conducted using adequate Tool
[f] If Risk of Bias in single studies identified, then adequate actions taken in meta-analysis
[g] Appropriateness of data extraction and transformations
[h] Data pooling done in appropriate way
[i] Adequate statistical method used

A detailed description of the criteria for the quality assessment of the reviews is provided in Table 3.2.

Table 3.14: Characteristics of the identified original studies investigating the effects of transportation noise on mental health

Paper	Cohort [a] (Country) [b]	Study type	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c]	Comment
			N	Sex / Age	Follow-up						
Lin (2023)	Taiwan Biobank (TW)	Cross-sectional	3,191	Both / mean 48 years	-	Road	LUR model	$PM_{2.5}$	$L_{eq,24h}$	Depression: Road:1.62 (1.03, 2.55) per 4.7 dB	Non-European study
Bloemsma (2022)	PIAMA (NL)	Cross - sectional	3,059	Both / 11, 14, 17, 20 years	-	Road Rail	STAMINA	-	L_{den}	Poor mental wellbeing: Road: 1.04 (0.94, 1.17) per 7.4 dB Rail: 1.03 (0.93, 1.16) per 9.2 dB	Five-item Mental Health Inventory
Hao (2022a)	UK Biobank (UK)	Cross - sectional	334,986, 90,706	Both / mean 56.1 years	-	Road	CNOSSOS-EU	-	L_{den}	Depression symptoms: 52.1-54.9 vs. <52.1 dB: 0.95 (0.90, 1.00) 54.9–57.8 vs. <52.1 dB: 1.01 (0.96, 1.07) ≥57.8 vs. <52.1 dB: 1.00 (0.94, 1.06)	Additionally: symptoms of 'nerves, anxiety, tension, or depression', Bipolar disorder
Eze (2020)	SAPALDIA (CH)	Longitudinal	4,581	Both / 29-73 years	2001-2011	Road Rail Aircraft	SonBase	-	L_{den}	Depression: Road: 1.07 (0.93, 1.22) Rail: 0.88 (0.76, 1.03) Aircraft: 1.20 (0.92, 1.55)	
Cerlettia (2020)	SAPALDIA (CH)	Cross-sectional	2,035	Both / mean 57 years	-	Road Rail Aircraft	sonROAD, FLULA2	NO_2	L_{den}	Mental health: Road: 0.13 (−0.70; 0.97)* Rail: −0.76 (−1.88; 0.36)* Aircraft: 0.89 (−0.71; 2.50)*	36-Item Short Form Health Survey used

[a] PIAMA = Prevention and Incidence of Asthma and Mite Allergy, SAPALDIA = Swiss Cohort Study on Air Pollution and Lung and Heart Diseases in Adults

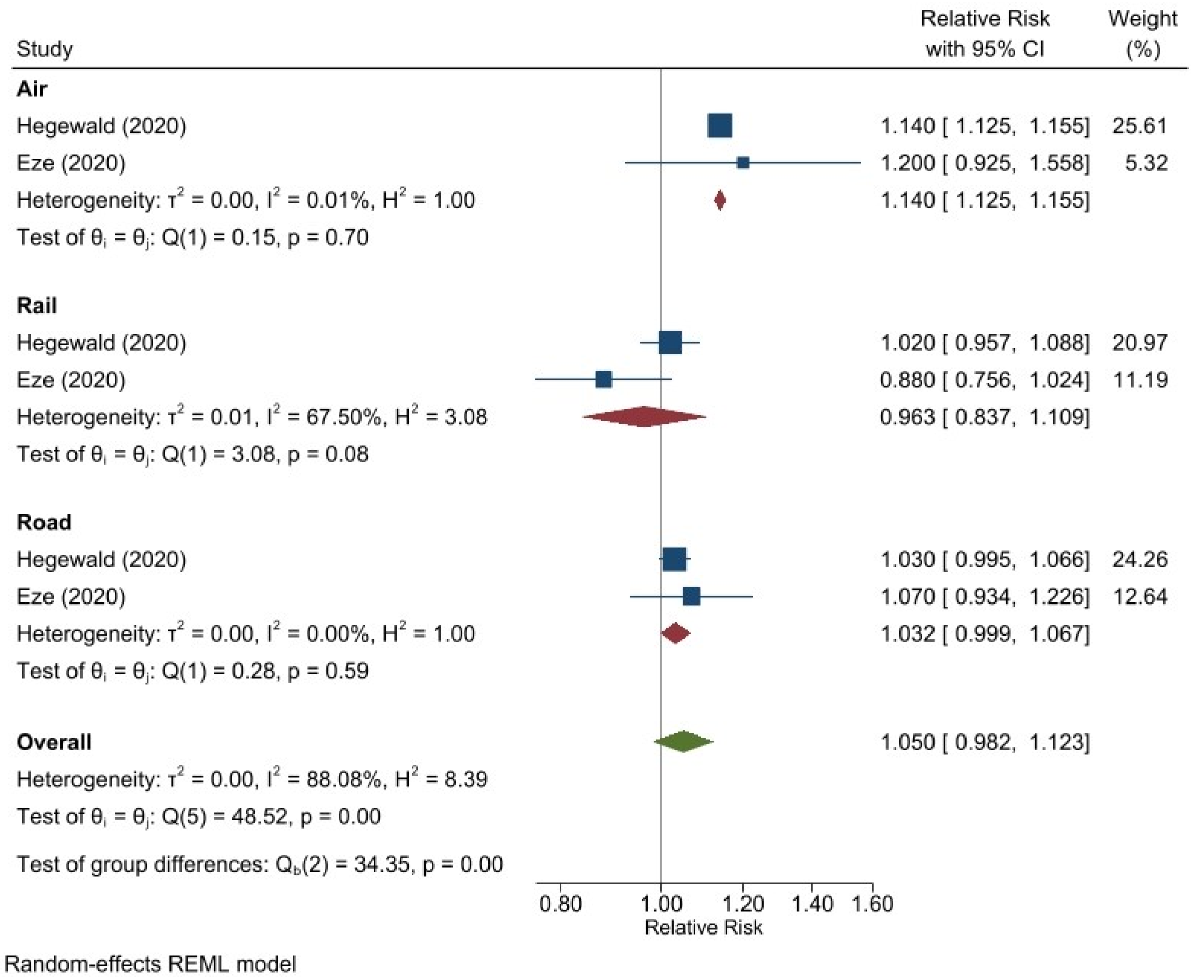
[b] CH = Switzerland, NL = Netherlands, UK = United Kingdom, TW = Taiwan

[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

* Original results reported as beta.

Abbreviations: N = Number of participants

Figure 3.10: Meta-analysis of the most recent systematic review on depression (Hegewald et al., 2020) in relation to transportation noise with subsequent cohort studies, stratified by source. Relative risks refer to a 10 dB increase in L_{den}



3.3.5. Cognition

The literature search resulted in 177 records, of which 156 were excluded by screening the title and abstract. The full-text review of the remaining 12 reviews and nine original papers, led to the additional exclusion of two reviews (Paul et al., 2019; Basner et al., 2017) that did not systematically investigate the association between cognition and transportation noise and three reviews (Stansfeld, 2015; Stansfeld and Clark, 2015; Tzivian et al., 2015) that were published before the end of the search of the WHO ENG. Further, one original study (Seabi et al., 2015) was published before the WHO ENG and thus excluded. Therefore, seven reviews (Dohmen et al., 2022; Terzakis et al., 2022; Thompson et al., 2022; Zaman et al., 2022; Zhao et al., 2021; Clark et al., 2020; Clark and Paunovic, 2018) and eight studies (Tangermann et al., 2023; Foraster et al., 2022; Raess et al., 2022; Julvez et al., 2021; Mac Domhnaill et al., 2021; Fuks et al., 2019; Tzivian et al., 2016a, 2016b) were selected. The PRISMA flow diagram is shown in Figure 8.5 of Annex 2. The characteristics of the reviews and their quality assessment are shown in Table 3.15 and Table 3.16, respectively.

In the WHO ENG a range of cognitive outcomes were evaluated. Since most of the studies were of cross-sectional design, the certainty of evidence across outcomes rated in the WHO ENG ranged from being of moderate quality (aircraft noise effects on reading comprehension, standardized assessment test and long-term memory or railway noise on standardized assessment), to low and very low for all other outcomes such as attention and executive function and other sources (road traffic noise).

The most recent review on human cognition (Thompson et al., 2022), which is also the only review including a meta-analysis, evaluated 16 new studies together with 32 studies previously reviewed by WHO ENG (Clark and Paunovic, 2018). A meta-analysis from three studies found that reading comprehension scores in quiet classrooms were 0.80 (95%-CI: 0.40-1.20) points higher than children in noisier classrooms (L_{eq}: 59-69.9 dB vs. 54.4-57 dB). A meta-analysis from three studies found significantly increased odds (OR: 1.40, 95%-CI: 1.18-1.61) of cognitive impairment in people aged 45+ with higher residential noise exposure (either 10 dB higher noise levels than quieter reference addresses or mean noise level 50-73.8 dB vs. 30.5-49.9 dB). Using GRADE (Grading of Recommendations Assessment, Development and Evaluation) for the evidence synthesis, there was high-quality evidence for an association between environmental noise and cognitive impairment in middle-to-older adults, moderate quality evidence for an association between aircraft noise and reading and language in children, and moderate quality evidence against an association between aircraft noise and executive functioning in children. For other cognitive outcomes, the literature was supportive for an association but with low or very low-quality evidence.

Three papers, characteristics of which are shown in Table 3.17, were published after Thompson et al. (2022) of which one assessed non-European population (Raess et al., 2021) and therefore was excluded from the meta analysis. The two European studies (Tangermann et al., 2023; Foraster et al., 2022) analysed the association of road traffic noise and memory, concentration and inattentiveness. Tangermann et al. (2023) did find a significant reduction in figural memory in the cross-sectional analysis and a significant reduction of concentration constancy z-scores between baseline and follow-up in the longitudinal analysis. Foraster et al. (2022) identified statistically significant associations between school-outdoor noise levels and a slower development of working memory, complex working memory, and a slower improvement of inattentiveness over 12 months.

In summary, these studies provide compelling evidence for a link between transportation noise and cognition in both, children and adults. However, since the outcomes are very diverse, certainty of evidence was not rated. A systematic review is preferable over the Umbrella+ review approach to be able to standardize the outcomes in a manner that they can be included in meta-analysis to derive the ERF.

Table 3.15: Characteristics of the identified reviews investigating the effect of transportation noise on cognition

Review	Outcome [a]	Noise Source [b]	End of search period	Studies				Individual study quality [c]	Evidence rating [d]
				No	N	Countries [c]	Pop		
Zaman (2022)	Cognitive impairment, cognitive function	Road	2021	2	288 – 2,050	DE	Adults	-	-
Thompson (2022)	Different measures of cognitive function	Road, Rail, Aircraft, Env	Jul 2021	48	54 - 191,054	DE, CN, NO, KR, UK, FR, ES, LT, GR, TH, USA, IR, other	Adults, Children	OHAT	GRADE
Terzakis (2022)	Cognitive development, cognitive performance	Road, Aircraft, Env	2020	30	236 - 11,000	NA	Children	-	-
Dohmen (2022)	Reading, Attention, reaction time, memory	Aircraft, Env	2022	8	123 - 340	USA, DE, UK,AT	Children	-	-
Zhao (2021)	Cognitive impairment	Env	Jan 2021	1	288	DE	Adults		-
Clark (2020)	Reading comprehension, mathematics, memory, attention, distraction, adult cognition	Road, Rail, Aircraft	Mar 2019	9	134 - 4,814	DE, GR, ES, ZA, USA	Adults, Children	Own measure	GRADE
Clark (2018)	Reading and oral comprehension, memory, Executive function deficit	Road, Rail, Aircraft, Env	Jun 2015	35	NA	NA (Europe)	Children	WHO	GRADE

[a] Env = Environmental

[b] AT = Austria, CN = China, DE = Germany, ES = Spain, FR = France, GR = Greece, IR = Iran, KR = South Korea, LT = Lithuania, NO = Norway, TH = Thailand, UK = United Kingdom, USA = United States of America, ZA = South Africa; others = not named

[c] OHAT = Office of Health Assessment and Translation Risk of Bias Rating Tool for Human and Animal Studies

[d] GRADE = Grading of Recommendations Assessment, Development and Evaluation

Abbreviations: No = Number of papers, N = Number of participants, Pop = Populations

Table 3.16: Quality assessment of the reviews investigating the effect of transportation noise cognition

Review	Literature search				Risk of Bias		Methodology for Meta-analysis			Comment	Selected for outcome
	a	b	c	d	e	f	g	h	i		
Zaman (2022)	✓	✓	✗	✓	✗	✗	✗	✗	✗	No meta-analysis conducted	
Thompson (2022)	✓	✓	✓	✓	✓	✓	✓	✓	✓		Cognitive functioning
Terzakis (2022)	✓	✓	✓	✓	✓	✗	✗	✗	✗	No meta-analysis conducted	
Dohmen (2022)	✓	✓	✓	✓	✓	✗	✗	✗	✗	No meta-analysis conducted	
Zhao (2021)	✓	✓	✓	✓	✓	✗	✗	✗	✗	No meta-analysis conducted	
Clark (2020)	✓	✓	✓	✓	✓	✓	✗	✗	✗	No meta-analysis conducted	
Clark (2018)	✓	✓	✓	✓	✓	✓	✗	✗	✗	No meta-analysis conducted	

[a] Relevant data base considered
[b] Clearly and adequately defined search terms/keywords
[c] Inclusion/exclusion criteria adequately defined and explained
[d] No critical studies missed
[e] Risk of Bias conducted using adequate Tool
[f] If Risk of Bias in single studies identified, then adequate actions taken in meta-analysis
[g] Appropriateness of data extraction and transformations
[h] Data pooling done in appropriate way
[i] Adequate statistical method used

A detailed description of the criteria for the quality assessment of the reviews is provided in Table 3.2.

Table 3.17: Characteristics of the identified original studies investigating the effects of transportation noise on cognition

Paper	Cohort [a] (Country) [b]	Study type	Study population				Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Beta (95% confidence interval) [c]	Comment
			N	Sex / Age	Follow-up	Setting						
Tangermann (2023)	HERMES (CH)	Longitudinal, cross-sectional	899	Both / 10-17 years	2012-2014; 2014-2016	Residential, school	Road	SonBase	PM_{10}	L_{den}	Cross-sectional: Total memory: -0.09 (-0.43, 0.25) Concentration constancy: 0.00 (-0.09, 0.08) Longitudinal: Total memory: -0.09 (-0.58, 0.41) Concentration constancy: -0.13 (-0.25, 0.00)	Additional outcomes: Verbal and figural memory, concentration accuracy
Raess (2022)	SPROC (BR)	Longitudinal, cross-sectional	3,385 / 1,546	Both / 3, 6 years	2015-2020	Residential	Com-munity	LUR model	-	L_{den}	Cross-sectional: 3-year olds: 0.08 (-0.04, 0.19) 6-year olds: -0.49 (-2.71, 1.74) Longitudinal: 3-year olds: -0.27 (-0.55, 0.00) 6-year olds: 0.27 (-0.55, 0.00)	Non-European study
Foraster (2022)	BREATHE (ES)	Longitudinal	2,680	Both / 7-10 years	2012-2013	Residential, school	Road	Strategic Noise Map for Barcelona	NO_2	L_{den}	Working memory: 1.35 (−0.83, 3.53) per 5dB Complex working memory: 0.40 (−1.32, 2.12) per 5 dB Inattentiveness: -0.52 (−2.06, 1.01) per 5dB	

[a] HERMES = Health effects related to mobile phone use in adolescents, SPROC = Sao Paulo Western Region Birth Cohort, BREATHE = Brain Development and Air Pollution Ultrafine Particles in School Children

[b] CH = Switzerland, ES = Spain, BR = Brazil

[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

Abbreviations: N = Number of participants

3.3.6. Dementia

The literature search revealed 30 records. By screening title and abstract, 19 records were excluded resulting in five reviews and six original studies to undergo the full-text review. Thereof, two reviews (Oudin, 2020; Paul et al., 2019) were not systematic and one original study (Cole-Hunter et al., 2022) had the sole focus on mortality. Thus, three reviews (Zhao et al., 2021; Hegewald et al., 2020; Clark et al., 2020) and five original studies (Yu et al., 2023; Cantuaria et al., 2021; Yuchi et al., 2020; Carey et al., 2018; Andersson et al., 2018) were selected. The PRISMA flow diagram is shown in Figure 8.6 of Annex 2.

The characteristics of the reviews and their quality assessment are shown in Table 3.18 and Table 3.19, respectively. Only one review included a meta-analysis (Zhao et al., 2021). However, this review included not only transportation noise, but also occupational noise in the meta-analysis. As no meta-analysis excluding occupational noise was conducted, no review was selected as starting point.

The characteristics of the five original studies published after 2015 are shown in Table 3.20. Two of these studies (Yu et al., 2023; Yuchi et al., 2022) analysed non-European populations and thus were excluded from further consideration. The three European studies included the investigation of associations between road traffic noise (additionally: one study railway noise) and dementia and two studies also included Alzheimer's and subtypes of dementia. Andersson et al. (2018) identified no significant association on the risk of dementia with a HR of 0.97 (95%-CI: 0.58-1.60) for individuals exposed to noise levels ≥ 55 dB vs. < 55. Carey et al. (2018) also did not find a significant association for dementia (HR: 1.02, 95%-CI: 1.00-1.05), Alzheimer's (HR: 1.03, 95%-CI: 0.99-1.07), vascular dementia (HR: 1.00, 95%-CI: 0.96-1.05) or non-specific dementia (HR: 1.03, 95%-CI: 0.99-1.07) per IQR (2.68 dB L_{night}). The nationwide cohort study by Cantuaria et al. (2021) did find an association between transportation noise and higher risk of all-cause dementia and dementia subtypes, especially Alzheimer's disease. A linearization of the ERF conducted for this report resulted in risk estimates of 1.054 (95%-CI: 1.009-1.102, road) and 1.068 (95%-CI: 1.023-1.115, rail) for dementia and 1.044 (95%-CI: 0.991-1.101, road) and 1.089 (95%-CI: 1.042-1.138, rail) Alzheimer's disease.

Figure 3.11 and Figure 3.12 show the meta-analysis stratified by transportation noise for dementia and Alzheimer's, respectively. The relative risk to develop dementia for road traffic noise is 1.058 (95%-CI: 1.017-1.10) per 10 dB increase in L_{den} without noticeable heterogeneity (p=0.87) between estimates. The relative risk for railway noise is based on one estimate and is 1.068 (95%-CI: 1.023-1.115). The pooled exposure-response estimate is with 1.062 (95%-CI: 1.032-1.094) per 10 dB increase in L_{den} significantly increased and shows no noticeable heterogeneity (p=0.87) between estimates. For Alzheimer's, the risk estimates for railway noise (RR: 1.089, 95%-CI: 1.042-1.138) and road traffic noise (RR: 1.053, 95%-CI: 1.002-1.106) are both significantly increased per 10 dB L_{den} increase. Considering both noise sources combined in the meta-analysis, a pooled RR of 1.072 (95%-CI: 1.034-1.112) per 10 dB L_{den} in relation to Alzheimer's incidence was determined. Associations with rail and road noise were significant in one cohort study and thus the certainty of evidence for an association is considered to be moderate.

Table 3.18: Characteristics of the identified reviews investigating the effect of transportation noise on incidence of dementia

Review	Outcome	Noise Source [a]	End of search period	Studies				Individual study quality [c]	Evidence rating [d]
				No	N	Countries [b]	Pop		
Zhao (2021)	Dementia, Alzheimer	Env	Jan 2021	5	694 - 633,949	CA, SE, UK, USA, DE	Adults	NOS	Own method
Hegewald (2020)	Alzheimer's disease, vascular dementia	Road	Dec 2019	3	1,721 - 3,116,897	DE, ES, SE, UK	Adults	SIGN, CASP	-
Clark (2020)	Incidence of vascular dementia	Road, Rail, Aircraft	Mar 2019	4	1,721 - 754,005	DE, ES, SE, UK	Adults	Own measure	GRADE

[a] Env = Environmental

[b] CA = Canada, DE = Germany, ES = Spain, SE = Sweden, UK = United Kingdom, USA = United States of America

[c] CASP = Critical Appraisal Skills Program, NOS = Newcastle-Ottawa Scale , SIGN = Scottish Intercollegiate Guidelines Network

[d] GRADE = Grading of Recommendations Assessment, Development and Evaluation

Abbreviations: No = Number of papers, N = Number of participants, Pop = Populations

Table 3.19: Quality assessment of the reviews investigating the effect of transportation noise on incidence of dementia

Review	Literature search				Risk of Bias		Methodology for Meta-analysis			Comment	Selected for outcome
	a	b	c	d	e	f	g	h	i		
Zhao (2021)	✓	✓	✓	✓	✓	✓	✗	✗	✓		
Hegewald (2020)	✓	✓	✓	✓	✓	✗	✗	✗	✗	No meta-analysis conducted	
Clark (2020)	✓	✓	✓	✓	✓	✓	✗	✗	✗	No meta-analysis conducted	

[b] Clearly and adequately defined search terms/keywords
[c] Inclusion/exclusion criteria adequately defined and explained
[d] No critical studies missed
[e] Risk of Bias conducted using adequate Tool
[f] If Risk of Bias in single studies identified, then adequate actions taken in meta-analysis
[g] Appropriateness of data extraction and transformations
[h] Data pooling done in appropriate way
[i] Adequate statistical method used

A detailed description of the criteria for the quality assessment of the reviews is provided in Table 3.2.

Table 3.20: Characteristics of the identified original studies investigating the effects of transportation noise on dementia

Paper	Cohort [a] (Country) [b]	Study type	Study population			Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c]	Comment
			N	Sex / Age	Follow-up						
Yu et al. (2023)	SALSA (USA)	Longitudinal	1,612	Both / mean 70.2 years	1998 - 2007	Road	SoundPLAN	-	$L_{eq,24h}$	Dementia: 1.19 (0.92, 1.53) per 11.6 dB	Non-European study
Cantuaria (2021)	Nationwide cohort (DK)	Longitudinal	1,938,994	Both / 60+ years	2004 - 2017	Road Rail	Nordic prediction method	-	L_{den}	Dementia: Road: 1.054 (1.009, 1.102) [$] Rail: 1.068 (1.023. 1.115) [$] Alzheimer's: Road: 1.044 (0.991, 1.101) [$] Rail: 1.089 (1.042, 1.138) [$]	Additional outcomes: vascular dementia
Yuchi et al. (2020)	Metro Vancouver (CA)	Longitudinal	678,000	Both / 45-84 years	1994-2003	Env	CadnaA	-	L_{den}	Dementia: 1.01 (0.99, 1.04) per 5.53 dB Alzheimer's: 0.99 (0.92, 1.08) per 5.75 dB	Non-European study
Carey (2018)	CPRD (UK)	Longitudinal	130,978	Both / 50-79 years	2005 - 2013	Road	TRANEX	-	L_{night}	Dementia: 1.02 (1.00, 1.05) per 2.68[§] Alzheimer's: 1.03 (0.99, 1.07) per 2.7[§]	Additional outcomes: Vascular dementia non-specific dementia
Andersson (2018)	Betula project (SE)	Longitudinal	1,721	Both / 50+ years	1988 - 2010	Road	Umeå Municipality Noise Survey	NO_X	L_{den}	Dementia: ≥ 55 dB vs. < 55: 0.97 (0.58, 1.60)	

[a] CPRD = Clinical Practice Research Datalink, SALSA = Sacramento Area Latino Study on Aging
[b] CA = Canada, DK = Denmark, SE = Sweden, UK = United Kingdom, USA = United States of America
[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.
[§] The relative risk has been converted to per 10 dB (based on reported effect size per increment in original study).
[$] Estimate derived from categorical results.

Abbreviations: N = Number of participants

Figure 3.11: Meta-analysis of cohort studies on dementia in relation to transportation noise, stratified by source. Relative risks refer to a 10 dB increase in L_{den}

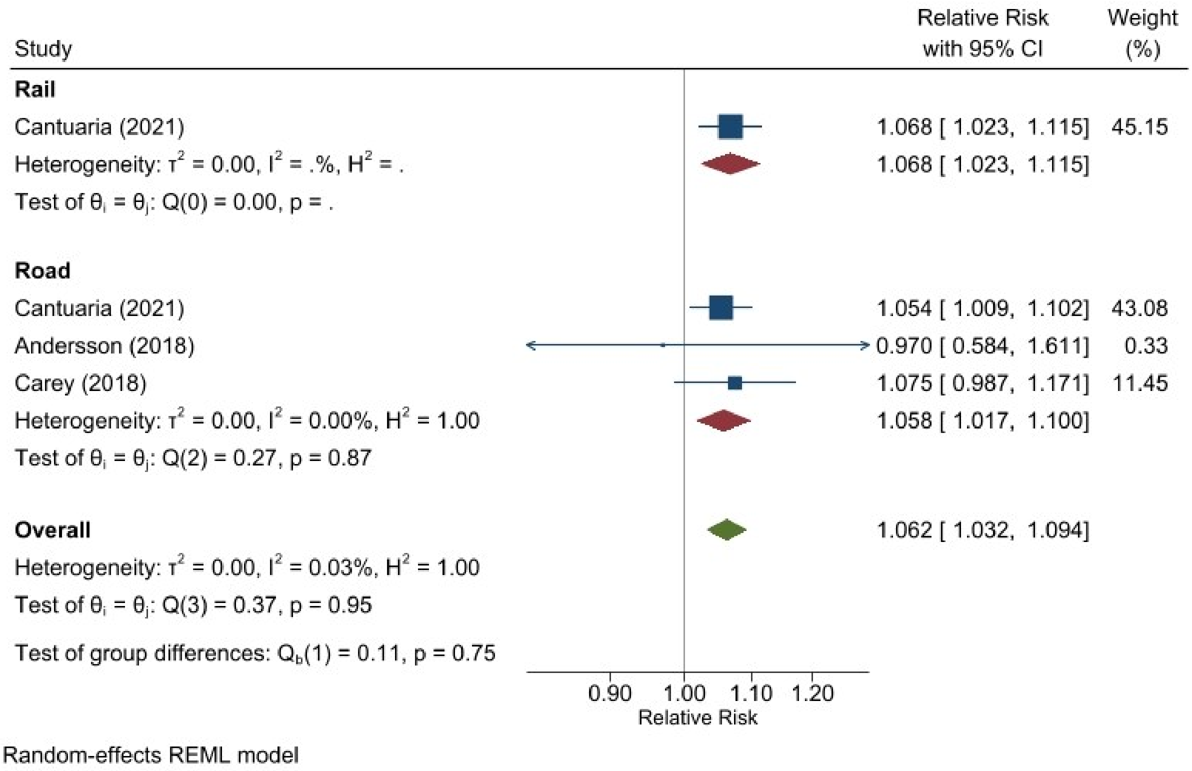
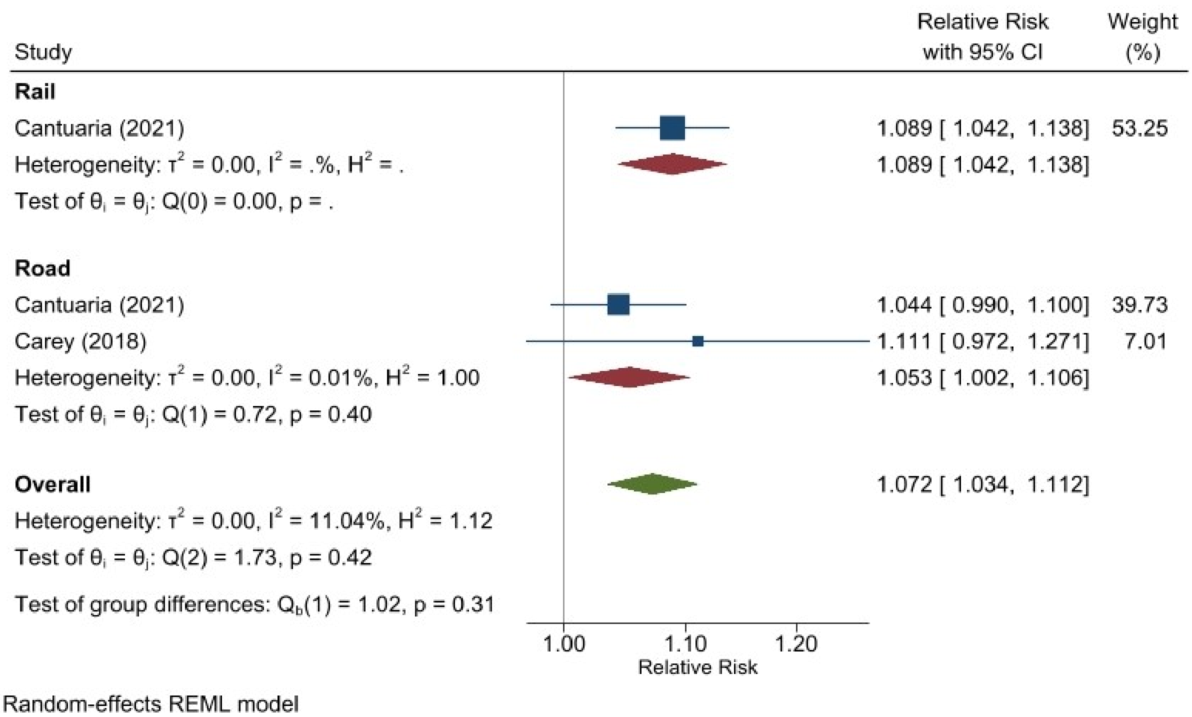


Figure 3.12: Meta-analysis of cohort studies on Alzheimer's in relation to transportation noise, stratified by source. Relative risks refer to a 10 dB increase in L_{den}



3.3.7. Behavioural problems

After identifying 158 papers through the literature search and one additional by manual search, 147 records were excluded by screening the titles and abstracts and no further papers during the full-text review. Therefore, four reviews (Baird et al., 2023; Schubert et al., 2019; Zare Sakhvidi et al., 2018a; Clark and Paunovic, 2018) and eight original studies (Bao et al., 2022; Essers et al., 2022; Raess et al., 2022; Tangermann et al., 2022; Yuchi et al., 2022; Zijlema et al., 2021; Forns et al., 2016; Hjortebjerg et al., 2016) were selected for further consideration. The PRISMA flow diagram is shown in Figure 8.7 of Annex 2.

The characteristics of the four reviews are shown in Table 3.21. The sole focus of all reviews is on children. Most studies included in the four reviews investigated the impact of road traffic and aircraft noise. Further, the measurement tool for behaviour primarily used among all studies is the Strength and Difficulties Questionnaire (SDQ).

The only review that included a meta-analysis is by Schubert et al.(2019). The meta-analysis is on the association between road traffic noise and behavioural problems in children. Using the SDQ four behavioural-related outcomes including emotional symptoms, conduct problems, hyperactivity/inattention and peer relationships problem were assessed as well as all four outcomes combined. The results of the meta-analysis indicated a 11% (95%-CI: 4-19%) increased odds for hyperactivity/inattention and a 9% (95%-CI: 2-16%) increase in the odds for total difficulties per 10 dB L_{den} of road traffic noise at home. All three studies were of cross-sectional design. Although 10 studies were identified in the review, only three of them were included in the meta-analysis. The decision to exclude these seven studies was based on the outcome measure. Although all studies used comparable scales, studies in which the questionnaire scores were used as a continuous variable in the analysis were excluded from the meta-analysis. The loss of information using a meta-analysis estimates on the dichotomized outcome measure is thus substantial.

The characteristics of the six original studies that were published after the end of the search period of the selected review are shown in Table 3.22. However, three studies (Bao et al., 2022; Raess et al., 2022; Yuchi et al., 2022) were conducted with non-European populations and therefore excluded from further consideration. Essers et al. (2022) did not find strong indications for an association between noise at home and continuously measured behavioural problems using EU maps from road traffic noise and total noise (road, aircraft, railway, and industry). Tangermann et al. (2022) observed significant increased peer problems in relation to road traffic noise at home in a cross-sectional analysis, whereas the longitudinal analysis provided little indication of a link. This study analysed continuous and categorical outcomes measures. Zijlema et al. (2021) analysed the association between road traffic noise and symptoms of Attention deficit/hyperactivity disorder (ADHD), for which the association was not significant, and ADHD diagnosis, for which a significant association was identified. We have also identified a large study from Sao Paolo and thus not eligible for this review. Nevertheless, the results in this young age (3 and 6 years) group living in a noisy community (L_{den} levels mostly between 60 and 70 dB) were remarkable. Here, the cross-sectional study assessing the association between community noise and the total difficulty score describing the behavioural development indicated a 32% (95%-CI: 4- 68%) increased odds for the 3 year olds per 10 dB L_{den} and an increase of the total difficulty score by 0.72 (95%-CI: 0.55, 0.88) in the 6 year olds per 10 dB L_{den}. The longitudinal association also showed an increase in the total difficulty score (β=0.62 (95%-CI: 0.38-0.87)) per 10 dB L_{den} in community noise.

In summary, several studies observed an association between behavioural problems and transportation noise. Current available pooled estimates from a meta-analysis are based on a small sample of studies and thus does not represent all available evidence from all published studies. As discussed for cognition, the outcomes and analytical approaches are diverse and the current Umbrella+ review approach did not allow to rate the certainty of evidence. A more systematic review is preferable to be able to standardize the outcomes in a manner that they can be included in meta-analysis to derive the ERF.

Table 3.21: Characteristics of the identified reviews investigating the impact of transportation noise on behaviour problems

Review	Outcome [a]	Noise Source	End of search period	Studies					Individual study quality [c]	Evidence rating [d]
				No	N	Countries [b]	Pop	Setting		
Baird (2023)	All included in SDQ, other	Road, Rail, Aircraft	Jun 2022	11	275 - 7,958	CN, UK, ES, NL, FR, DE	Children	Residential, School	PR	GRADE
Schubert (2019)	All included in SDQ	Road, Rail, Aircraft	Feb 2019	10	311 - 46,940	RS, UK, DK, ES, NL, DE, KR, NO	Children	Residential, School	SIGN, CASP	-
Zare Sakhvidi (2018a)	All included in SDQ	Road, Rail, Aircraft	Mar 2018	12	399 - 46,940	NO, DK, ES, DE, EU countries, UK, MK, BG, AT	Children	Residential, School, Work	PR	GRADE
Clark (2018)	Emotional and Conduct Disorders, Hyperactivity, other	Road, Rail, Aircraft	Oct 2015	13	NA	NA (Europe)	Children	Residential, School	WHO	GRADE

[a] SQD = Strength and Difficulties Questionnaire

[b] AT = Austria, BG = Bulgaria, CN = China, DE = Germany, DK = Denmark, ES = Spain, EU = European Union, FR = France, KR = South Korea, MK = Macedonia, NL = Netherlands, NO = Norway, RS = Serbia, UK = United Kingdom

[c] CASP = Critical Appraisal Skills Program, PR = previously used checklist, SIGN = Scottish Intercollegiate Guidelines Network, WHO = WHO checklist

[d] GRADE = Grading of Recommendations Assessment, Development and Evaluation

Abbreviations: No = Number of studies, N = Number of participants, Pop = Population

Table 3.22: Characteristics of the identified original studies investigating the effects of transportation noise on behaviour problems

Paper	Cohort [a] (Country) [b]	Study type	Study population				Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c,d]	Comment
			N	Sex / Age	Follow-up	Setting						
Yuchi (2022)	Metro Vancouver (CA)	Longitudinal	28,797	Both / 3-10 years	2003 - 2010	Residential	Env	CadnaA	-	L_{den}	ADHD: 1.00 (0.95-1.05) per 6.91 dB	Non-European study Binary outcome measure
Tangermann (2022)	HERMES (CH)	Longitudinal, cross-sectional	899	Both / 10-17 years	2012-2016	Residential	Road	SonBase	PM_{10}	L_{den}	Total Difficulties: Cross-sectional: 0.16 (-0.21, 0.53)* Longitudinal: -0.20 (-0.60, 0.20)*	Continuous and categorical outcome measure Additional: sub-outcomes of SDQ
Essers (2022)	INMA (ES), Generation R (NL)	Longitudinal	534, 7,424	Both / 4, 7, 9 years, Both / 18 months, 3, 5, 9 years	2004-2015, 2002-2011	Residential	Road	Noise maps developed in 2012 for Rotterdam and 2006 and 2012 for Sabadell	-	L_{den}	Aggressive: 0.00 (-0.02, 0.01)* Emotional: -0.01 (-0.03, 0.00)* ADHD: 0.00 (-0.02, 0.02)*	Continuous outcomes measure
Bao (2022)	Guangzhou (CN)	Cross-sectional	3,236	Both / 7–13 years	-	Residential	Road	Modelled maps with a resolution of 8.0 × 8.0 m	NO_2	L_{dn}	Total Difficulties: Categorical outcome measure (abnormal): 1.07 (0.84, 1.36)* Continuous outcome measure: 0.33 (0.08, 0.59)*	Non-European study Continuous and categorical outcomes measure Additional sub-outcomes of SDQ
Raess (2022)	SPROC (BR)	Longitudinal	3,385 / 1,546	Both / 3, 6 years	2015-2020	Residential	Com-munity	LUR model	-	L_{den}	Cross-sectional: 3-year olds: 1.32 (1.04, 1.68) 6-year olds: 0.72 (0.55, 0.88)	Non-European study

Paper	Cohort [a] (Country) [b]	Study type	Study population				Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c,d]	Comment
			N	Sex / Age	Follow-up	Setting						
											Longitudinal: 3-year olds: 0.62 (0.38, 0.87) 6-year olds: 0.52 (0.28, 0.77)	
Zijlema (2021)	TRAILS (DK)	Cross-sectional	1,710	Both / mean 10.6 years	-	Residential, School	Road	STAMINA	-	L_{den}	ADHD diagnosis: Residential: 0.929 (0.893, 0.965) School: 0.945 (0.910, 0.981) ADHD symptoms: Residential: 0.994 (0.969, 1.019) School: 0.997 (0.972, 1.022)	Categorical outcome measure

[a] HERMES = Health effects related to mobile phone use in adolescents, INMA = INfancia y Medio Ambiente, Environment and Childhood, SPROC = Sao Paulo Western Region Birth Cohort, TRAILS = Tracking Adolescents' Individual Lives Survey

[b] BR = Brazil, CA = Canada, CH = Switzerland, CN = China, ES = Spain, NL = Netherlands

[c] If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

[d] ADHA = Attention Deficit/Hyperactivity Disorder

* Original results reported as beta

Abbreviations: N = Number of participants

3.3.8. Overweight

The literature search revealed 79 potential papers. By screening the titles and abstracts, 59 records were excluded, resulting in 5 reviews and 15 original studies for the full-text evaluation. During the full-text evaluation one review was excluded due to being not systematic (Belojević and Paunović, 2016) and one original study due to having analysed noise perception and not measured or modelled noise levels (Huang et al., 2020). Therefore, 4 reviews (Gui et al., 2022; Malacarne et al., 2022; Wang et al., 2021; An et al., 2018) and 14 original studies (de Bont et al., 2021; Cai et al., 2020; Bloemsma et al., 2019a, 2019b; Cramer et al., 2019; Wallas et al., 2019; Foraster et al., 2018; Weyde et al., 2018; Pyko et al., 2017; Christensen et al., 2016a, 2016b, 2015; Oftedal et al., 2015; Pyko et al., 2015) were selected. The PRISMA flow diagram is shown in Figure 8.8 of Annex 2.

The characteristics of the reviews and the results of their quality assessment can be found in Table 3.23 and Table 3.24, respectively. All reviews included the outcomes body mass index (BMI) and overweight, but also additional outcomes such as obesity, waist circumference, and waist-hip-ratio were analysed. The sole focus of two reviews was on children, whereas the other two reviews also included adults.

Two of the four reviews did include a meta-analysis (Gui et al., 2022; An et al., 2018). The latest review published by Gui et al. (2022) is of good quality and its meta-analysis was on the association between road, railway and aircraft noise and a number of weight-related health outcomes. The meta-analysis showed no statistically significant association with obesity for any noise source with pooled ORs ranging from 0.974 (n=2, 95%-CI: 0.841-1.107) to 1.032 (n=14, 95%-CI: 0.989-1.076) per 10 dB L_{den} increase. However, road traffic noise was significantly associated with overweight with an OR of 1.279 (n=3, 95%-CI: 1.051-1.507) per 10 dB increase in L_{den}. Furthermore, a statistically significant increase in BMI (β: 0.026 (n=4, 95%-CI: 0.001-0.051)) was identified for railway noise, an increase in waist-hip ratio (β: 0.320 (n=1, 95%-CI: 0.110-0.530)) and weight (β: 0.011 (n=3, 95%-CI: 0.000-0.022)) for road traffic noise as well as for aircraft noise (waist-hip ratio: β: 1.160 (n=4, 95%-CI: 0.0.450-1.870), weight: β: 0.030 (n=1, 95%-CI: 0.015-0.045)). Most of the estimates calculated by Gui et al. (2022) were based on one to five studies, resulting in a relatively small sample size per analysis. However, for the associations between waist circumference, BMI and obesity and either transportation noise or road traffic noise eight to 18 studies were available. For the various outcomes, Gui et al. (2022) rated the evidence as moderate or very low.

Since the end of the search period of Gui et al. (2022), one new study by de Bont et al. (2021) was published. The cross-sectional study presented no statistically significant association between road traffic noise (per 8 dB L_{den} increase) and overweight (OR: 1.080 (95%-CI: 0.970-1.220)), waist circumference (β: 0.020 (95%-CI: -0.030-0.080)), BMI (β: 0.060 (95%-CI: -0.000-0.120)) or waist-hip ratio (β: 0.020 (95%-CI: -0.030-0.070)). Since the study conducted by de Bont et al. (2021) is cross-sectional, the evidence rating remained the same.

Table 3.23: Characteristics of the identified reviews investigating the effect of transportation noise on overweight

Review	Outcome [a]	Noise Source [b]	End of search period	Studies				Individual study quality [d]	Evidence rating [e]
				No	N	Countries [c]	Pop		
Gui (2022)	BMI, WC, WHR, obesity, overweight, body fat, weight	Road, Rail, Aircraft, Env	Feb 2021	13	484 - 412,934	SE, DK, UK, NO, BG, SK, CH, NL	Adults, Children	NTP/OHAT	GRADE
Malacarn (2022)	BMI, overweight	Traffic	Feb 2020	4	3,403 - 40,974	SE, DK, NL, NO	Children	NOS	-
Wang (2021)	BMI, obesity, overweight	Traffic	Jan 2019	6	115 - 40,974	NL, DK, AT, DE, SE, NO	Children	NIH	-
An (2018)	BMI, WC, WHR, body fat, weight	Road, Rail, Aircraft	Feb 2018	11	132 - 52,456	SE, IT, DK, BG, NO, USA, SK	Adults, Children	Self-developed	GRADE

[a] BMI = Body mass index, WC = Waist circumference, WHR = Waist-hip-ratio

[b] Env = Environmental

[c] AT = Austria, BG = Bulgaria, CH = Switzerland, DE = Germany, DK = Denmark, IT = Italy, NL = Netherlands, NO = Norway, SE = Sweden, SK = Slovakia, UK = United Kingdom, USA = United States of America

[d] NIH = National Institutes of Health's Quality Assessment Tool for Observational Cohort and Cross-Sectional Studies, NOS = Newcastle-Ottawa Scale, NTP/OHAT = National Toxicology Program/Office of Health Assessment and Translation Risk of Bias Rating Tool for Human and Animal Studies

[e] GRADE = Grading of Recommendations Assessment, Development and Evaluation

Abbreviations: No = Number of studies, N = Number of participants, Pop = Population

Table 3.24: Quality assessment of the reviews investigating the effect of transportation noise on overweight

Review	Literature search				Risk of Bias		Methodology for Meta-analysis			Comment	Selected for outcome
	a	b	c	d	e	f	g	h	i		
Gui (2022)	✓	✓	✓	✓	✓	✓	✓	✓	✓	Possibly data from SPDD cohort have been used multiple times in the same meta-analysis (Pyko 2015, Eriksson 2014, Pyko 2017)	BMI, WC, WHR, obesity, overweight, body fat, weight
Malacarn (2022)	✓	✓	✓	✓	✓	✗	✗	✗	✗	No meta-analysis conducted	
Wang (2021)	✓	✓	✓	✓	✓	✗	✗	✗	✗	No meta-analysis conducted	
An (2018)	✓	✓	✓	✓	✓	✓	✓	✓	✓	Newer Review on WC and BMI available	

[a] Relevant data base considered
[b] Clearly and adequately defined search terms/keywords
[c] Inclusion/exclusion criteria adequately defined and explained
[d] No critical studies missed
[e] Risk of Bias conducted using adequate Tool
[f] If Risk of Bias in single studies identified, then adequate actions taken in meta-analysis
[g] Appropriateness of data extraction and transformations
[h] Data pooling done in appropriate way
[i] Adequate statistical method used

A detailed description of the criteria for the quality assessment of the reviews is provided in Table 3.2.

Table 3.25: Characteristics of the identified original studies investigating the effect of transportation noise on overweight

Paper	Cohort [a] (Country) [b]	Study type	Study population N	Study population Sex / Age	Study population Follow-up	Noise source	Exposure characterization	Adjustment for air pollution	Exposure metric	Relative risk (95% confidence interval) [c,d]	Comment
de Bont (2021)	ECHOCAT, INMA (ES)	Cross-sectional	2,213	both / 9-12 years	-	Road	GENCAT	-	L_{den}	BMI: 0.06 (0.00-0.12)* WC: 0.02 (-0.03-0.08)* Body fat: 0.02 (-0.03-0.07)* Overweight: 1.08 (0.97-1.22)	Separate analysis of L_{night}

a ECHOCAT = Urban built environment and childhood obesity in Catalonia, INMA = INfancia y Medio Ambiente, Environment and Childhood

b ES = Spain

c If not otherwise indicated, relative risks refers to a 10 dB increase related to the maximum noise value.

d BMI = Body Mass Index, WC = Waist Circumference

* Original results reported as beta

Abbreviations: N = Number of participants

3.4. Summary of evidence review

3.4.1. Selection of outcomes

Table 3.26 gives an overview of the evidence rating for all evaluated outcomes in relation to road, railway and aircraft noise. The evidence for all-cause mortality was considered to be high and thus this outcome is selected to calculate years of life lost (YLL) from transportation noise. To avoid double counting, no cause specific mortality such as cardiovascular or ischaemic heart disease mortality is selected. Thus, our literature review and meta-analyses focus on cohort studies that addressed incidence (or also prevalence in the case of hypertension). These studies will be used to calculate years of healthy life lost due to disability (YLD).

In terms of cardiovascular disease, we found few studies addressing incidence of all types of cardiovascular disease. On the other hand, there were many more studies addressing specific diagnostic groups of cardiovascular disease. Although for some combinations of specific diagnostic groups with specific noise sources, the number of studies is very small, in particular for railway noise (Table 3.26). We have pooled results of the five main cardiovascular diagnostic groups in relation to transportation noise (Figure 3.7) and road traffic noise (Figure 3.8). Since these diagnoses account for the vast majority of cardiovascular diseases, the combined effect estimates of all these cardiovascular outcomes will be used to calculate YLD from all cardiovascular diagnoses.

The literature review provided also compelling evidence for a link between transportation noise and diabetes and this outcome will also be considered in the HRA. Further, the association of reading impairments in children due to aircraft noise is mentioned in the Annex III of the Environmental Noise Directive (END) and will thus be included again.

In the literature review we also identified a number of emerging effects from transportation noise which are not mentioned in the Annex III of the END in 2020. These effects are not (yet) considered in the HRA for various reasons discussed in the following. There is convincing evidence that cognition in both children and adults is affected by transportation noise. However, the outcomes are heterogeneous and to be included in a HRA would need a more systematic approach to synchronize the study outcomes with available baseline cognition data. Therefore, we will only consider reading and language impairment in children in relation to aircraft noise for the HRA as previously specified in previous EEA noise EU-wide HRA assessments (EEA, 2020a).

A similar situation was observed for behavioural problems in children and adolescent in relation to transportation noise. Several studies point to an association, but the current results have not been systematically collated to obtain an ERF. This includes reconciling the different scales to measure behavioural problems and different analytical methods. Therefore, a systematic review of the association between transportation noise and behaviour would be required to include all available information and the outcome need to be synchronized with available baseline data on behavioural problems including determination of disability weights.

Further, several studies reported an association between transportation noise with overweight, increased BMI or waist-to-hip ratio. Whereas change in BMI and waist-to-hip ratio have some ambiguity in terms of their meaning for health, an increase in the obesity rate would be of high public health relevance. However, a recent meta-analysis from Gui et al. (2022) did not find a significant association between transportation noise exposure and obesity.

Dementia and depression has been also identified as new emerging topic in health research to be considered in an extended HRA.

Table 3.26: Overview of the updated evidence rating and selection of outcomes

Outcome	Road	Railway	Aircraft	Selected for main HRA	Selected for additional HRA
%HA[$]	High	High	High	yes	-
%HSD[$]	High	High	High	yes	-
All-cause mortality	High	Very low	Very low	yes	-
All cardiovascular diseases*	High	Moderate	Low	yes	-
Ischaemic heart disease	*High*	*Moderate*	*Low*	*Included in all CV*	-
Hypertension	*Very low*	-	*Low*	*Included in all CV*	-
Arrhythmia	*Very low*	*Very low*	*Very low*	*Included in all CV*	-
Stroke	*High*	*Very low*	*Very low*	*Included in all CV*	-
Heart failure	*High*	*Very low*	*Low*	*Included in all CV*	-
Diabetes	High	Low	Low	*yes*	-
Depression	Low	Very low	High	no	yes
Cognitive impairment in adults	Moderate (environmental noise)			no	yes
Reading and language in children	-	-	Moderate	yes	-
Executive function	-	-	Moderate	no	yes
Dementia	Moderate	Moderate	Very low	no	yes
Behavioural problems	Moderate	Very low	Very low	no	yes
Overweight	Very low to moderate for various markers of overweight			no	no

[$] Not evaluated in this literature review, evidence rating according to WHO ENG.
* Summary rating across all cardiovascular diagnostic groups

3.4.2. Exposure-response functions

Table 3.27 shows the proposed ERFs and outcomes to be used in an EU-wide HRA. This proposal is based on the results of the meta-analyses and evidence review presented in previous chapters of this report. For %HA and %HSD the relationships outlined by the WHO Environmental Noise Guidelines are proposed to be used, as those relationships are part of the END and were rated as high quality. The number of studies and their precision allowed for calculation of source specific ERF curves. For mortality, cardiovascular disease and diabetes, a pooled ERF for road traffic noise is proposed to be used to estimate the health risks of also rail and aircraft noise. This is based on the assumption that the biological mechanisms involved are thought to be similar for different sources but research on these two sources is much more scarce. A particular concern is that noise from railway and aircraft is often masked by the substantially more prevalent road traffic noise and thus the exposure-response association may not be accurately estimated. For cognitive impairment, the proposed ERF relates to aircraft noise only and is based on the same relationship used in the previous EEA Noise in Europe Report (EEA, 2020a).

Table 3.27: Overview of the proposed ERFs and outcomes to be used in an EU-wide HRA

Outcome	Source	ERF	Reference
High noise annoyance (prevalence in adults)	Road	$\%HA = 78.9270 - 3.1162 \cdot L_{den} + 0.0342 \cdot L_{den}^2$	Guski et al. (2017)
	Railway	$\%HA = 38.1596 - 2.05538 \cdot L_{den} + 0.0285 \cdot L_{den}^2$	Guski et al. (2017)
	Aircraft	$\%HA = -50.9693 + 1.0168 \cdot L_{den} + 0.0072 \cdot L_{den}^2$	Guski et al. (2017)
	Industry	%HA = 1-normal (72 - (-126.52 + (L_{den})·(2.49)))/sqrt(2054.43))	Miedema and Vos (2004)
High sleep disturbance (prevalence in adults)	Road	$\%HSD = 19.4312 - 0.9336 \cdot L_{night} + 0.0126 \cdot L_{night}^2$	Basner and McGuire (2018)
	Railway	$\%HSD = 67.5406 - 3.1852 \cdot L_{night} + 0.0391 \cdot L_{night}^2$	Basner and McGuire (2018)
	Aircraft	$\%HSD = 16.7885 - 0.9293 \cdot L_{night} + 0.0198 \cdot L_{night}^2$	Basner and McGuire (2018)
	Industry	%HSD=1-normal(72 - (-90.70 + (L_{night})·(1.80)))/sqrt(1,789 + 272))	Miedema and Vos (2007)
All-cause mortality (adults)	Road, rail and aircraft	Relative risk (RR) derived from road noise RR= 1.055 (95%-CI: 1.014-1.069) per 10 dB	Meta-analyses Chapter 3.3.1
Cardiovascular disease (incidence in adults)	Road, rail and aircraft	Relative risk (RR) derived from road noise RR=1.032 (95%-CI: 1.012-1.052) per 10 dB	Meta-analyses Chapter 3.3.2
Diabetes (incidence in adults)	Road, rail and aircraft	Relative risk (RR) derived from road noise RR=1.062 (95%-CI: 1.036-1.088) per 10 dB	Meta-analyses Chapter 3.3.3
Reading Comprehension (prevalence in children)	Aircraft	P(reading)= 1/(1 + exp(- (ln(0.1/0.9) + (ln(1.38)/10·(L_{den} - 50)))) if $L_{den} \geq 50$ dB and 0.1 if $L_{den} < 50$ dB	Clark et al. (2006) and van Kempen (2008)

3.4.3. *Effect threshold*

In the previous EEA HRA, health effects were quantified above the END reporting threshold of 55 dB due to lack of exposure data below this level. However, the exposure-response functions for %HA and %HSD were based on ERFs that had their minimum at lower levels (L_{den} ≤45 dB for annoyance and L_{night} ≤40 dB for sleep disturbance) Table 3.27. For ischaemic heart disease, the exposure-response function was based on a minimum level of 53 dB and for reading comprehension on 50 dB L_{den}. However, the health impact estimates were calculated on levels starting at 55 dB L_{den} due to lack of data below this threshold.

Table 3.28 shows the evaluation of the lowest effect threshold from our Umbrella+ review. Out of 46 effect estimates (each represented as a row), information about the exposure-response curve was available for 36 estimates. For about half of these estimates (20), the risk monotonically (or nearly monotonically) increased from the lowest modelled level. In 21 out of 35 exposure-response functions, the risk was observed to increase at L_{den} levels of 45 or below.

Thus, to calculate the health impacts of noise, it is proposed to use a threshold of L_{den}=45 dB for %HA, cardiometabolic outcomes and mortality, a threshold of L_{den}=50 dB for reading comprehension in children and a threshold of L_{night}=40 for %HSD.

Table 3.28: Overview of the lowest effect thresholds for studies on the selected outcomes mortality, cardiovascular diseases and diabetes

Reference	Outcome	Noise source	L_{den} with RR>1[1]	L_{den} with RR>1 and p<0.05[1]	Monotonic increase from lowest level	Comments
Vienneau (2023)	mortality	road	35	38	yes	Refers to splines from Vienneau (2022) on cardiovascular mortality
Vienneau (2023)	mortality	rail	30	32	yes	
Vienneau (2023)	mortality	aircraft	50	55	no	
Sørensen (2023a)	mortality	road	45	45	yes	
Sørensen (2023a)	mortality	rail	35	35	yes	
Cole-Hunter (2022)	mortality	road	20	40	yes	refers to 23-year mean
Hao (2022b)	mortality	road	-	-	-	Not reported
Klompmaker (2021)	mortality	road	-	-	-	Not reported
Klompmaker (2021)	mortality	rail	-	-	-	Not reported
Thacher (2020)	mortality	road	55	55	yes	Data only presented for stratified analysis ($PM_{2.5}$ ≤ 20 µg/m³)
Andersson (2020)	mortality	road	63	>63	no	
Thacher (2022a)	IHD	road	45	50	yes	
Thacher (2022a)	IHD	rail	-	-	-	Not reported
Thacher (2022a)	IHD	aircraft	45	>55	yes	
Pyko (2023)	IHD	road	55	59	yes[2]	
Pyko (2023)	IHD	rail	40	44	yes[2]	
Pyko (2019)	IHD	aircraft	40	>50	yes[2]	
Carey (2016)	IHD	road	74	>74	no	Converted from L_{night}
Kourieh (2022)	Hypertension	aircraft	-	-	-	Not reported
Gu (2023)	stroke	road	53	>56	yes	
Roswall (2021)	stroke	road	45	60	yes[2]	
Roswall (2021)	stroke	railway	45	>70	no	
Sørensen (2021)	stroke	road	45	45	yes	
Sørensen (2021)	stroke	railway	35	35	no	
Dimakopoulou (2017)	stroke	road	-	-	-	Not reported
Dimakopoulou (2017)	stroke	aircraft	-	-	-	Not reported
Carey (2016)	stroke	road	>74	>74	no	Converted from L_{night}

Reference	Outcome	Noise source	L_{den} with RR>1[1]	L_{den} with RR>1 and p<0.05[1]	Monotonic increase from lowest level	Comments
Thacher (2022a)	heart failure	road	45	50	yes	
Thacher (2022a)	heart failure	rail	-	-	-	Not reported
Thacher (2022a)	heart failure	aircraft	45	45	yes[2]	
Lim (2021)	heart failure	road	-	-	-	Not reported
Sørensen (2017)	heart failure	road	55	60	no	
Carey (2016)	heart failure	road	74	>74	no	Converted from L_{night}
Sørensen (2023b)	diabetes	road	50	>65	yes[2]	
Zuo (2022)	diabetes	road	57	>57	yes	
Thacher (2021a)	diabetes	road	45	45	yes	
Thacher (2021a)	diabetes	rail	40	42.5	no	
Thacher (2021a)	diabetes	aircraft	45	50	no	
Jørgensen (2019)	diabetes	road	40	>58	yes	
Ohlwein (2019)	diabetes	road	52.2	>61.1	no	
Roswall (2018)	diabetes	road	-	-		Not reported
Eze (2017)	diabetes	road	40	-		Significance not derivable
Eze (2017)	diabetes	rail	>70	>70		
Eze (2017)	diabetes	aircraft	30	-		Significance not derivable
Dimakopoulou (2017)	diabetes	road	-	-	-	Not reported
Dimakopoulou (2017)	diabetes	aircraft	-	-	-	Not reported

[1] Any value in this column with ">" means that no significant effect was observed. The value refers to the highest category of the corresponding study.

[2] Slight deviations from monotonic increase were observed, which are considerably smaller than the overall exposure-response pattern.

4. PART III: Calculation Methods

4.1. Overview on the calculation method

To estimate the burden of disease attributable to noise exposures from road, rail, aircraft, and industry we propose to use the standard approach for calculating attributable fraction (PAF), the attributable number of cases, and subsequent calculation of YLD, YLL and DALYs (for formula see Annex 3). This approach was already used in several EEA products and has also been used in recent studies on the burden of disease of noise (Jephcote et al., 2023). The calculation includes the following elements:

- Assessment of the population noise exposure (see PART I)
- Deriving exposure-response association including threshold value (PART II)
- Deriving the country specific health data (see Chapter 4.2).
- Determining the DW (see Chapter 4.3)
- Deriving the costs for the quantification (see Chapter 4.4)
- Calculation of the attributable fraction and numbers (see Chapter 4.1)

Calculation of the attributable numbers

Disability-adjusted life years (DALY) is defined as the sum of years of life lost due to death (YLL) and years of life lost due to health restrictions (YLD).

$$DALY = YLL + YLD \qquad (1)$$

For cardiometabolic effects the exposure-response association is expressed in relative risk increase per 10 dB increase in transportation noise. To calculate the attributable fraction (AF_{tot}) and YLDs is relative risk per each noise exposure category ($RR[N_i]$) of the target population compared to the effect threshold (E_{thres}) is calculated from relative risk per 10 dB (RR_{10dB}) as following:

$$RR[N_i] = \exp\left(\frac{\ln(RR_{10dB})}{10} \cdot (M[N_i] - E_{thres})\right) \qquad (2)$$

Where $M[N_i]$ is the midpoint of the corresponding noise category in dB (L_{den}).

Total attributable fraction from noise exposure (AF_{tot}) corresponds then to the sum of the attributable fractions in each noise exposure category (AF_{N_i}) as following:

$$AF_{tot} = \sum_{i=1}^{n} AF_{N_i} = \sum_{i=1}^{n} \frac{p_i \cdot (RR[N_i] - 1)}{p_i \cdot (RR[N_i] - 1) + 1} \qquad (3)$$

Where p_i is the proportion of the population in each noise exposure category N_i and $RR[N_i]$ is the relative risk of the corresponding noise exposure categories (N_i).

For %HA, %HSD and reading impairment in children the attributable fraction (AF_{tot}) is obtained by summing up the product of the exposure-response function of the all noise exposure categories (ERF_{N_i}) (Table 3.27) multiplied with the corresponding proportion of the population (p_i) exposed in this noise exposure category.

$$AF_{tot} = \sum_{i=1}^{n} p_i \cdot ERF_{N_i} \qquad (4)$$

YLD for %HA, %HSD and reading impairment in children is calculated according to Equation 5. In this case C_o represents the total target population size. Disability weights used for %HA and %HSD outcomes are based on Charalampous et al. (2024) and WHO Europe (2024) and for reading impairment on WHO Europe (2018).

$$YLD = DW \cdot AF_{tot} \cdot C_o \qquad (5)$$

In principle, YLD for cardiometabolic outcomes could also be obtained from Equation 5 by multiplying the disability weight (DW) with the AF_{tot} and with the number of observed cases (C_o) in each country. However, it should be noted that for cardiometabolic outcomes, no DW in relation to noise effects have been derived. The YLD for CVD and diabetes type 2 per each country is thus extracted directly from the GBD study by multiply the YLD of the disease with the population attributable fraction of noise (AF_{tot}). The disease specific GBD value of YLD already includes the DW.

Likewise, YLL are calculated using the 2019 GBD study. YLL of all-natural cause mortality per country (YLL_{nat}) is derived from the GBD study and is multiplied by the population attributable fraction for noise (AF_{tot}).

$$YLL = YLL_{nat} \cdot AF_{tot} \qquad (6)$$

4.2. Country specific health data

For %HA, %HSD, and cognitive impairment, ERFs provide directly the corresponding prevalence for each noise exposure categories, and no country specific data is needed. This calculation will be conducted for the adult population starting at 18 years old. The demographic information at country level is outlined in Annex 3 and refers to the 2021 population data from Eurostat.

The impairment of reading in children due to aircraft noise is to be conducted with school aged children from 7 to 17 years old inclusive, using demographic information at country level from Eurostat.

For cardiovascular disease, diabetes, and mortality the ERFs relate to relative risk and attributable fraction is obtained as described in Section 4.1. For the calculation of incident cases, YLD and YLL, country specific health data from the Global Burden of Disease Study (IHME, 2023) are to be used (i.e. disease rates, YLD per disease and YLL for all-cause natural mortality). Latest available data to be used for the HRA from GBD 2019 (IHME, 2023) are outlined in Annex 4. Despite ERFs are based on adults, the all-ages category (i.e. incidence rate from GBD) is used for the calculation of cardiometabolic diseases and mortality since cardiometabolic diseases and mortality at younger ages is very small or absent. The resulting number of incident cases is multiplied by the fraction of adult population at country level from Eurostat.

4.3. Disability weights

To calculate the YLD for HA and HSD the prevalence rates of disease need to be multiplied by the disease-specific disability weights. The disability weights (DWs) to be used are those recommended by the WHO and are outlined in Table 4.1. Disability weights from other sources may also considered for sensitivity analysis. Similarly, the prevalence of reading comprehension impairment due to noise in children is multiplied by the DW specified in Table 4.1. For the estimation of YLD for the cardiovascular diseases and diabetes, the specific country value estimates are extracted from the GBD 2019 (IHME, 2023). The YLD values extracted from each country due to the specific diseases i.e. CVD and diabetes type 2, are calculated with the relevant DWs outlined in the GBD Study 2019 (IHME, 2023).

Table 4.1: Overview of disability weights

Health condition	Disability weight (i.e. severity on health)[a]
Long-term high annoyance	0.011 (Charalampous et al., 2024; WHO Europe, 2024)
Long-term high sleep disturbance	0.010 (Charalampous et al., 2024; WHO Europe, 2024)
Cardiovascular Disease	DALYs of CVD outcomes in GBD Study 2019 https://ghdx.healthdata.org/record/ihme-data/gbd-2019-disability-weights
Diabetes	DALYs of Diabetes Type 2 in GBD Study 2019 https://ghdx.healthdata.org/record/ihme-data/gbd-2019-disability-weights
Reading comprehension	0.006 (WHO Europe, 2018)

The DW for long-term high annoyance and high sleep disturbance are based on Charalampous et al. (2024) and WHO Europe (2024). The values come from an empirical evaluation of disability weights in the context of noise research.

The work by Charalampous et al. (2024) on the update of noise DW, is based on an online survey among a cohort of individuals from Hungary, Italy, Swede and the Netherlands. It included paired comparison questions on different health states. Different severity levels for annoyance i.e. moderate and severe, are included. For sleep disturbance there are no different severity levels, only the distinction between sleep disturbance with and without environmental noise as the source.

For this study the DW for severe annoyance (i.e. 0.011) is used since our health outcome is "high annoyance". For sleep disturbance the DW defined as "Sleep disturbance with environmental noise as the source" with a value of 0.01 is used.

The study from Charalampous et al. (2024) also includes DWs for many other health outcomes that are associated with exposure to environmental noise. However, these values are not used to calculate the Burden of Disease (BoD) for the other health outcomes included in the EU-wide HRA (i.e. CVD outcomes, Diabetes Type 2 and mortality). The DWs for these outcomes are based on the YLD or YLL rates from the GBD 2019 study.

4.4. Monetization

Based on literature on monetization, an incident disease or a death causes three components of costs: direct, indirect and intangible components. Direct Health Costs include medical resources such as consultations, drugs, in-patient and out-patient hospitalizations, emergency room visits and cost of rehabilitation. Further non-medical costs are related to the health outcome (e.g. home care, transportation, and major home modifications) or death outcomes (e.g. funeral).

Indirect costs are related to the resource lost such loss of production by the patient (lower productivity, absence from work, early retirement or premature death), or by the patient's relatives (e.g., a parent taking time off work). The direct and indirect costs are tangible and can be derived from market costs in terms of national wealth and Gross Domestic Product (GDP).

The last component is the intangible (or social) costs (IC) which includes grief, fear, pain, unhappiness, loss of well-being, and loss of quality of life. They apply to the patient but also to their social network. There is no consensus on the best way to account for intangible costs. One approach to monetize the intangible cost is the willingness to pay approach. Several methods are proposed to derive corresponding cost components.

A challenge for the monetization of environmental noise is the broad spectrum of outcomes where noise affects health and other aspects related to wellbeing. The direct and indirect costs may differ considerably between different outcomes and are a particular challenge for outcomes such as %HA or %HSD. A common approach to quantify the costs for these latter outcomes are hedonic pricing approach based on housing and renting costs in relation to noise exposure and controlled for other outcomes.

In terms of quantifying the economic cost of early mortality, the value of a statistical life (VSL) and the value of a life year (VOLY) are most common approaches (Bayat et al., 2019). Both approaches assign monetary values to specific life spans lost or gained. VSL is typically based on observed or inferred individual willingness to pay (WTP) for small reductions in the risk of dying. The VOLY is derived from the willingness to pay for increasing life expectancy by one additional year; in some cases, the VOLY is also scaled by gross domestic product (GDP) per capita.

In the EC "handbook on the external costs of transport" (European Commission et al., 2020) external costs of transportation noise are calculated by attributing costs for each noise exposed person by determining two components of costs: costs of annoyance and costs of health.

The costs of annoyance are based on a stated preference approach, where questionnaires or experiments are used to obtain the willingness to pay to avoid the damage of the externality (Bristow et al., 2015). The health costs are based on a quantification of DALYs (or QALYs, respectively) in terms of noise induced diseases adjusted to also incorporate the medical costs (Department for Environment, Food & Rural Affairs, 2014).

The proposed approach of the EC handbook is simple and appealing since it only needs an input on noise exposure distribution data of the target population. However, health costs evaluation is based on previous evidence rating and does not include all outcomes to be considered relevant in this report.

Thus, for monetization of the noise induced health costs, we will follow the same approach as in EEA (2020a) by attributing a monetary value for each V_{DALY} multiplied with the DALY for each outcome (i): $DALY_i$ as following.

$$Costs = \sum_{i=1}^{n} DALY_i \cdot V_{DALY} \qquad (7)$$

There is no official consensus regarding the appropriate value of a DALY/QALY. The most common approach is the conversion based on the VOLY, where different economic values of DALYs are allocated per income country group level as defined by the World Bank.

The OECD report "Mortality Risk Valuation in Environment, Health and Transport Policies" (OECD, 2012) cites the European Commission 2009 Impact Assessment Guidelines. According to this document a number of different approaches to valuation are discussed, and suggests using the methodology that is appropriate to the circumstances. The Guidelines indicate, however, that the VSL has been estimated at EUR 1-2 million in the past (no year indicated) and EUR 50,000 – EUR 100,000 for VOLY, and suggest that these range are used "if no more context specific estimates are available" (European Commission, 2009, Annexes, p. 43).

For the last HRA of the environmental noise burden by the EEA a value of EUR 78 500 was used to calculate the economic costs of the health impacts of noise (Torfs, 2003). In other literature reviews on various studies aiming at monetization of environmental health burdens or health care interventions similar values were found (e.g. Daroudi et al., 2021; Grandjean and Bellanger, 2017; Sanchez et al., 2014; IGCB(N), 2010). Within the 2020 EC report "Handbook on the external costs of transport" (European Commission et al., 2020) a literature showing different VOLY values from different studies as well as adjustments for the year 2016 was presented. Based on the literature a VOLY of 70,000 EUR (for the year 2016) was used. Therefore, based on this list of studies it is proposed to use a value of EUR 70,000 as a monetary cost per DALY.

5. Discussion

5.1. Changes in methods since the last HRA

Whereas the general approach remained the same as in the EEA (2020a) assessment, we updated the methods to reflect new insights in noise health risk research. Specifically, new outcomes (all-cause mortality, all cardiovascular disease and diabetes) with new exposure-response functions are considered. The threshold for the quantification of negative health impacts is proposed to be reduced to L_{den} of 45 dB as new evidence shows effects at these levels. Of note, the risk increase per 10 dB transportation noise for cardiovascular disease incidence is smaller than previously used for ischaemic heart disease. This reflects a consistent pattern of noise research from the last 10 years. New studies indicate associations of noise with all types of cardiovascular diseases, not only IHD. Further, new studies with high-quality exposure models tend to find associations with noise at lower levels (i.e. below 55 dB L_{den}). However, the risk increase per 10 dB in most recent studies is lower than previously observed. Possibly this are direct consequence of the use of better exposure assessments in the low dose range. Older studies with a high cut-off for the reference group (a.g. <55 dB), may have actually included in the reference group people with low exposure that resulted in an overestimate of the regression slope.

Even if the risk increase to be used in future noise EEA assessments is lower than used previously, the inclusion of multiple cardiovascular disease outcomes together (as opposed to using only IHD) is expected to result in a significant increase on the number of incident cases as well as on the burden of disease morbidity due to noise compared to the estimates presented in the Noise in Europe report of 2020. In addition to this, mortality estimates will also be significantly higher due to the change from mortality due to IHD to all-cause mortality.

5.2. Strengths and limitations of the assessment of noise exposure below the END thresholds

In order to estimate the health risks due to environmental noise in the European area, an extrapolation of the number of people affected by noise at levels below the END thresholds is needed in those areas where strategic noise maps are produced (agglomerations above 100.000 inhabitants, major roads with more than 3 million vehicles per year, major railways with more than 30.000 train passages per year and major airports with more than 50.000 movements per year).

In order to estimate the exposure distribution to lower levels in 5 dB bands, we used the methodology implemented by Houthuijs et al. (2018). There were no newer studies found in the literature to update or improve the current methodology at European scale.

Current data reported for 2022 reference year is still scarce. For two noise sources (major airport exposure outside agglomerations and industrial noise inside agglomerations), a review of the percentages of people exposed above the END thresholds will be needed to confirm that the applied method is still valid.

An improvement for transferring exposure distributions from 5 dB to 1 dB has been proposed. This improvement avoids abrupt changes of exposure estimations within 1 dB differences, and results in a smoother distribution of people exposed from 25 dB to 79 dB.

5.3. Strengths and limitations of Umbrella+ review

We conducted an Umbrella+ review for future EEA EU-wide assessments on the health risks of noise. A systematic review for all possible outcomes was beyond our capacity and thus the Umbrella+ review was a good compromise that allowed us to capture the most up to date literature relevant to future

HRA. However, this type of literature review has some limitations. We relied partly on evidence rating from other authors, which may have resulted in some variability of the criteria related to the evaluation of certainty of evidence. It should also be noted that for most recent studies, we only included high-quality studies whereas the reviews that served as starting point may have been more inclusive. Usually, prospective cohort studies are considered the most reliable study design specially for incident diseases which may be also fatal. However, for prevalent diseases such as hypertension, behaviour and cognitive problems, prospective cohort studies may have some limitations, since people may have developed a noise induced disease prior to baseline investigation. In general, noise exposure is not markedly changing from year to year and thus in a steady-state situation cohort studies may not capture adequately changes in disease rate. For this reason, case-control and cross-sectional studies not included in this review could contribute substantially to the existing evidence on specific outcomes related to prevalence.

Another challenge in Umbrella+ reviews is to deal with multiple cohorts from the same country. We cannot completely ensure that the same person is not part of multiple cohorts, and thus some may be entered multiple times in our meta-analysis. For instance there are several studies representing different cohorts in Denmark, including a new nation-wide cohort, that were included on the basis they had different follow-up periods and/or confounder control (Jørgensen et al., 2019; Roswall et al., 2018; Thacher et al., 2021a). However, overall the proportion of potential double counting is small and would mostly result in a slight underestimation of the precision but not affect the point estimate.

Separate meta-analyses are conducted for road traffic, railway and aircraft noise. Effect estimates were sometimes quite different for the three sources of transportation noise. Since the characteristic and the diurnal pattern of noise exposure from different sources varies, it is, in principle, plausible that this translates into differences in the effect estimates that are related to L_{den}. However, the number of studies for railway and in part also for aircraft noise were mostly scarce, and observed heterogeneity may be mainly introduced by different methods, e.g. the precision of noise exposure assessment. Relatively few people are exposed to railway noise and aircraft noise and thus the power of these studies is often lower than for road traffic noise studies. Since road traffic noise is much more common it is well possible that in these studies moderate levels of railway noise were masked with road traffic noise, and this may be another reason why these studies have sometimes failed to observe an association. There was consistent high-quality evidence for relationships between road traffic noise and cardiovascular health outcomes, mortality and diabetes. Therefore, we propose to use these relationships for estimating the health risks of road, railway and aircraft noise. It is assumed that the cardiometabolic effects of road traffic noise can be extrapolated to aircraft and railway noise given that the biological mechanisms involved are similar.

5.4. Considerations for the application of the noise HRA

Tackling multiple exposures and multiple outcomes in a noise HRA

In situations where several outcomes are considered in the same HRA, the question arises whether the effects are cumulative or even multiplicative. The ENG concluded that in terms of the proposed outcomes and noise sources the effects are cumulative (WHO Europe, 2018). Evidence for this, for example, is presented by Seidler et al. (2019), who stated that for cardiovascular diseases and depression the multiplication of epidemiological risks seems to provide a better estimate of the health risks of combined traffic noise exposures compared to energetic addition. Further, several studies have estimated several transportation noise sources concurrently and thus by mutual adjusting effect estimates are cumulative. In principle, impacts should not be summed up for studies that only considered one noise source, if exposure sources are highly correlated. However, given the high spatial resolution, correlation between road, rail and aircraft noise is usually low according to high-quality models (Vienneau et al., 2022). This implies that considering the cumulative effect of all noise sources is justified.

For interpretation, it needs to be considered that the same person may be annoyed from more than one source or may be annoyed and concurrently develop a disease from noise exposure. Thus, simply adding up the number of highly affected people may be misleading if not made transparent. In terms of DALYs, however, it is not critical whether this risk increase accumulates in the same person or in different persons. This justifies to sum up DALYs from different outcomes.

Implications of current EU exposure assessment in noise HRA

Several calculation methods and approaches were used to produce noise maps or for deriving the noise exposure distribution of the population. Systematic differences were due to a variety in the quality of input data such as traffic counts or the implementation of buildings in the model (Garg and Maji, 2014; Morley et al., 2015; Murphy and Douglas, 2018; Nijland and Van Wee, 2005; Vienneau et al., 2019b). In addition to this, in previous reporting rounds of noise mapping of the END, there was no common method for noise mapping in place. Therefore, countries used their own national noise assessment methods or the interim methods indicated in the END.

In order to harmonise noise mapping assessments and increase comparability across countries, the EU developed a common method for noise mapping i.e. CNOSSOS, that is to be employed for the calculations of new strategic noise maps (Commission Directive (EU) 2015/996 of 19 May 2015 establishing common noise assessment methods to Directive 2002/49/EC of the Parliament and of the Council).

The strategic noise maps reported under the END in 2022 will be the first ones that have been produced using the new CNOSSOS methodology. Therefore, the variability in exposure estimations across countries will need to be evaluated. Potential discrepancies may be related to i.e. decisions by individual cities on which roads should be included as input data in the calculation of strategic noise maps. For instance, some cities map all roads while others only map the busiest roads (EEA, 2020a). In cities where only the busiest roads are mapped an underestimation in the number of people exposed to noise is expected. Other discrepancies may be related to input data used in the prediction method or the criteria to attribute number of people in buildings to estimate exposure values at different dB levels.

Therefore, some heterogeneity could be still expected, which could affect the comparison between countries. Some new studies from Nordic Countries indicate that health effects are more prominent at the least exposed façade than at the maximum exposed facade (Thacher et al., 2022a, 2021b). It is argued that sleeping rooms are mostly facing the quiet side of the house and thus that noise exposure at the quiet side is most relevant night-time exposure (Klompmaker and Laden, 2021). Such aspects are not yet considered in the proposed HRA but may be an important avenue for the future, which also has implication for noise regulation.

Uncertainties from baseline health data, exposure-response functions and disability weights used in the noise HRA

Baseline health statistics, such as incidence of and mortality rates per country, are used to estimate the number of cases of cardiovascular disease, diabetes and the number of premature deaths attributable to noise per year. In addition, in order to estimate health risks expressed in terms of disability-adjusted life-years (DALYs) the use of disability weights as well as data on years of life lost and years lived with disability for the selected outcomes per country are needed. These data are used from the Institute for Health Metrics and Evaluation (see Annex 4). Baseline morbidity and mortality data are based on national statistics. Therefore, using national health data for other sub-national units (e.g. urban areas) may bring about uncertainties, as health baseline data may not apply to the territory uniformly. Therefore, it is recommended to use these values at an aggregated level, either EU level or country level.

Possible underestimation and overestimation

The proposed methods aim to estimate the burden of disease from noise as accurate as possible. In all calculation steps there is substantial uncertainty involved. For instance, for some people exposure will be overestimated and for some underestimated. However, without any systematic error, these errors will compensate each other when estimating the burden of disease per countries.

Nevertheless, there are some critical factors which may produce and under or overestimation of the burden of disease. Main aspects that may produce an overestimation are residual confounding in the original studies, which may result in overestimated relative risks. Similarly, publication bias could produce some overestimation of the pooled effect estimates and corresponding burden of disease estimate. On the other hand, non-differential exposure misclassification in the original studies is expected to be substantial, translating into an underestimation of the burden of disease. Further, given limited research for many outcomes, it is also conceivable that we did not include all relevant outcomes. One additional relevant source for underestimation is the fact that under the END not all sources, urban areas, roads, railways and airports across Europe are covered. Therefore, the HRA will underestimate the part of the territory that is not covered in the EU exposure assessment. Overall, it is thus more likely that the future HRA will underestimate rather than overestimate the burden of disease from noise.

6. Conclusions

Based on the meta-analyses from PART II of this report as well as previous studies on noise BoD and expert judgement, we suggest a set of health outcomes to be quantified in a noise HRA at European level. In the HRA only those health outcomes that have demonstrated a reasonable causal relationship between noise exposure and negative health effects are proposed. Long-term high annoyance and long-term high sleep disturbance for road, rail and aircraft are suggested to be quantified using the relationships described in the WHO Environmental Noise Guidelines for the European Region. In the case of industrial noise, which was outside of the WHO guidelines, we propose to use relationships from older studies that refer to industrial noise and %HA as well as %HSD. Due to many recent studies showing the negative effects of transport noise on many different cardiovascular outcomes, we suggest pooling together a relationship for cardiovascular outcomes. In addition, we suggest quantifying the incidence of diabetes in a noise HRA. Based on the meta-analyses we suggest using all-cause mortality instead of specific disease mortality as done in the previous EEA HRA. Most evidence comes from studies looking at the health impacts of road traffic noise, which showed mostly higher certainty of evidence than studies for railway and aircraft noise. For cardiometabolic and mortality outcomes, is not possible to use specific-source curves due to limited evidence for railway and aircraft noise. Therefore, it is proposed to use the relationships for road to estimate the health impacts of rail and aircraft traffic noise based on the assumption that biological mechanisms are thought to be similar. Cognitive impairment in children is proposed to be assessed only for aircraft noise and specifically for reading impairment.

The body of evidence shows negative effects due to transport noise at much lower levels that those captured in the END exposure assessments (i.e. 55 dB L_{den}, and 50 dB L_{night}). We therefore suggest to assess the health risks of noise at levels of 45 dB L_{den} and 40 dB L_{night}). A method for estimating the number of people exposed to noise levels below the END thresholds is described in PART I.

The relationships and DWs used to estimate the burden of disease attributable to transportation noise for the adult population in Europe in 2022 are described in Table 4.1. The proposed source for calculating health impacts at country level is the 2019 GBD study. Regarding quantification of economic costs of health risks at EU level, we proposed to use a monetary value for a DALY of 70,000 EUR.

List of abbreviations

Abbreviation	Name	Reference
ADHD	Attention deficit/hyperactivity disorder	
BoD	Burden of Disease	
BMI	Body Mass Index	
CI	Confidence interval	
CNOSSOS	Common Noise Assessment Methods according to Directive 2002/49/EC	https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32015L0996
DALYs	Disability-adjusted life years	
DDCH	Danish Diet, Cancer and Health cohort	
DEBATS	Discussion on the health effect of aircraft noise	
DNC	Danish Nurse Cohort	
DW	Disability weights	
EEA	European Environment Agency	https://www.eea.europa.eu/en
EIONET	European Environment Information and Observation Network	https://www.eionet.europa.eu/
END	Environmental Noise Directive	https://environment.ec.europa.eu/topics/noise/environmental-noise-directive_en
ENG	Environmental Noise Guidelines	
ERF	Exposure-response function	
ETC/ACM	European Topic Centre on Air pollution and Climate change mitigation	https://www.eea.europa.eu/data-and-maps/data-providers-and-partners/air-pollution-and-climate-change
EU	European Union	https://european-union.europa.eu/index_en
GBD	Global Burden of Disease	
GRADE	Grading of Recommendations Assessment, Development and Evaluation	
HA	High noise annoyance	
%HA	Percentage of people highly annoyed	
HRA	Health risk assessment	
ICBEN	International Commission on Biological Effects of Noise	https://www.icben.org/
IHD	Ischaemic heart disease	
IHME	Institute For Health Metrics and Evaluation	https://www.healthdata.org/
MI	Myocardial infarction	
OR	Odds ratio	
PAF	Population attributable fraction	
PECOS	Population, Exposures, Comparators, Outcomes and Study design	
PRISMA	Preferred Reporting Items for Systematic reviews and Meta-Analyses	
RR	Relative risk	
SD	High sleep disturbance	

Abbreviation	Name	Reference
%HSD	Percentage of people highly sleep disturbed	
SDQ	Strength and Difficulties Questionnaire	
SOER	State of the Environment Report	
VOLY	Monetary Value of a Life Year	
WHO	World Health Organization	https://www.who.int/
WHO ENG	Environmental Noise Guidelines from the World Health Organization	https://www.who.int/europe/publications/i/item/9789289053563
YLD	Years of healthy life lost due to disability	
YLL	Years of life lost	

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Yuchi, W., et al., 2020, 'Road proximity, air pollution, noise, green space and neurologic disease incidence: a population-based cohort study', *Environmental Health: A Global Access Science Source* 19(1), p. 8 (DOI: 10.1186/s12940-020-0565-4).

Yuchi, W., et al., 2022, 'Neighborhood environmental exposures and incidence of attention deficit/hyperactivity disorder: A population-based cohort study', *Environment International* 161, p. 107120 (DOI: 10.1016/j.envint.2022.107120).

Zaman, M., et al., 2022, 'Environmental noise-induced cardiovascular, metabolic and mental health disorders: a brief review', *Environmental Science and Pollution Research International* 29(51), pp. 76485-76500 (DOI: 10.1007/s11356-022-22351-y).

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7. Annex 1 - Search terms used to characterize population, exposure, outcomes and study type for the Umbrella+ review

Table 7.1: Search terms used for the Umbrella + review

PECOS criteria	Description	Query
Population	General population /adults	"humans"[MeSH Terms] OR "adult"[tiab] OR "adults"[tiab] OR "adult"[MeSH Terms] OR "aged"[tiab] OR "aged"[MeSH Terms] OR "man"[tiab] OR "men"[tiab] OR "woman"[tiab] OR "women"[tiab]
Population	Infants / children / adolescents	"Child"[tiab] OR "children"[tiab] OR "pupils"[tiab] OR "preschooler"[tiab] OR "preschoolers"[tiab] OR "student"[tiab] OR "students"[tiab] OR "Adolescent"[tiab] OR "adolescents"[tiab] OR "Infant"[tiab] OR "infants"[tiab] OR "toddler"[tiab] OR "toddlers"[tiab] OR "newborn"[tiab] OR "baby"[tiab] OR "babies"[tiab] OR "boy"[tiab] OR "boys"[tiab] OR "girl"[tiab] OR "girls"[tiab] OR "postnatal*" [tiab] OR "post-natal" [tiab] OR "*school*" [tiab] OR "pediatric*" [tiab] OR "paediatric*" [tiab] OR "prenatal"[tiab] OR "preterm"[tiab] OR "birth"[tiab] OR "gestational"[tiab] OR "pregnancy"[tiab] OR "fetal"[tiab] OR "parturition"[MeSH Terms] OR "Adolescent"[MeSH Terms] OR "Birth Cohort"[MeSH Terms] OR "Child"[MeSH Terms] OR "Infant"[MeSH Terms]
Population	Unborn children	"foetus"[tiab] OR "fetus"[tiab] OR "embryo"[tiab] OR "unborn"[tiab] OR "pregnan*"[tiab]
Exposure	Noise	"Noise, Transportation" [Mesh terms] OR (noise [tiab] AND traffic[tiab]) OR (noise [tiab] AND transportation [tiab]) OR (noise [tiab] AND road [tiab]) OR (noise [tiab] AND (airplane [tiab] OR aircraft [tiab])) OR (noise [tiab] AND rail* [tiab]) OR (noise[tiab] AND environmental[tiab]) OR (noise[tiab] AND community[tiab])
Outcome	All-cause mortality	Mortality [tiab] OR death [tiab]
Outcome	Cardiovascular diseases	Vascular Diseases [Mesh Terms] OR blood pressure [Mesh Terms] OR heart rate [Mesh Terms] OR vascular Diseases [tiab] OR blood pressure [tiab] OR heart rate [tiab] OR pulse [tiab] OR “hypertension” [tiab] OR hypertension [Mesh Terms] OR ischemic heart disease [tiab] OR ischemic heart disease [Mesh Terms] OR stroke [tiab] OR stroke [Mesh Terms] OR CVD [tiab] OR "Heart Failure"[Mesh Terms] OR heart failure [tiab] OR "Myocardial Infarction"[Mesh Terms] OR myocardial infarction[tiab] OR "Coronary Disease"[Mesh Terms] OR coronary disease[tiab] OR "Arrhythmias, Cardiac"[Mesh] OR Arrhythmia[tiab]
Outcome	Mental health problems	"Depression"[Mesh Terms] OR depress*[tiab] OR "Depressive Disorder"[Mesh Terms] OR "Mental Health"[Mesh Terms] OR "Mental Disorders"[Mesh Terms] OR mental health[tiab]
Outcome	Diabetes	See below
Outcome	Cognitive impairment	“Cognitive Psychology” [Mesh terms] OR “Neuropsychological Tests” [Mesh terms] OR “Cognitive Neuroscience” [Mesh terms] OR Cogniti* [tiab] OR “Cognitive functions” [tiab] OR learning [tiab] OR reading [tiab] OR comprehen* [tiab]
Outcome	Dementia	"Dementia"[Mesh] OR "Alzheimer Disease"[Mesh] OR Dement*[tiab] OR Alzheimer Disease[tiab]

PECOS criteria	Description	Query
Outcome	Behavioural issues	Behaviour [MeSH Terms] OR behaviour [Title/Abstract] OR hyperactivity [MESH Terms] OR hyperactivity [tiab] OR aggression [Mesh Terms] OR aggression [tiab]
Outcome	Overweight	Overweight [Mesh Terms] OR overweight [tiab] OR BMI [tiab] OR body mass index [tiab] OR waist circumference [tiab] OR BMI [Mesh Terms] OR body mass index [Mesh Terms] OR waist circumference [Mesh Terms]
Study type	Reviews	"systematic review"[tiab] OR "metaanalysis"[tiab] OR "meta analysis"[tiab] OR "systematic review"[Publication Type] OR "meta analysis"[Publication Type] OR "Meta-Analysis as Topic"[MeSH Terms] OR review[tiab] OR"Review" [Publication Type]
Study type	Original studies	"Cohort Studies"[Mesh Terms] OR cohort[tiab]
Study type	Original studies	"Cross-Sectional Studies"[Mesh] OR cross-sectional[tiab]
Search period [a]		(2015/01/01:JJJJ/MM/DD[dp])

[a] JJJJ/MM/DD = date of search

Search terms within each search grid category were expanded with "OR" and the different categories combined with "AND".

Table 7.2: Search terms used for the Umbrella + review of Diabetes

#	String	Hits
1	((rail* or aircraft or airport* or road* or traffic* or automobile* or vehicle*) adj noise).tw.	1,557
2	exp *Transportation/	50,857
3	Aircraft/ or Airports/ or Railroads/ or Motor Vehicles/	19,414
4	*Noise/	13,303
5	Noise, transportation/	1,673
6	(1 or 2 or 3) and (1 or 4 or 5)	2,447
7	exp Diabetes Mellitus/	501,727
8	exp Obesity/ or exp Overweight/ or exp Body Mass Index/	350,912
9	(diabetes or obesit* or overweight or bmi or body mass index).tw.	911,145
10	7 or 8 or 9	1,119,340
11	6 and 10	119
12	11 not child*.ti.	110
13	limit 12 to yr="2014-2019"	54
14	limit 12 to yr="2014-current"	84
15	limit 12 to yr="2019-current"	40

8. Annex 2 - PRISMA flow diagrams of the screening process to identify relevant reviews and cohort studies

Figure 8.1: PRISMA flow diagram for all-cause mortality.

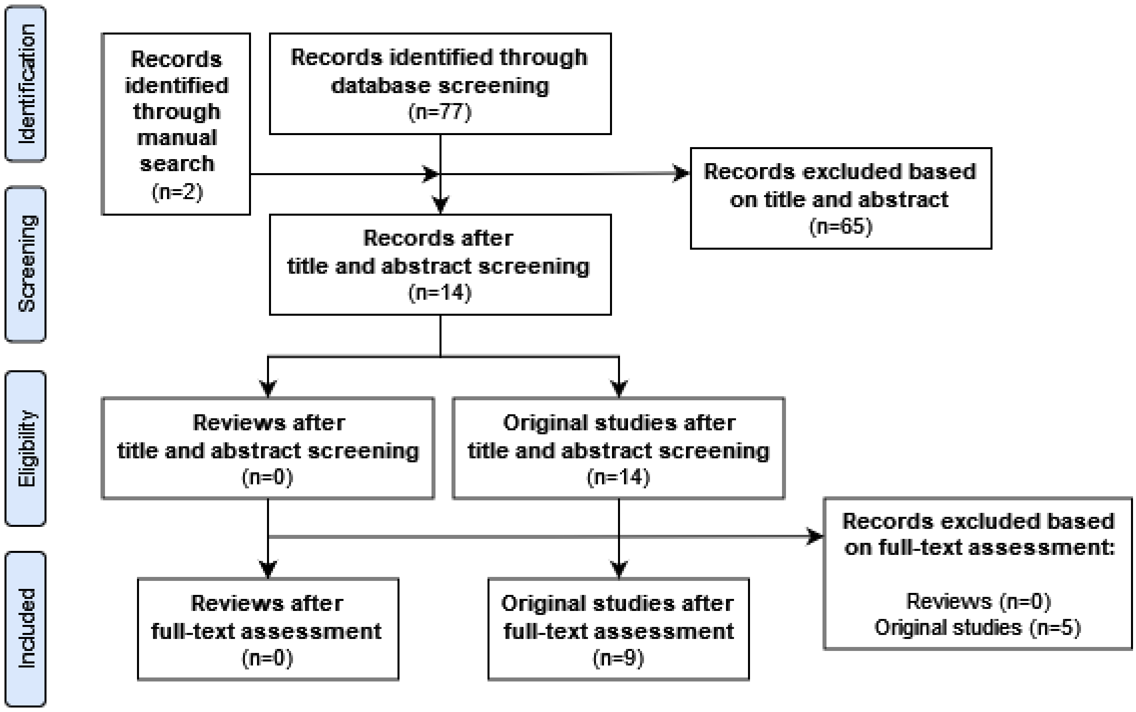


Figure 8.2: PRISMA flow diagram for cardiovascular diseases.

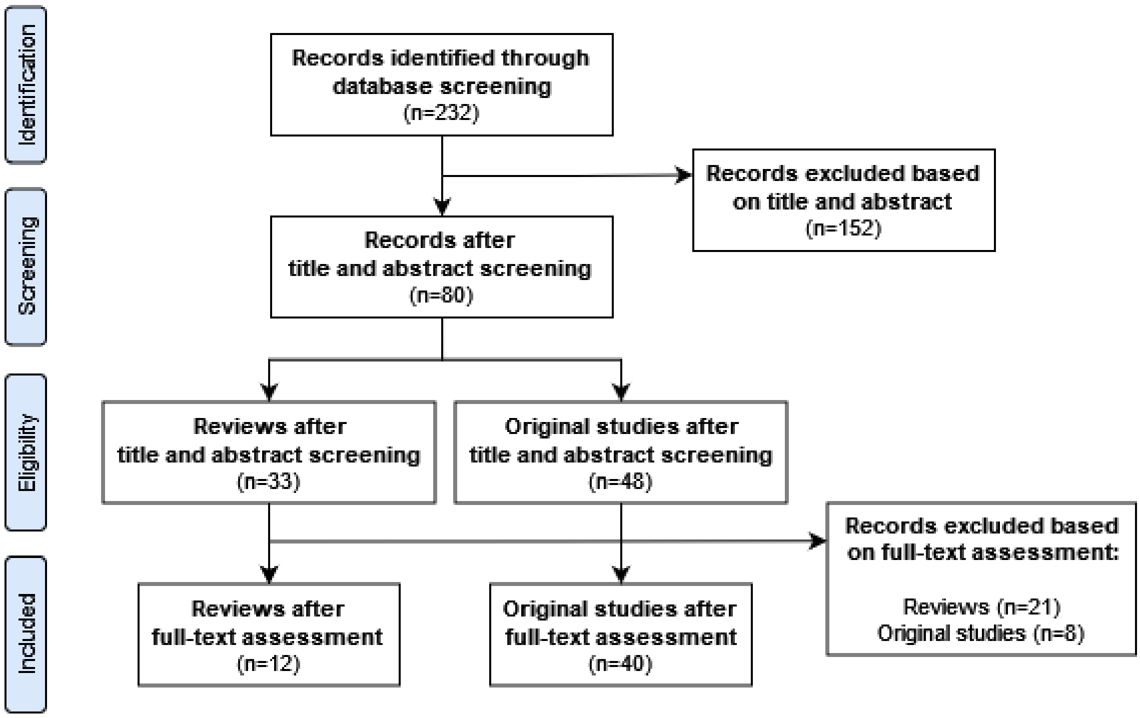
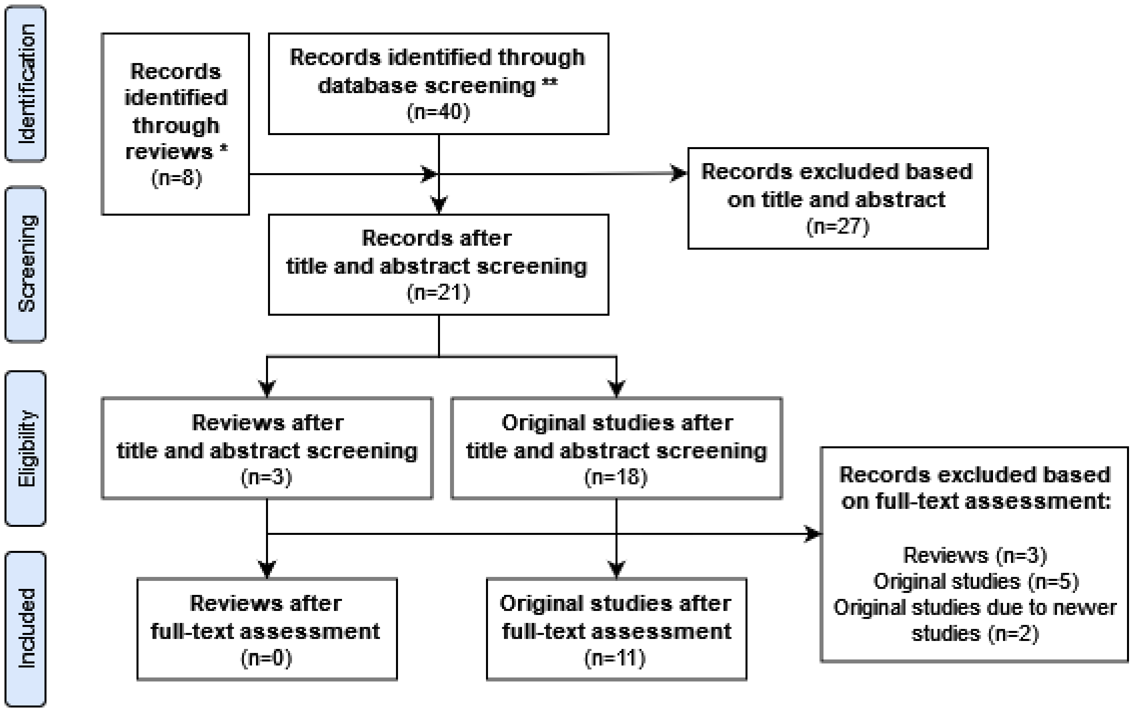


Figure 8.3: PRISMA flow diagram for diabetes.



* Van Kempen et al. (2018), Zare Sakhvidi et al. (2018b) and Vienneau et al. (2019a) covered search period for the WHO ENG until 2019.

** Search conducted 1 Jan 2019 to 03 May 2023.

Figure 8.4: PRISMA flow diagram for mental health.

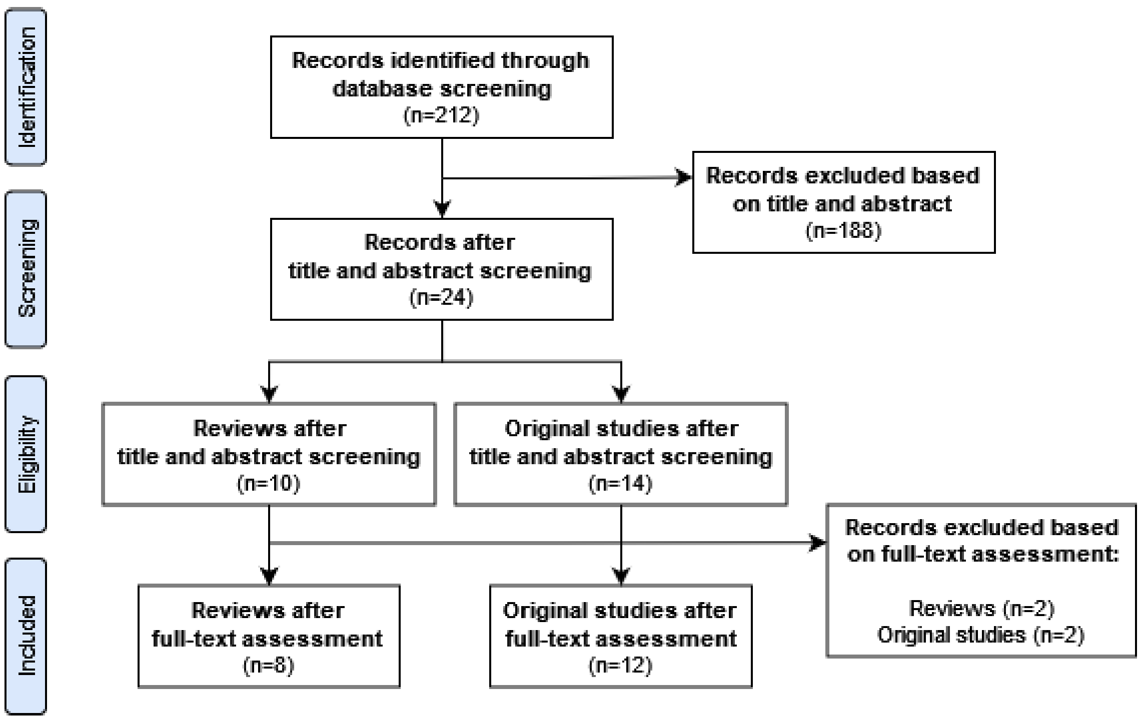


Figure 8.5: PRISMA flow diagram for cognition.

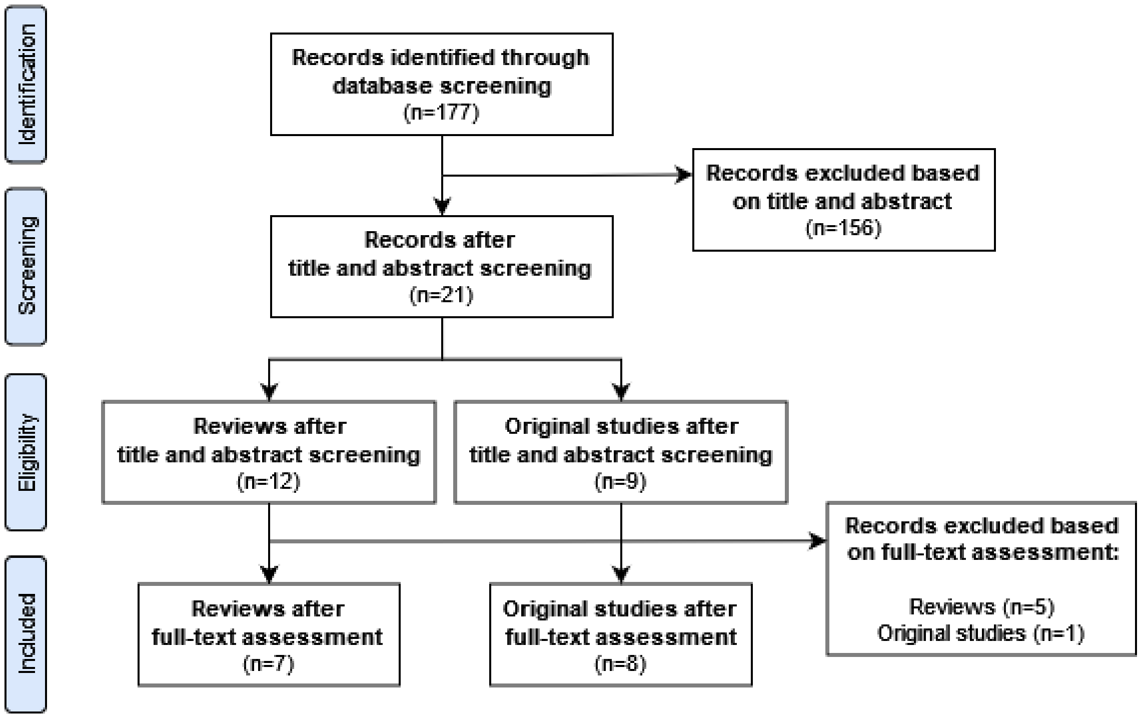


Figure 8.6: PRISMA flow diagram for dementia.

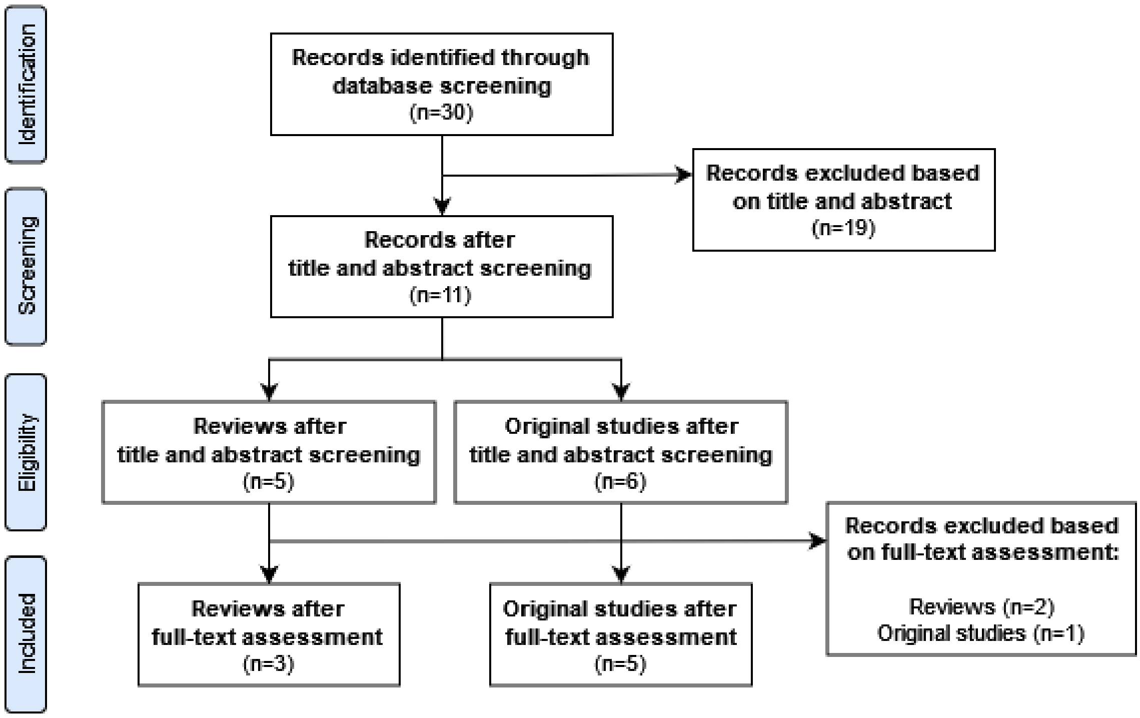


Figure 8.7: PRISMA flow diagram for behaviour.

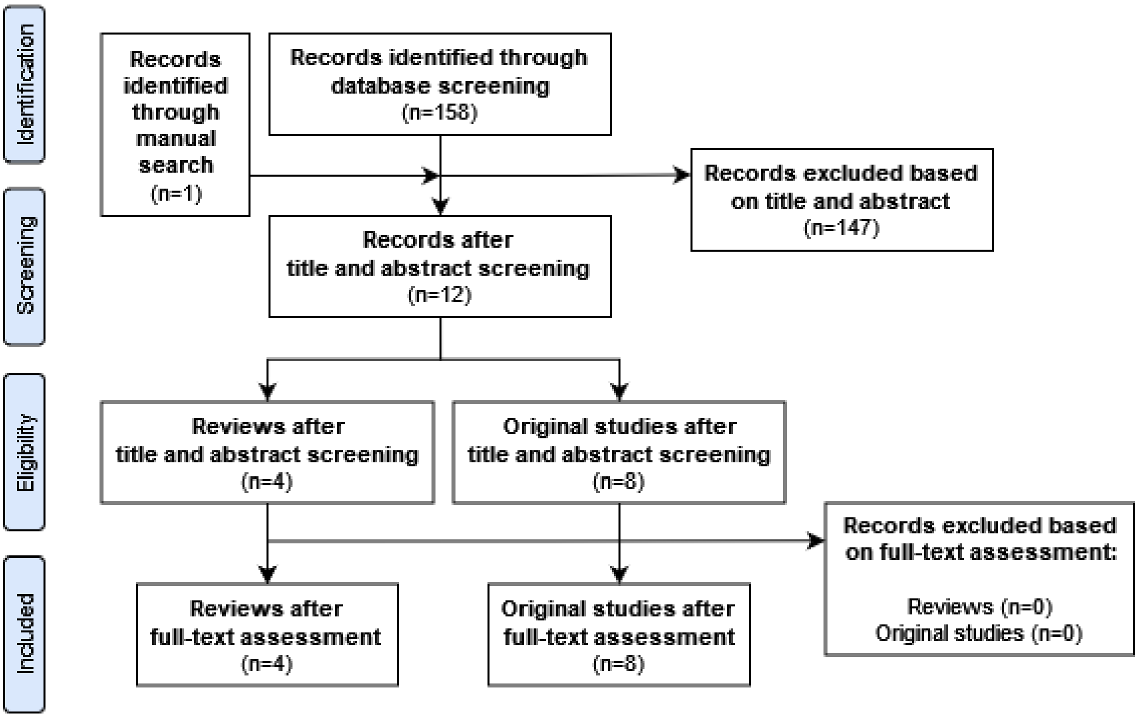


Figure 8.8: PRISMA flow diagram for overweight.

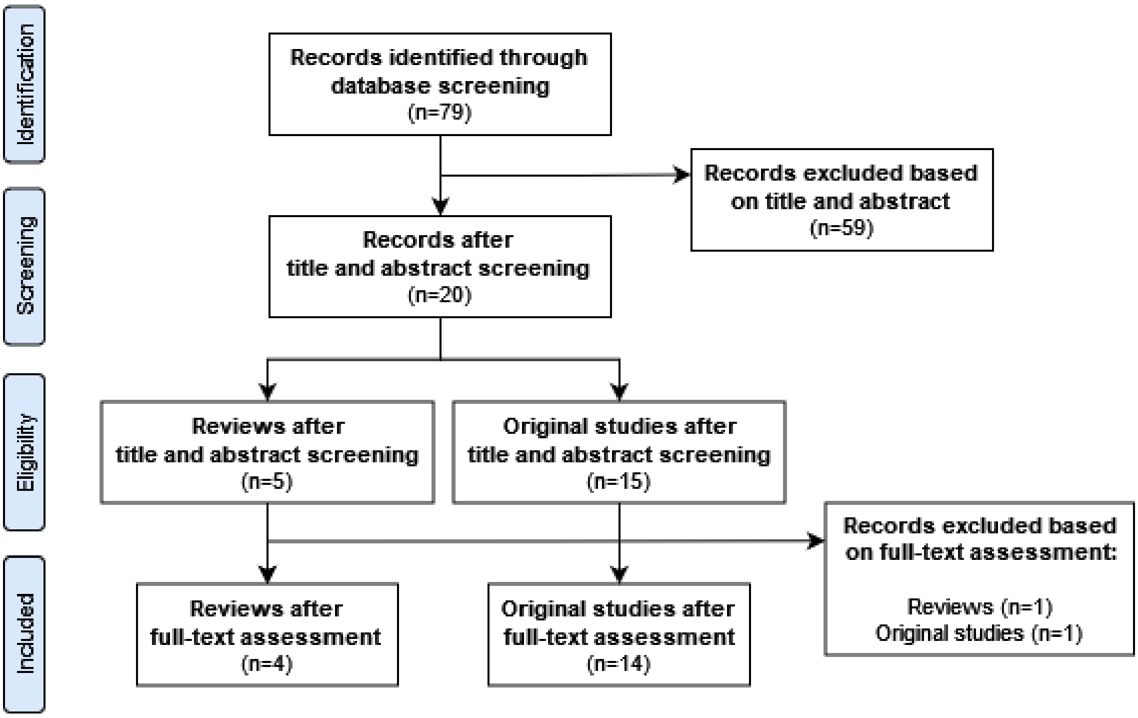


Table 8.1: Excluded Studies for diabetes

Type	#	Excluded Study	Reason
Original studies	1	Sørensen, M., (2013). "Long-term exposure to road traffic noise and incident diabetes: a cohort study." EHP 121(2): 217-222.	Replaced by Roswall 2018 with longer follow-up
	2	Ohlwein, S., (2017). "Road traffic noise and incident diabetes mellitus after 5 years of follow-up–Results from the Heinz Nixdorf Recall Study." Das Gesundheitswesen 79(08/09):V-153.	Replaced by Ohlwein 2019
	3	Klompmaker, J. O., (2019). "Associations of Combined Exposures to Surrounding Green, Air Pollution, and Road Traffic Noise with Cardiometabolic Diseases." EHP 127(8): 87003.	Outcome: diabetes prevalence
	4	Huang, T., (2020). "The Association between Noise Exposure and Metabolic Syndrome: A Longitudinal Cohort Study in Taiwan." IJERPH 17(12).	Outcome: metabolic syndrome
	5	So, R., J. T. (2020). "Long-term exposure to low levels of air pollution and mortality adjusting for road traffic noise: A Danish Nurse Cohort study." Env Int 143: 105983.	Air pollution study
	6	Sorensen, M., (2022). "Air pollution, road traffic noise and lack of greenness and risk of type 2 diabetes: A multi-exposure prospective study covering Denmark." Env Int 170: 107570.	Already include Thacher 2021 with larger population
	7	Sorensen, M., (2023). "Effects of Sociodemographic Characteristics, Comorbidity, and Coexposures on the Association between Air Pollution and Type 2 Diabetes: A Nationwide Cohort Study." EHP 131(2): 27008.	Air pollution study
Review with meta-analysis	1	Liu, C., (2023). "Dose-response association between transportation noise exposure and type 2 diabetes: A systematic review and meta-analysis of prospective cohort studies." Diab/Metab Res Reviews 39(2): e3595.	Quality concerns (e.g. not using latest study within DDCH cohort, missing estimates for SAPALDIA)
	2	Wu, S., (2023). "The association between road traffic noise and type 2 diabetes: a systematic review and meta-analysis of cohort studies." Env Sci Poll Res Int 30(14): 39568-39585.	Quality concerns (e.g. two cases of double counting same study, double counting DDCH cohort)

9. Annex 3 – Country specific demographic data

Table 9.1: Total number and fraction of children and adults per country to be used in HRA.

Country IOS Code	Total children 7-17 years [a]	Total adults ≥ 18 years [b]	Fraction children 7-17 years [a]	Fraction adults ≥ 18 years [b]
AT	936,141	7,388,778	0.104799755	0.827163990
BE	1,464,265	9,233,555	0.126723888	0.799112176
BG	741,697	5,726,002	0.107235141	0.827869914
CH	940,519	7,114,731	0.108475947	0.820586485
CY	104,847	724,531	0.117015827	0.808622031
CZ	1,201,494	8,513,691	0.114484304	0.811226683
DE	8,214,465	69,411,087	0.098784943	0.834719032
DK	724,341	4,687,050	0.124030037	0.802570871
EE	159,513	1,071,841	0.119928455	0.805854287
GR	1,189,218	8,841,684	0.111364265	0.827979089
ES	5,422,575	39,156,568	0.114403466	0.826110677
FI	678,175	4,492,267	0.122551566	0.811788045
FR	9,274,769	53,179,816	0.137085780	0.786024594
HR	436,229	3,344,506	0.108074983	0.828595602
HU	1,050,174	8,024,087	0.107922989	0.824609497
IE	760,790	3,811,534	0.151965794	0.761343852
IS	51,742	286,356	0.140301308	0.776470205
IT	6,140,254	49,885,100	0.103657099	0.842138575
LI	4,241	32,196	0.108590449	0.824375880
LT	298,175	2,297,362	0.106655626	0.821754278
LU	74,008	513,736	0.116597608	0.809377216
LV	216,165	1,534,689	0.114178309	0.810622415
MT	48,999	433,970	0.094940903	0.840864174
NL	2,098,053	14,164,193	0.120057406	0.810521124
NO	706,890	4,279,679	0.131115121	0.793801908
PL	4,264,667	30,917,547	0.112702613	0.817059889
PT	1,094,093	8,596,565	0.106240651	0.834759627

Country IOS Code	Total children 7-17 years [a]	Total adults ≥ 18 years [b]	Fraction children 7-17 years [a]	Fraction adults ≥ 18 years [b]
RO	2,241,916	15,550,331	0.116756352	0.809842971
SE	1,346,417	8,189,892	0.129721431	0.789060529
SI	232,411	1,734,767	0.110200822	0.822563262
SK	618,361	4,431,608	0.113257473	0.811682373
TR	13,946,762	60,863,705	0.166798642	0.727909698

[a] Between 7 and 17 years old (including both 7 and 17)

[b] From 18 years old

Data from Eurostat 2021:
https://ec.europa.eu/eurostat/databrowser/view/demo_pjan__custom_8056910/default/table?lang=en

10. Annex 4 – Country specific health data

Table 10.1: Incidence/mortality rate and YLD/YLL for each health outcomes per country from the 2019 GBD study.

Country IOS Code	All-cause mortality rate	All-cause YLL	IHD Incidence rate	IHD YLD	CVD Incidence rate	CVD YLD	Diabetes Type 2 Incidence rate	Diabetes Type 2 YLD
AT	0.0087674	0.1316422	0.0042335	0.0011051	0.0120707	0.0065827	0.0032021	0.0057290
BE	0.0093212	0.1391104	0.0038088	0.0011312	0.0101796	0.0052410	0.0031021	0.0062423
BG	0.0174096	0.3151997	0.0068656	0.0016574	0.0165589	0.0108028	0.0031021	0.0062423
CH	0.0074679	0.1072588	0.0033905	0.0008224	0.0095092	0.0043332	0.0032844	0.0055253
CY	0.0062163	0.1051628	0.0019063	0.0007401	0.0071569	0.0036821	0.0045452	0.0064284
CZ	0.0101748	0.1699571	0.0069247	0.0019984	0.0144155	0.0087499	0.0063350	0.0141876
DE	0.0108355	0.1631802	0.0054024	0.0011479	0.0126052	0.0064903	0.0050876	0.0086950
DK	0.0091417	0.1469670	0.0039090	0.0007462	0.0112283	0.0045246	0.0027375	0.0040608
EE	0.0115718	0.1965455	0.0121534	0.0016340	0.0195003	0.0072761	0.0023565	0.0056332
GR	0.0120714	0.1727592	0.0043642	0.0011615	0.0120134	0.0058034	0.0032427	0.0070265
ES	0.0089611	0.1285056	0.0036143	0.0008779	0.0099342	0.0049841	0.0042571	0.0081540
FI	0.0095720	0.1446333	0.0053601	0.0012892	0.0129181	0.0063202	0.0040099	0.0078205
FR	0.0084159	0.1243672	0.0034331	0.0008650	0.0095256	0.0053763	0.0019857	0.0029721
HR	0.0116664	0.1910702	0.0050724	0.0017284	0.0129233	0.0077567	0.0043931	0.0095895
HU	0.0127051	0.2282332	0.0064512	0.0016396	0.0151885	0.0088816	0.0043135	0.0095122
IE	0.0063401	0.1049722	0.0029525	0.0007209	0.0080928	0.0037822	0.0028816	0.0043619
IS	0.0058058	0.0910719	0.0034056	0.0008492	0.0085103	0.0040117	0.0028198	0.0046217
IT	0.0102218	0.1433480	ss0.0053031	0.0011863	0.0145804	0.0066603	0.0044941	0.0082556
LI	No data	No data	No data	No data	No data	No data	No data	No data

Country IOS Code	All-cause mortality rate	All-cause YLL	IHD Incidence rate	IHD YLD	CVD Incidence rate	CVD YLD	Diabetes Type 2 Incidence rate	Diabetes Type 2 YLD
LT	0.0128871	0.2241145	0.0092491	0.0017635	0.0178478	0.0085777	0.0018921	0.0045538
LU	0.0062848	0.1017047	0.0023098	0.0009557	0.0080667	0.0049161	0.0047425	0.0071563
LV	0.0135493	0.2376396	0.0092747	0.0019247	0.0182057	0.0088841	0.0024235	0.0059373
MT	0.0082965	0.1321944	0.0030372	0.0013570	0.0102802	0.0055890	0.0046950	0.0082777
NL	0.0086989	0.1360045	0.0049707	0.0009285	0.0112889	0.0048188	0.0026711	0.0048807
NO	0.0072824	0.1092313	0.0031805	0.0010581	0.0102694	0.0057983	0.0033583	0.0056162
PL	0.0100230	0.1811692	0.0022312	0.0017409	0.0089605	0.0077612	0.0038630	0.0086135
PT	0.0104685	0.1546038	0.0018087	0.0008933	0.0090242	0.0056258	0.0050529	0.0091548
RO	0.0131547	0.2396142	0.0056941	0.0015964	0.0142684	0.0090482	0.0027424	0.0063044
SE	0.0087175	0.1242747	0.0037646	0.0017728	0.0111591	0.0078736	0.0030412	0.0049843
SI	0.0093576	0.1470774	0.0052611	0.0016343	0.0126488	0.0073922	0.0034237	0.0075853
SK	0.0095253	0.1778753	0.0048896	0.0015642	0.0114306	0.0078732	0.0032411	0.0066186
TR	0.0053161	0.1157451	0.0034589	0.0007216	0.0070651	0.0039840	0.0031749	0.0051644

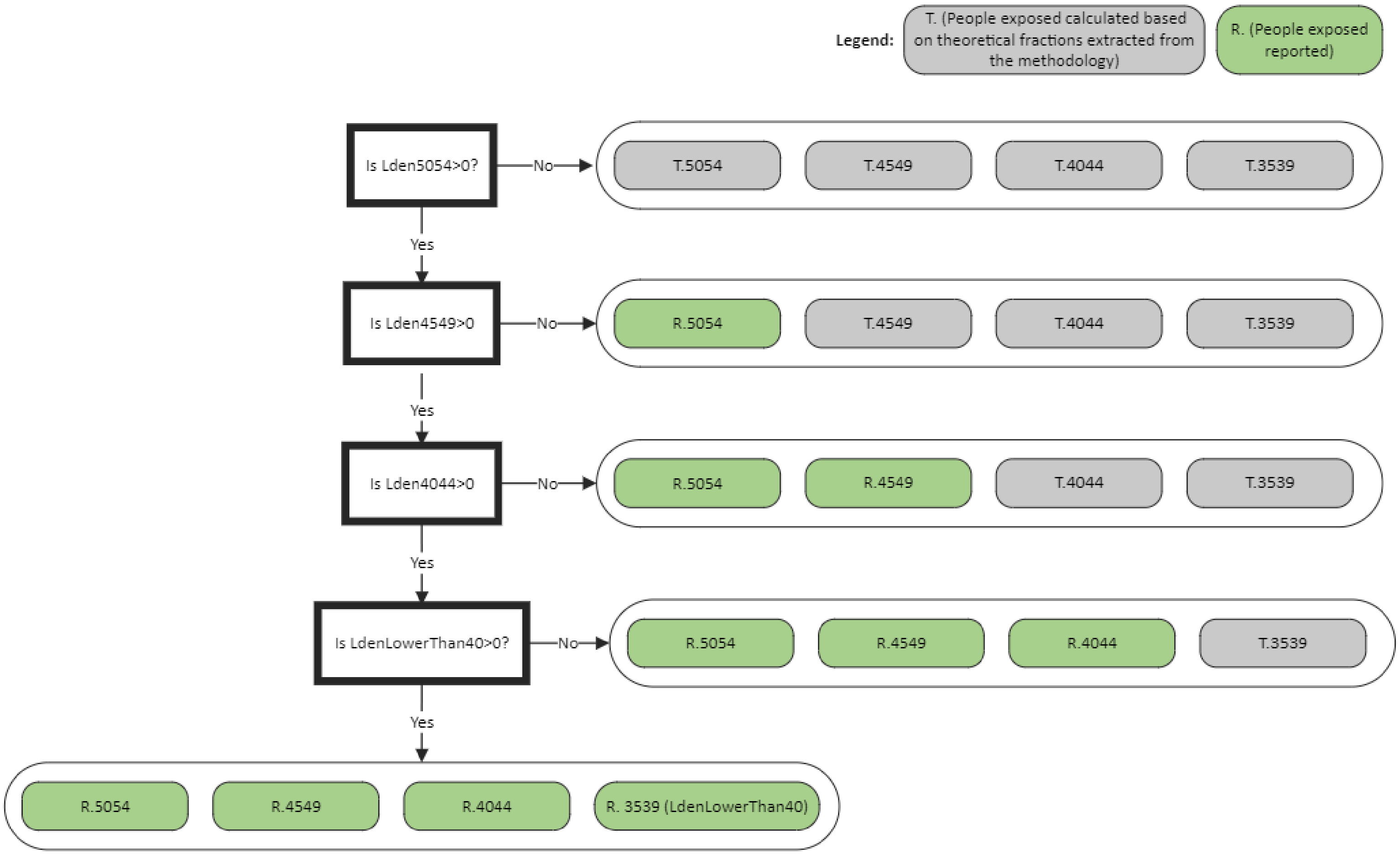
* All estimates per person and per year.

Abbreviations: IHD = Ischaemic heart disease, CVD = Cardiovascular disease, YLL = Years of life lost, YLD = Years of healthy life lost due to disability

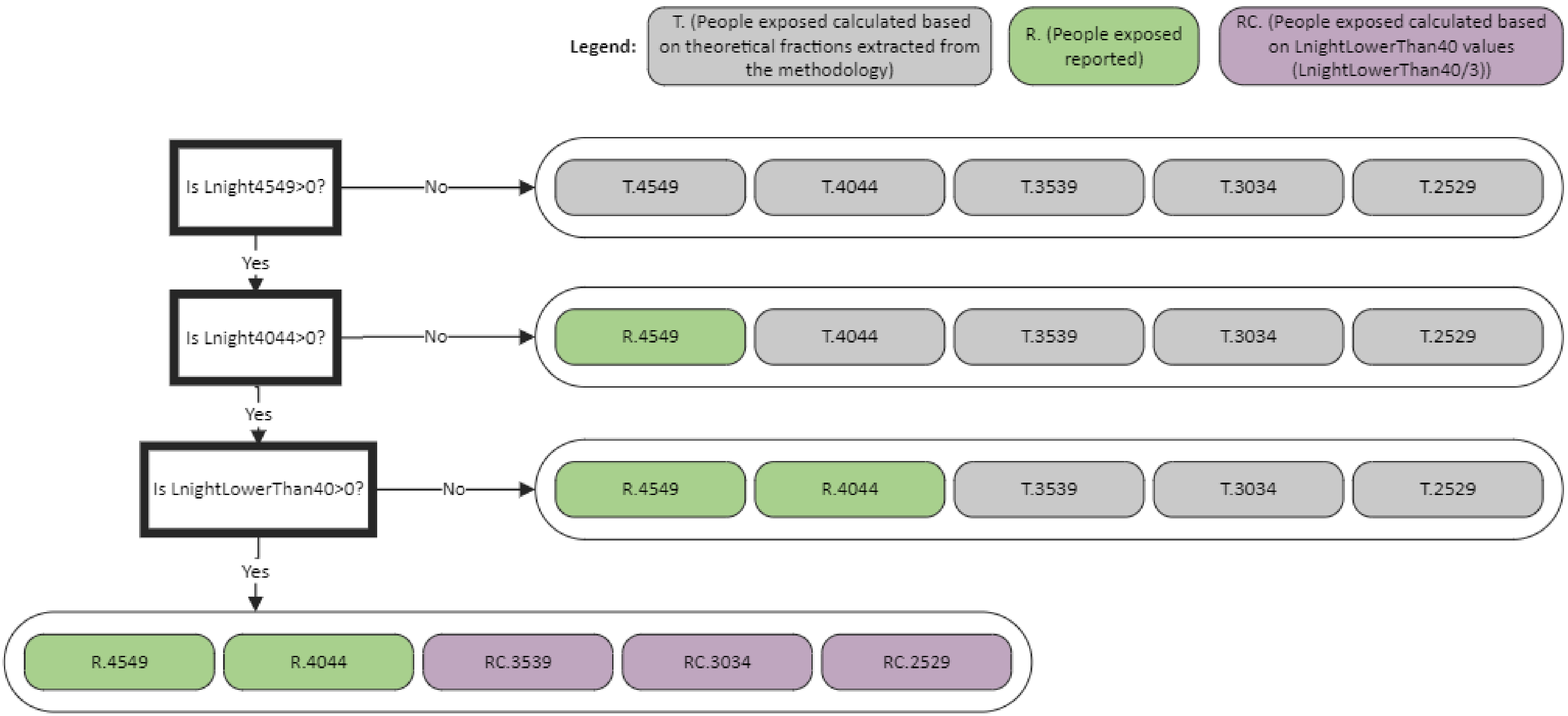
Data from IHME 2019 (https://vizhub.healthdata.org/gbd-results/) using the following specifications:

11.Annex 5 – Methodology followed to use the reported exposure values below the END thresholds

Case: reported exposure values below the END thresholds for L_{den} for the following noise sources: agglomeration air, agglomeration railway, agglomeration industry, major airports, major railways and major roads



Case: reported exposure values below the END thresholds for L_{night} for the following noise sources: agglomeration air, agglomeration railway, agglomeration industry, major airports, major railways and major roads

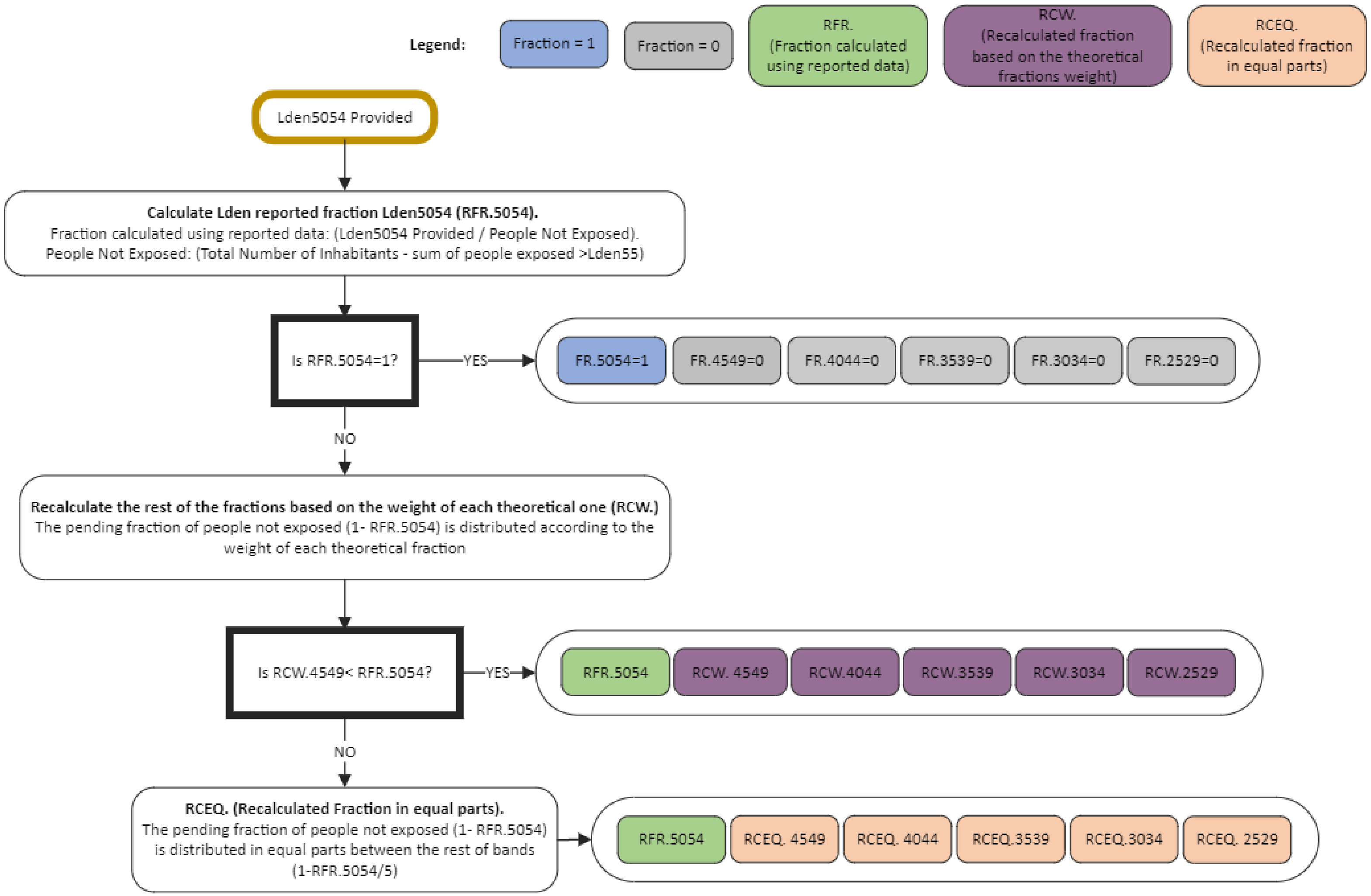


(*) Estimated values for L_{night} noise bands 30-34 dB and 25-29 dB are only included for calculation purposes, and discarded when including the results in the defined output.

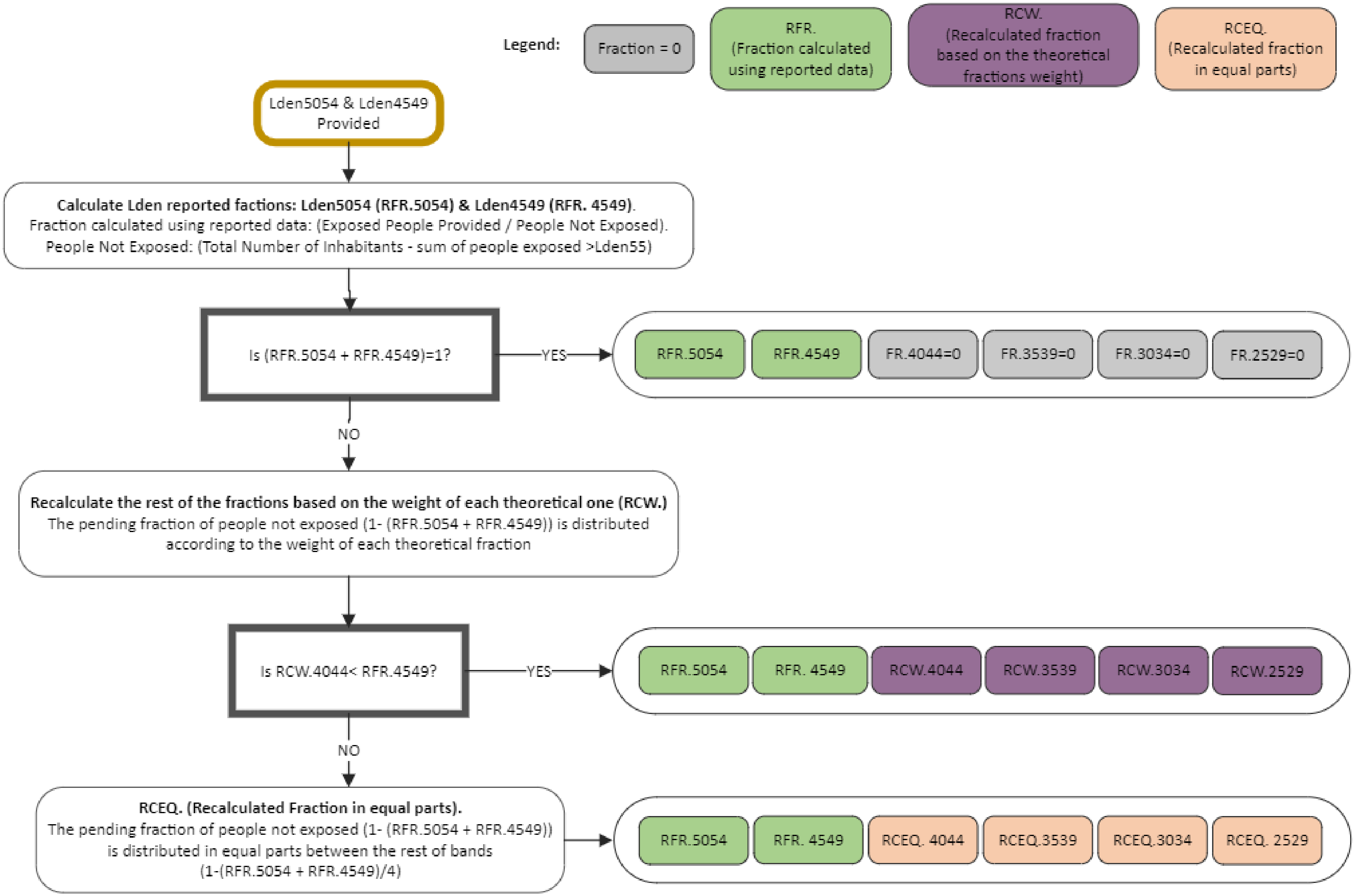
Case: reported exposure values below the END thresholds for L_{den} for agglomeration road noise source

Different processes are applied depending on the lower bands provided, which are detailed in the following schemas:

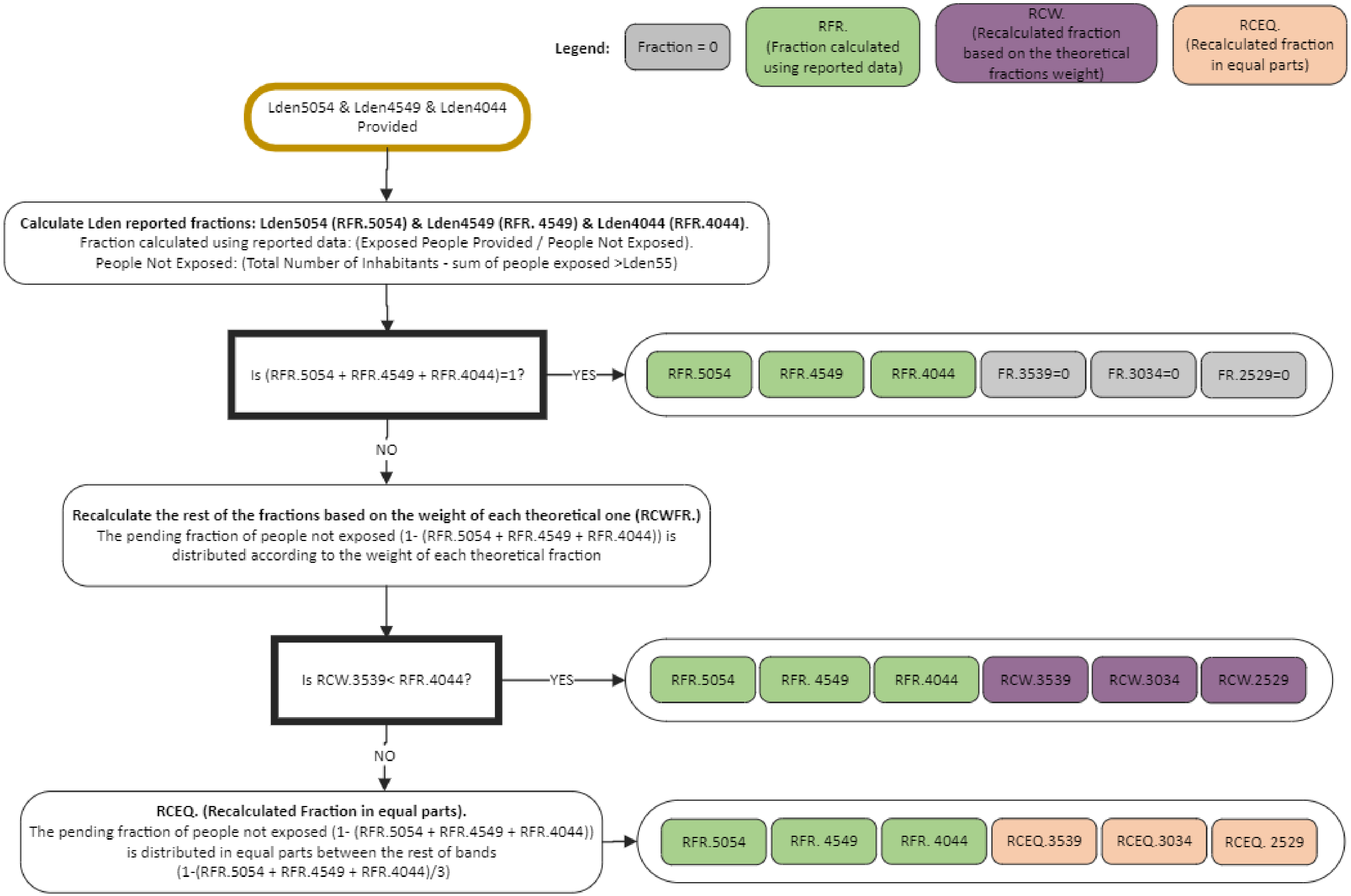
AggRoad: Lden5054 provided



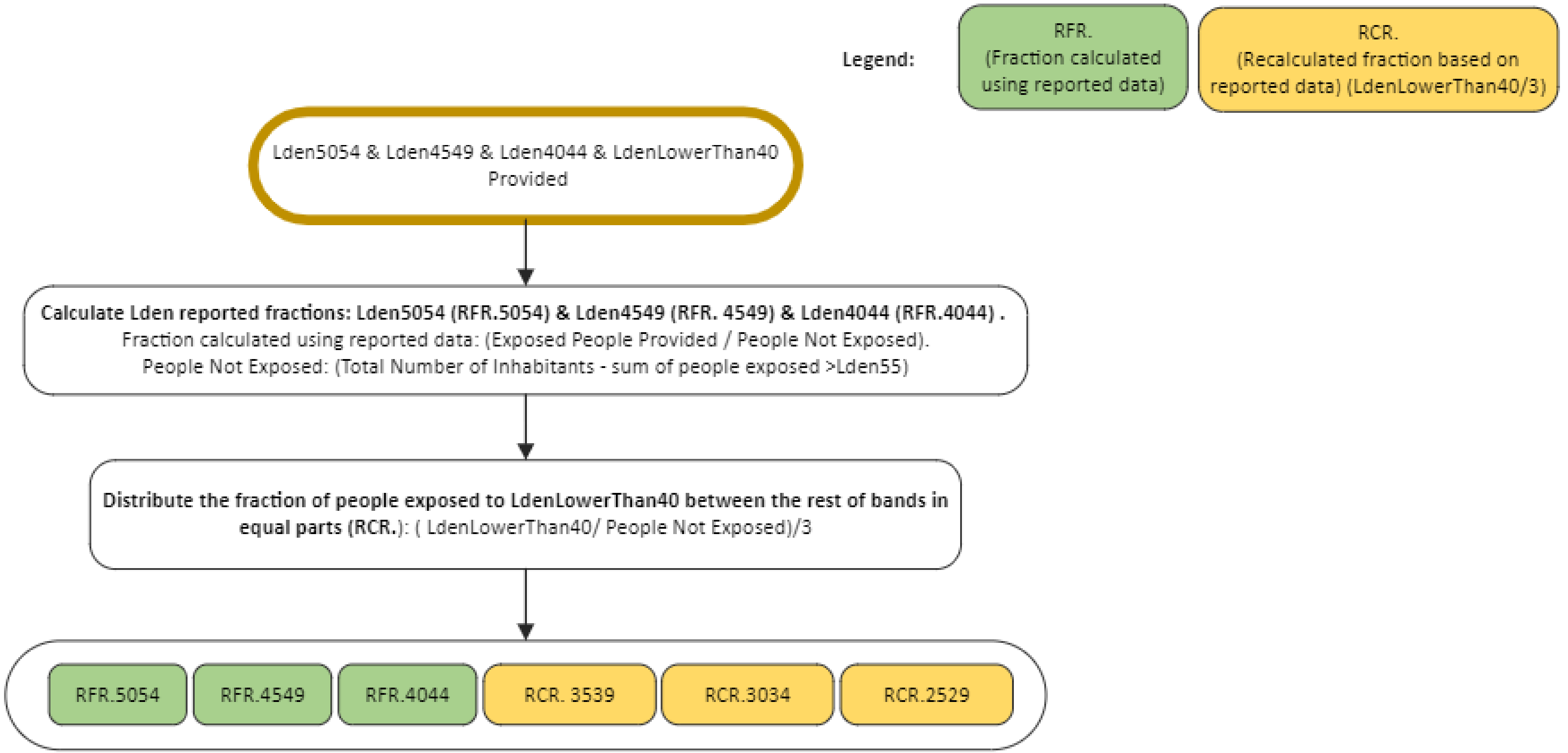
AggRoad: Lden5054&Lden4549 provided



AggRoad: Lden5054&Lden4549& Lden4044 provided

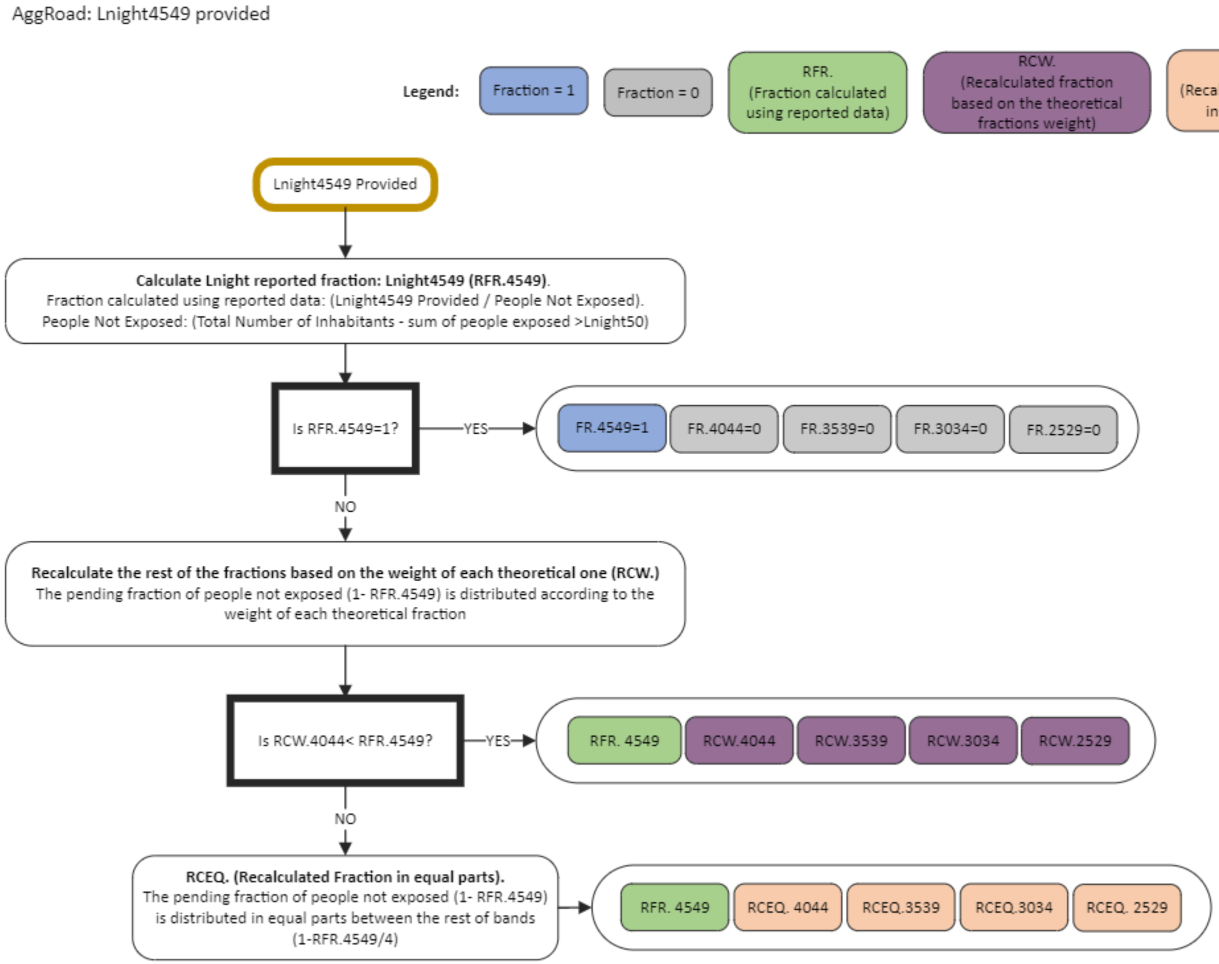


AggRoad: Lden5054&Lden4549&Lden4044&LdenLowerThan40 provided



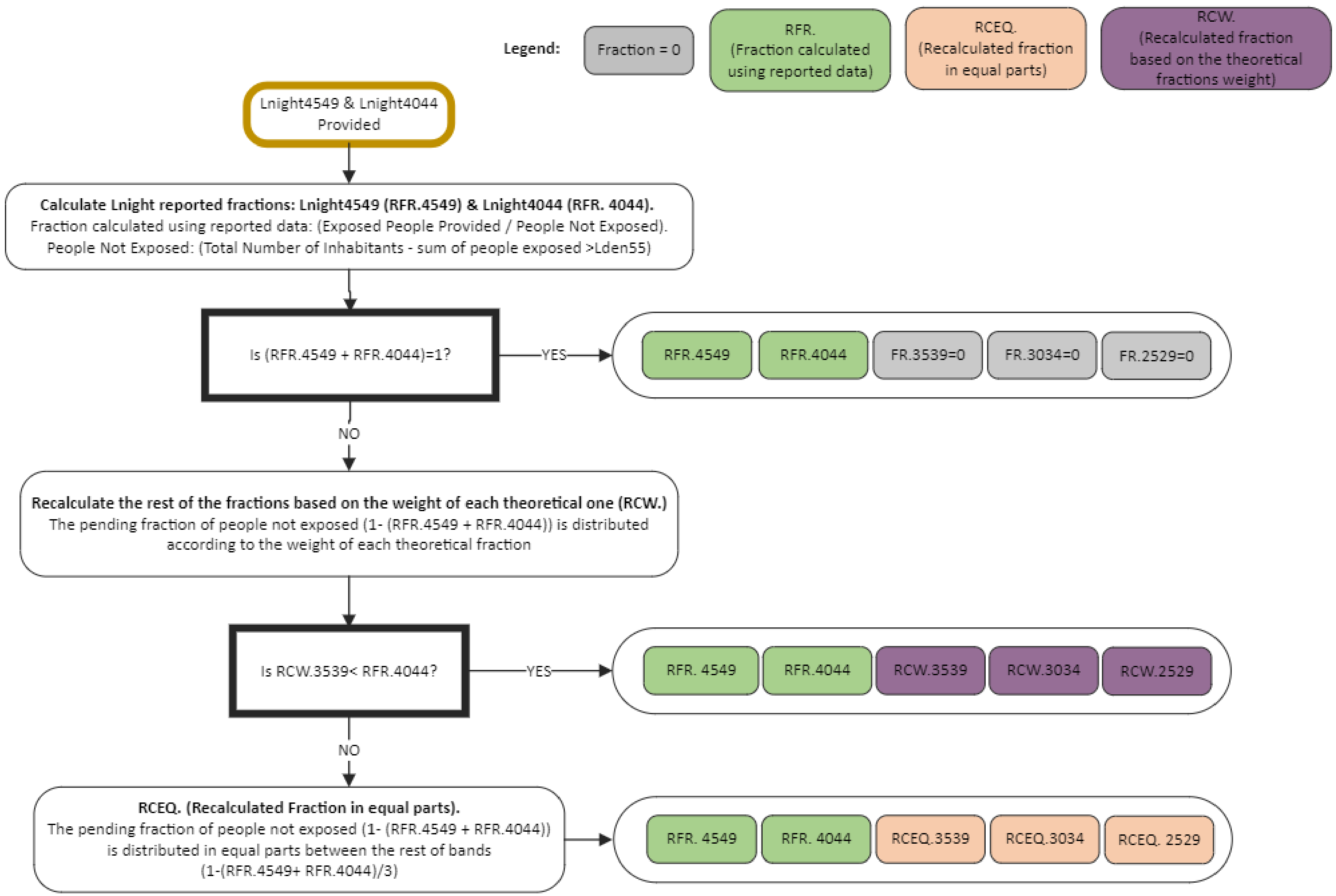
Case: reported exposure values below the END thresholds for L_{night} for agglomeration road noise source

Different processes are applied depending on the lower bands provided, which are detailed in the following schemas:



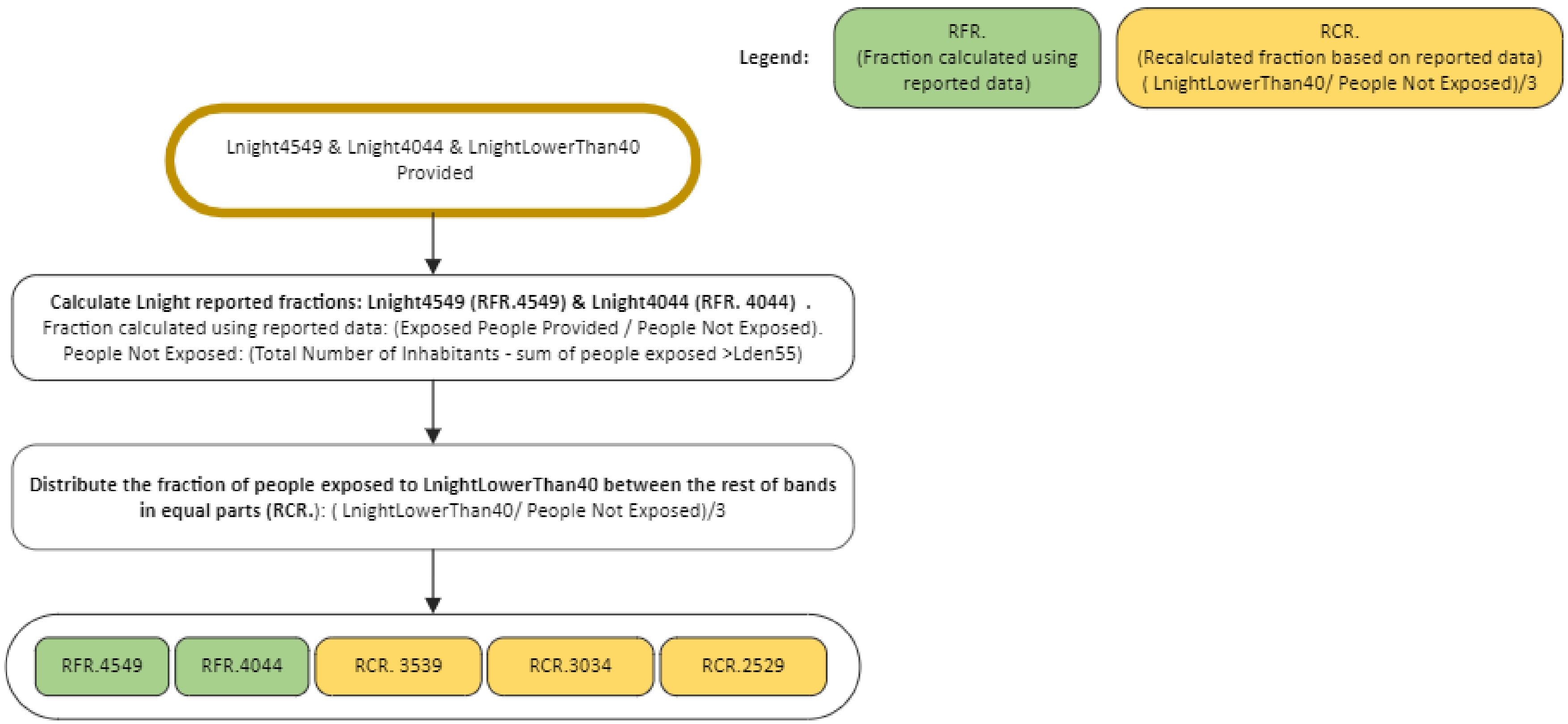
(*) Estimated values for L_{night} noise bands 30-34 dB and 25-29 dB are only included for calculation purposes, and discarded when including the results in the defined output

AggRoad: Lnight4549&Lnight4044 provided



(*) Estimated values for L_{night} noise bands 30-34 dB and 25-29 dB are only included for calculation purposes, and discarded when including the results in the defined output.

AggRoad: Lnight4549&Lnight4044&LnightLowerThan40 provided



(*) Estimated values for L_{night} noise bands 30-34 dB and 25-29 dB are only included for calculation purposes, and discarded when including the results in the defined output.

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