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Review article

Transportation noise exposure and anxiety: A systematic review and meta-analysis

Yuliang Lan [a,*], Hannah Roberts [a], Mei-Po Kwan [a,b], Marco Helbich [a]

[a] Department of Human Geography and Spatial Planning, Utrecht University, the Netherlands
[b] Department of Geography and Resource Management and Institute of Space and Earth Information Science, Chinese University of Hong Kong, Hong Kong, China

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ABSTRACT

Background: Exposure to transportation noise is hypothesized to contribute to anxiety, but consistent associations have not been established.
Objective: To provide a comprehensive synthesis of the literature by examining associations between traffic-related noise (i.e., road traffic noise, railway noise, aircraft noise and mixed traffic noise) and anxiety.
Methods: We systematically searched Web of Science, Scopus, Embase, PubMed, and PsycINFO for English-language observational studies published up to February 2020 reporting on the traffic noise-anxiety association. We appraised the risk of bias using an assessment tool and the quality of evidence following established guidelines. A random-effects meta-analysis was performed for pooled and separated traffic-related noise sources.
Results: Of the 3575 studies identified, 11 fulfilled the inclusion criteria and 9 studies were appropriate for meta-analysis. For the pooled overall effect size between transport noise and anxiety, we found 9% higher odds of anxiety associated with a 10 dB(A) increase in day-evening-night noise level (L_{den}), with moderate heterogeneity (OR = 1.09, 95% CI: [0.97; 1.23], $I^2 = 70\%$). The association was more likely to be significant with more severe anxiety (OR = 1.08, 95% CI: [1.01; 1.15], $I^2 = 48\%$). Sub-group analysis revealed that the effects of different noise sources on anxiety were inconsistent and insignificant. The quality of evidence was rated as very low to low.
Conclusions: Our findings support the hypothesis of an association between traffic noise and more severe anxiety. More high-quality studies are needed to confirm associations between different noise types and anxiety, as well as to better understand underlying mechanisms.

1. Introduction

Anxiety is a global public health priority (Patel et al., 2018; World Health Organization, 2013). It is estimated that anxiety has a prevalence of 7.3% (Baxter et al., 2013), affecting over 260 million people worldwide (World Health Organization, 2017).

Transportation noise is increasingly recognized as a risk factor of human health (European Environment Agency, 2020; World Health Organization, 2018). Existing reviews have suggested that transportation noise is associated with numerous physical health outcomes including ischaemic heart disease (van Kempen et al., 2018), hearing loss (Śliwińska-Kowalska and Zaborowski, 2017), tinnitus (Śliwińska-Kowalska and Zaborowski, 2017), and adverse birth outcomes (Dzhambov and Lercher, 2019a; Nieuwenhuijsen et al., 2017).

Previous reviews have also indicated a relationship between transportation noise and mental health. However, these have been limited in several ways. Clark et al. (2007) identified 11 papers published until 2005 that examined the effects of environmental noise exposure on mental health, and neither differentiated the range of psychological outcomes nor separated transportation noise from environmental noise. Two recent reviews (Clark and Paunovic, 2018; Clark et al., 2020) assessed the effects of transportation noise on anxiety and depression, but no meta-analysis was conducted and the disorders were examined as a whole. Dzhambov and Lercher, 2019b considered depression and anxiety separately, though only examined the effects of road traffic noise; aircraft noise and railway noise were not included. In sum, previous reviews failed to provide overall quantitative evidence for the effects of transportation noise exposure on anxiety, and how such

* Corresponding author. Department of Human Geography and Spatial Planning, Faculty of Geosciences, Utrecht University, Princetonlaan 8a, 3584, BC, Utrecht, the Netherlands.
E-mail address: y.lan@uu.nl (Y. Lan).

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associations differ across different types of transportation noise (i.e., road traffic noise, railway noise, aircraft noise, and mixed traffic noise).

This review aims to obtain a comprehensive understanding of the associations between exposure to multiple sources of transport-related noise and anxiety. Our objectives are 1) to summarize and evaluate the evidence on the effects of transportation noise exposure on anxiety narratively and quantitatively through a meta-analysis; 2) to compare the different effects of road traffic noise, railway noise, aircraft noise, and mixed traffic noise; and 3) to assess the quality of evidence and to highlight key limitations in current research.

2. Methods

This review followed the PRISMA guidelines (Moher et al., 2009) (Table S1 in the supplementary materials shows the checklist). The protocol was registered on PROSPERO (Lan et al., 2020).

2.1. Eligibility criteria

Our systematic review was concretized according to the population-exposure-outcome outline (Moher et al., 2009) (Table 1). We only considered original peer-reviewed journal papers. Review articles, conference proceedings, abstracts, editorials, reports, letters, notes, chapters, books, and dissertations were excluded. There was no limitation on publication years and geographical areas, but only studies written in English were eligible.

2.2. Search strategy

We searched five databases (i.e., Web of Science, Scopus, Embase, PubMed, PsycINFO) without any time limitations on February 14th, 2020. Search terms were related to transportation noise and anxiety. For example, Web of Science was searched using the search string: (TS = ((traffic OR transport OR transportation OR environment OR environmental OR aircraft OR aircrafts OR railway OR railways OR road OR roads) AND noise)) AND (TS = (mental OR anxiety OR anxious OR anxiousness OR anxiolytic OR anxiolytics)) AND LANGUAGE: (English). We also made use of Medical Subject Headings terms and Embase's controlled vocabulary terms in PubMed and Embase. The search was conducted by the first author (YL). The full search strategies per database are available in Table S2.

2.3. Study selection

The records obtained from each database were merged in Endnote. After removing the duplicates, the screening was conducted by two reviewers (YL and HR) independently according to the eligibility criteria. The degree of agreement between both reviewers was measured by means of Cohen's kappa (Cohen, 1960). The kappa coefficients for the abstract screening and full-text eligibility were 0.95 and 0.94, indicating an almost perfect agreement between both reviewers. For studies that dealt with transportation noise and anxiety but did not report the relevant effect sizes, we contacted the authors, resulting in two more eligible papers (Baudin et al., 2018; Halonen et al., 2012).

Table 1
Study inclusion and exclusion criteria.

Category	Inclusion	Exclusion
Population	Studies on adults (≥18 years) or with all ages included	Studies only focusing on children or adolescents (<18 years); animal studies
Exposure	Road traffic, railway traffic, aircraft noise, or overall transportation noise assessed objectively (e.g., modeled noise) or subjectively (e.g., perceived noise)	Wind turbine noise, construction noise, residential noise, indoor noise, occupational noise, noise annoyance, noise as a covariate, no measurement of noise
Outcomes	Anxiety measured using validated scales (e.g., Beck Anxiety Inventory), questionnaires (e.g., self-reported question), or medication use (e.g., the use of anxiolytics)	Physical health, other mental health symptoms, anxiety in combination with other outcomes (e.g., insomnia, depression), quality of life, well-being

2.4. Data extraction

Relevant information was extracted by the first author using a standardized data extraction form which included the following: author and publication year, study area, participant information, study design, noise type and assessment, outcome and measures, adjustments, and reported effect sizes. All extracted data was reviewed by a second researcher, and any disagreement was solved by discussion until consensus was reached.

2.5. Effect sizes conversion

To prepare the data for meta-analysis, the extracted effect sizes were transformed (Table S3). We used odds ratios (OR) as the unified effect size for pooling since it was most commonly used in the selected studies. Two studies (Generaal et al., 2019; Klompmaker et al., 2019) reported effect sizes both with and without other types of exposure (e.g., greenery, air pollution) as confounders. To be more comparable with other included studies, we chose the one that did not adjust for other types of environmental exposure.

For studies using objective noise measurement, we rescaled the OR to a 10 dB(A) increment in noise exposure by using the following expression for comparability: exp((ln(reported effect estimate)/original unit increase) × 10). If categorical effect sizes across noise exposure groups were reported (Okokon et al., 2018), the effect sizes were summarized based on the generalized least square method (Orsini et al., 2006). We use L_{den} (day-evening-night noise level) as the noise metric to report our findings. Non-L_{den} noise indicators would not affect the OR estimate (Dzhambov and Dimitrova, 2016) when the effect was reported per 10 dB, so non-L_{den} noise indicators were not converted (Floud et al., 2011). One study (Floud et al., 2011) only measured daytime and night-time aircraft noise level and reported daytime OR and night time OR separately. Therefore, we averaged the two effect sizes to obtain a daily effect size.

Two studies reported effect sizes that were not ORs, therefore transformation was required. One study reported their results using a correlation coefficient (r) (Honold et al., 2012). We transformed r into an OR using the appropriate calculation (Table S3) (Lipsey and Wilson, 2001; Polanin and Snilstveit, 2016), and we used the variance to calculate the 95% confidence interval (CI). One study only reported mean and standard error (Stansfeld et al., 1996). We first calculated Cohen's d and its variance (Lipsey and Wilson, 2001). The category with the lowest noise level was used as the control group, while other categories were regarded as the experimental group. Cohen's d was then converted to an OR (Lipsey and Wilson, 2001; Polanin and Snilstveit, 2016). Full details on effect size transformation can be found in Table S3.

2.6. Meta-analysis

Random-effects meta-analysis was used to pool the extracted effect sizes. We applied the Sidik-Jonkman method rather than the maximum-likelihood method to estimate τ^2 as this was shown to result in more robust estimates when the number of studies is small (Sidik and Jonkman, 2007). Because the two transformed effect sizes (Honold et al., 2012; Stansfeld et al., 1996) may not be comparable with others, we conducted a sensitivity analysis to test model robustness by omitting them from the meta-analysis.

We used the Q statistic to test study heterogeneity (Cochran, 1954);

Table 2
Descriptive characteristics of the studies.

Studies	Sample	Country	Study design	Noise source	Noise measurement	Exposure assessment	Measures for anxiety	Adjustments in selected model	Effect sizes
Baudin, Lefèvre et al.	$n = 1244$; age range: ≥18	France	Cross-sectional	Aircraft	INM; yearly averaged L_{den} in 2013; no information about resolution	Noise level at home address	Self-reported anxiety; 2013	Age, sex, marital status, income, education	OR (95% CI): per 10 dB increase 0.73 [0.57; 0.93]
Díaz, López-Bueno et al.	$n = 1461$; age range: all	Spain	Ecological	Mixed traffic	Measured $L_{eq\text{-}Day}$ in 2010–2013; no information about resolution	City level averaged noise	Diagnosed anxiety based on ICD-10: F40-42; 2010–2013	No confounders at individual level	RR (95% CI): per 1 dB increase 1.20 [1.14; 1.26]
Floud, Vigna-Taglianti et al.	$n = 4642$; age range: 45-70	UK, Germany, Netherlands, Sweden, Italy, Greece	Cross-sectional	Aircraft Road	Aircraft: INM and ANCON-v2 (UK) models; yearly averaged L_{Aeq} and L_n in 2002; spatial resolution of the noise map was 250 × 250m Road: national models; yearly averaged $L_{Aeq\text{-}24h}$ in 2002; spatial resolution of the noise map was 10 × 10m	Noise level at home address	Self-reported use of anxiolytics; 2004–2006	Age, sex, BMI, alcohol intake, education, exercise and smoking status	OR (95% CI): per 10 dB increase Aircraft: LAeq-16h: 1.28 [1.04; 1.57] Ln Aircraft: 1.27 [1.01; 1.59] Road traffic: LAeq-24h: 1.06 [0.84; 1.33]
Generaal, Timmermans et al.	$n = 2980$; age range: 18-65	Netherlands	Cross-sectional	Mixed traffic (Road, railway, air)	Empara Noise tool; yearly averaged L_{den} in 2007; spatial resolution of the noise map was 25 × 25m	Noise level at home address	Beck Anxiety Inventory; 2004–2007	Age, sex, years of education and household income	OR (95% CI): per 1 S.D. increase 1.22 [1.09; 1.38]
Halonen, Vahtera et al.	$n = 6815$; age range: ≥18	Finland	Cross-sectional	Road	Nordic prediction method; yearly averaged Ln in 2005 (Vantaa) and 2006 (Helsinki); spatial resolution of the noise map was 10 × 10m	Noise level at the most exposed façades of residential buildings	6-item Trait Anxiety Inventory, 2004, 2005, 2008, 2009	Age, sex, marital status, occupational status	OR (95% CI): per 10 dB increase 0.97 [0.89; 1.05]
Honold, Beyer et al.	$n = 428$; age range: 16-91	Germany	Cross-sectional	Mixed traffic (Road, railway, subway, Tram)	Perceived noise; 2009	Proximate living environment	Self-reported anxiety; 2009	None	Correlation coefficient (r): 0.03
Klompmaker, Hoek et al.	$n = 354{,}827$; age range: ≥19; 42.8% elderly	Netherlands	Cross-sectional	Road Railway	STAMINA model; yearly averaged L_{den} in 2011; spatial resolution of the noise map was 80 × 80m - 10 × 10m	Noise level at home address	Prescription of anxiolytics; 2012	Sex, age, marital status, region of origin, education, income, smoking status/alcohol, physical activity, degree of urbanization	OR (95% CI): per interquartile range increase Road: 1.07 [1.03; 1.11] Rail: 1.01 [0.98; 1.04]
Ma, Li et al.	$n = 1125$; age range: 18-65	China	Cross-sectional	Road Railway	Perceived noise; 2017	Noise level at neighborhood level	Self-reported anxiety; 2017	Sex, age, income, education, employment status, marital status, residential status, housing tenure, perceived community traffic congestion, housing satisfaction, community attachment	OR (95% CI): Road: Very low: 1 (reference) Moderate: 0.899 [0.617; 1.311] High: 0.667 [0.435; 1.021] Rail: Very low: 1 (reference) Moderate: 1.257 [0.888; 1.781] High: 2.659 [1.639; 4.354]
Okokon, Yli-Tuomi et al.	$n = 5860$; age range: ≥25	Finland	Cross-sectional	Road	Nordic prediction method; yearly averaged L_{den} in 2011;	The highest L_{den} on façade points within 20m of home address	Self- reported use of anxiolytics; 2015	Sex, age, marital status, employment status, household income, alcohol	OR (95% CI): ≤45 dB: 1 (reference)

(*continued on next page*)

Table 2 (*continued*)

Studies	Sample	Country	Study design	Noise source	Noise measurement	Exposure assessment	Measures for anxiety	Adjustments in selected model	Effect sizes
					no information about resolution		(Helsinki) and 2016 (Espoo, Vantaa)	intake, current smoking status, level of physical activity and pet ownership	45.1–50 dB: 1.12 [0.77; 1.63] 50.1–55 dB: 1.09 [0.75; 1.58] 55.1–60 dB: 1.24 [0.85; 1.82] >60 dB: 1.34 [0.93; 1.93]
Stansfeld, Gallacher et al.	$n = 1725$; age range: 50-64	England	Cross-sectional	Mixed traffic	Modeled $L_{eq\text{-}16h}$ based on measured noise; no information about resolution	No details about exposure assessment	Anxiety subscales for GHQ-30	Age, social class, and noise sensitivity and anxiety at baseline	Mean (SE) of anxiety score: 51–55 dB: 4.70 (0.07) 56–60 dB: 5.20 (0.18) 61–65 dB: 4.89 (0.15) 66–70 dB: 5.02 (0.21)
Zock, Verheij et al.	$n = 4450$; age range: all	Netherlands	Cross-sectional	Road Railway	STAMINA Model; yearly averaged L_{den} in 2007 (road) and 2008 (rail); spatial resolution of the noise map was 10 × 10m	Noise level at neighborhood level	General practitioner data; 2013	Sex, age, household income and socio-economic status (individual level) and municipality and neighborhood (group level)	OR (95% CI): per 10 dB increase Road: 0.94 [0.59; 1.52] Rail: 1.06 [0.87; 1.29]

Abbreviations. STAMINA: Standard Model Instrumentation for Noise Assessments; INM: Integrated Noise Model; ANCON: Aircraft Noise Contour model; GHQ: Generalized Health Questionnaire; ICD: International Classification of Diseases; BMI: Body Mass Index; S.D.: Standard Deviation; RR: Relative risks; SE: Standard Error.

$p < 0.05$ indicated heterogeneity across studies. To quantify the effect of heterogeneity, we used the I^2 statistic (Higgins and Thompson, 2002). I^2 >50% indicated statistically significant heterogeneity, and ≤25%, 25–75%, and ≥75% represent low, moderate, and high heterogeneity (Mustafic et al., 2012). All analyses were conducted in R 3.6.1 (R Core Team, 2019) with the 'meta' and 'metafor' packages (Schwarzer et al., 2015; Viechtbauer, 2010).

2.7. Publication bias assessment

A funnel plot which plots the distribution of effect sizes against the precision of the effect estimation was used to assess the potential role of publication bias (Guski et al., 2017). Since OR was the selected effect size measurement, we used the inverse standard error on the Y-axis and the logged OR on the X-axis as advised elsewhere (Sterne and Egger, 2001). We used Egger's test to assess the asymmetry of the funnel plot; $p < 0.05$ indicated substantial asymmetry due to publication bias (Egger et al., 1997).

2.8. Quality assessment

The risk of bias was assessed based on criteria developed in a previous review on noise and mental health (Dzhambov and Lercher, 2019b). Studies were scored on ten items: publication type, study design, selection of participants, sample representativeness, noise exposure quality, noise exposure timeframe, assessment of anxiety, confounding factors, statistical analysis, and additional bias. The criteria are listed in Table S4. Because we additionally considered perceived noise, 'noise exposure quality' was adapted so that the relevant studies would score 0 for this item.

The overall quality of evidence for the effect of traffic noise on anxiety was assessed based on the adapted version of Grading of Recommendations Assessment, Development, and Evaluation (GRADE) guidelines (van Kempen et al., 2017). The starting level of the quality of evidence was determined according to the study design. This initial level was then increased or decreased considering several factors. Factors that lower the confidence of evidence were: risk of bias, inconsistency of results, indirectness of evidence, imprecision, publication bias, and number of studies. In contrast, a dose-response gradient, a large magnitude of effect, and confounding that underestimate the associations can increase confidence. Two researchers (YL and HR) completed the quality assessment independently. Any disagreement was discussed until consensus was reached.

3. Results

Fig. 1 shows the flow chart of study selection. We identified 63 papers for full text screening, 11 papers met the inclusion criteria. Detailed exclusion reasons for each of the 52 papers can be found in Table S6. Key descriptive characteristics of included studies are shown in Table 2. One paper (Ma et al., 2018) had categorical effect sizes based on perceived noise level (i.e., low, moderate, high), which could not be summarized to one effect size. One effect size was not comparable with others because sampling was not at the individual level (Díaz et al., 2020). Thus, 9 studies with 12 noise-anxiety effects were included in our meta-analysis.

3.1. Study characteristics

The 11 included studies were from 9 countries, most originated from the Netherlands ($n = 4$). Only one was conducted outside Europe (Ma et al., 2018). One was an ecological longitudinal study (Díaz et al., 2020); all others were cross-sectional. The majority of studies ($n = 8$) used logistic regression.

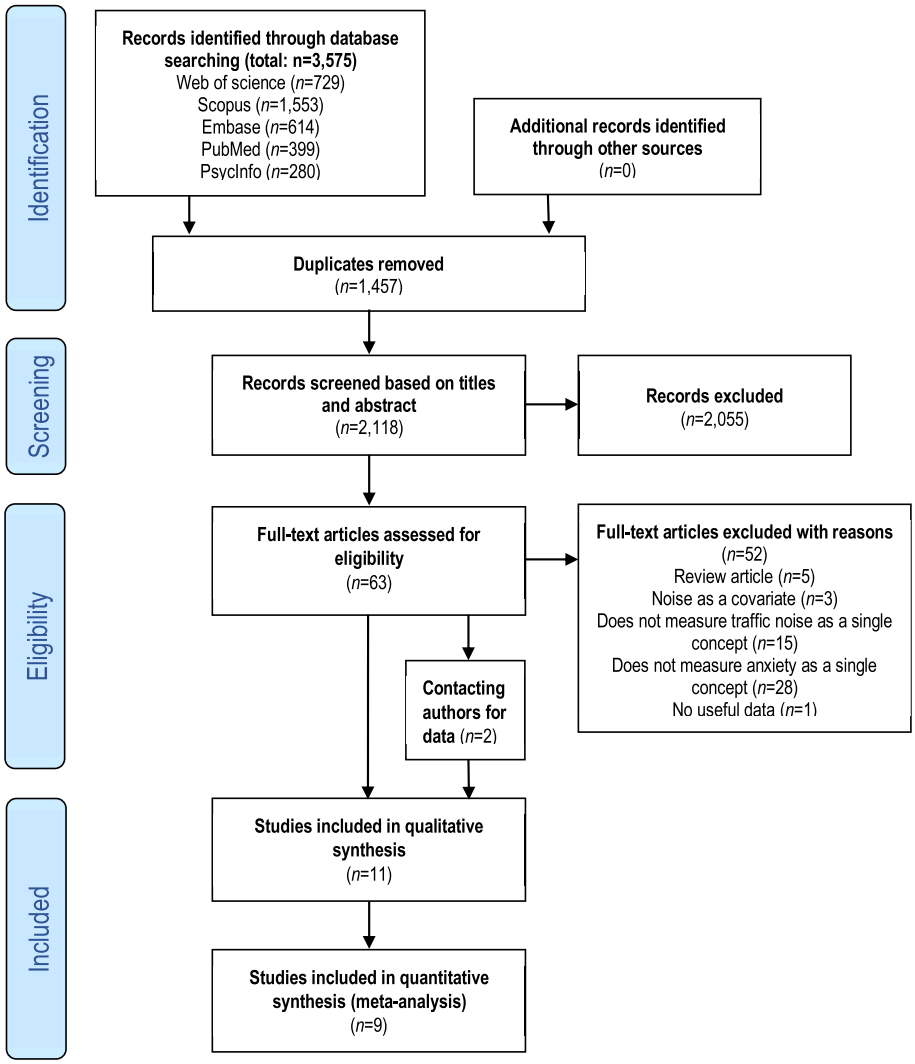

Fig. 1. Flow chart of study selection.

3.2. Participants

Sample sizes ranged from 428 (Honold et al., 2012) to 354,827 participants (Klompmaker et al., 2019), with a total of 385,557 individuals. Most studies included adults (≥18 years), and one oversampled the elderly (Klompmaker et al., 2019). Two studies included only middle-aged and elderly participants (Floud et al., 2011; Stansfeld et al., 1996). Two studies sampled participants that lived close to airports (Baudin et al., 2018; Floud et al., 2011). One focused on specific locations by sampling from specific street blocks (Honold et al., 2012), others on specific population groups (community, primary care, and specialized mental health care) (Generaal et al., 2019), public sectors (Halonen et al., 2012), and people with anxiety disorders (Díaz et al., 2020). Studies that were less representative of the general population were assigned lower scores (and therefore higher risk) in the risk of bias assessment (Table S5).

3.3. Measurements for transportation noise exposure

Three studies investigated mixed traffic noise (Díaz et al., 2020; Generaal et al., 2019; Honold et al., 2012), while others assessed one or two different types of noise (i.e., road traffic noise, railway noise, aircraft noise). Road traffic noise was most frequently assessed (Floud et al., 2011; Halonen et al., 2012; Klompmaker et al., 2019; Ma et al., 2018; Okokon et al., 2018; Stansfeld et al., 1996; Zock et al., 2018). Two articles dealt with aircraft noise (Baudin et al., 2018; Floud et al., 2011), and three with railway noise (Klompmaker et al., 2019; Ma et al., 2018; Zock et al., 2018). Most studies ($n = 8$) measured noise objectively using noise models. According to the criteria of 'noise exposure quality' item in Table S4, three studies used models with moderate accuracy (Generaal et al., 2019; Halonen et al., 2012; Okokon et al., 2018) and three with limited accuracy (Baudin et al., 2018; Floud et al., 2011; Klompmaker et al., 2019). The remaining studies had low accuracy due to using postcode-level exposure (Zock et al., 2018) or not providing resolution information (Stansfeld et al., 1996). One study used noise data from monitoring stations (Díaz et al., 2020) and two obtained perceived noise by questionnaire (Honold et al., 2012; Ma et al., 2018); these were scored as low accuracy in the quality assessment (Table S5).

The ecological study (Díaz et al., 2020) used the average noise at the city level, while other studies assessed residential noise exposure at the most exposed façades (Halonen et al., 2012; Okokon et al., 2018), at the home address (Baudin et al., 2018; Floud et al., 2011; Generaal et al., 2019; Klompmaker et al., 2019), or at the neighborhood level (Honold et al., 2012; Ma et al., 2018; Zock et al., 2018). In most studies, noise assessments were based on data preceding (Floud et al., 2011; Klompmaker et al., 2019; Okokon et al., 2018; Zock et al., 2018) or during the study period (Baudin et al., 2018; Díaz et al., 2020; Generaal et al., 2019; Halonen et al., 2012; Honold et al., 2012; Ma et al., 2018). One study had a five-year follow-up that ensured a long-term residential history (Stansfeld et al., 1996).

3.4. Anxiety measures

The most frequently used anxiety measure was self-reported anxiety level ($n = 6$) using validated questionnaires (e.g., Beck Anxiety Inventory). Two studies relied on self-reported anxiolytic use (Floud et al., 2011; Okokon et al., 2018) and two were based on register-based diagnosis and/or prescription data (Klompmaker et al., 2019; Zock et al., 2018). Only one was based on clinical diagnosis (Díaz et al., 2020). Two studies also considered participants' anxiety level before the study period: Stansfeld et al. (1996) included anxiety at baseline in adjustments; Generaal et al. (2019) identified the control group based on the diagnoses of anxiety in the year prior to study period.

3.5. Risk of bias

Table S5 shows the risk of bias assessments according to the criteria defined in Table S4. Five studies had low risk of bias (Floud et al., 2011; Klompmaker et al., 2019; Ma et al., 2018; Okokon et al., 2018; Zock et al., 2018) and no study was suspected to have a high risk of bias. Most studies were biased in terms of noise exposure and anxiety assessment, and nearly all studies were biased in 'selection of participants' and 'study design'.

3.6. Meta-analysis

3.6.1. Summary effect size

The observed 12 effect sizes ranged from 0.73 to 1.86. The summary OR from the random-effects model was 1.09 (95% CI: [0.97; 1.23]), indicating 9% higher odds of anxiety associated with a 10 dB(A) increase in L_{den} (Fig. 2a). The heterogeneity was moderate ($Q = 36.44$, $p < 0.01$; $I^2 = 70\%$). When the two transformed effect sizes (Honold et al., 2012; Stansfeld et al., 1996) were excluded (Fig. 2b), there were only minor changes in the summary effect size (OR = 1.07, 95% CI: [0.93, 1.23]). Sensitivity analysis signified the transformations were acceptable.

Considering that various anxiety measures would result in different definition of anxiety, we conducted subgroup meta-analyses (Fig S1) for 'inventory based' and 'medication use/diagnosis' groups, assuming that anxiety indicated by medication use/diagnosis is more severe. The result showed that the effect of traffic noise in the 'medication use/diagnosis' group was significant (OR = 1.08, 95% CI: [1.01; 1.15], $I^2 = 48\%$), implying that traffic noise was significantly associated with more severe anxiety. However, in the 'inventory based' group, the association between traffic noise and anxiety was not significant (OR = 1.11, 95% CI: [0.83; 1.49]). The high heterogeneity ($I^2 = 84\%$) could partially be caused by differences in the questionnaires used in the studies and discrepancies in participants' self-judgement.

To assess the effects of different noise sources (i.e., road traffic noise, railway noise, aircraft noise and mixed traffic noise) on anxiety, subgroup meta-analyses were conducted (Fig. 3). The effect of road traffic

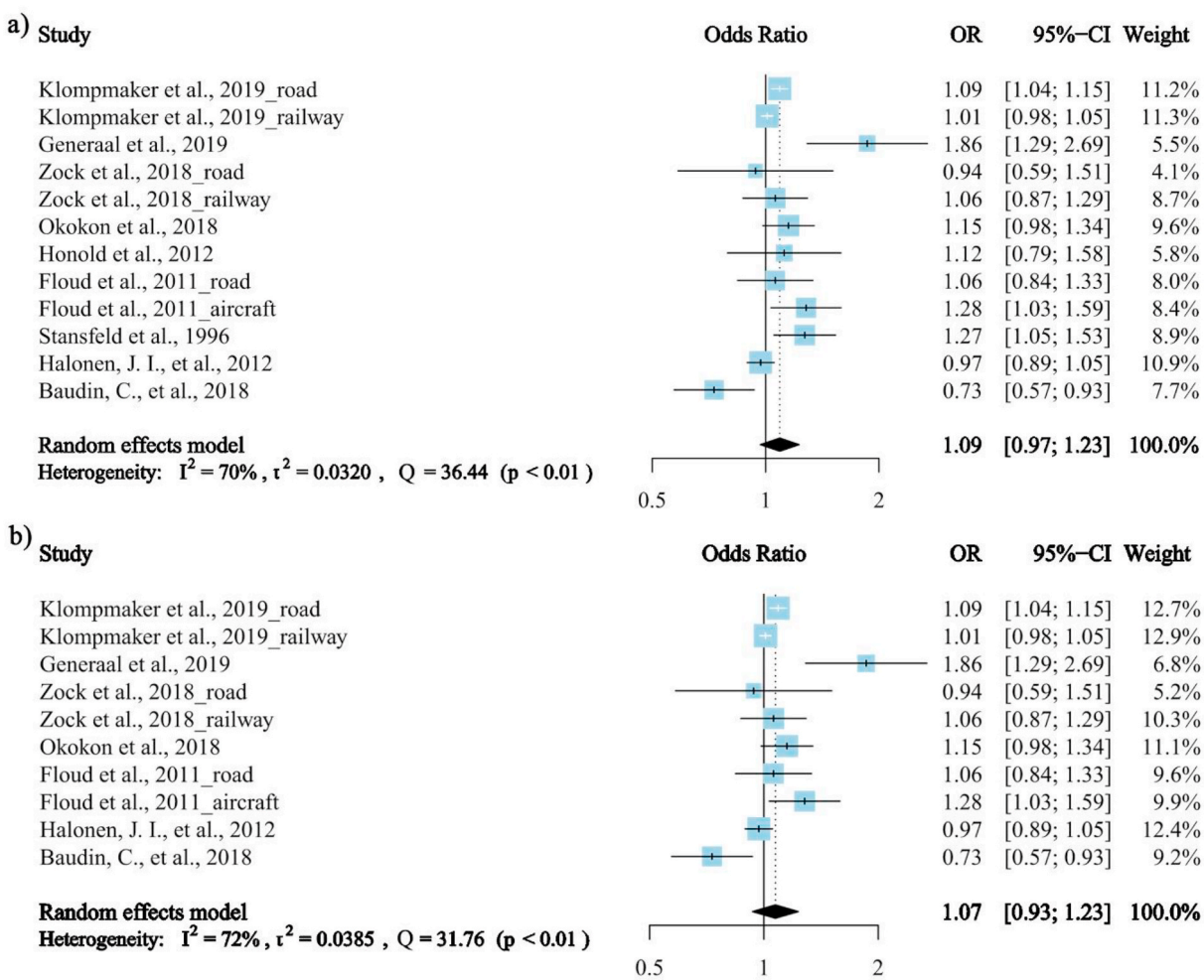

Fig. 2. Forest plot showing the effect of a 10 dB(A) increase in noise level on anxiety: a) all effect sizes included; b) without the two transformed effect sizes. (OR: odds ratio; CI: confidence interval).

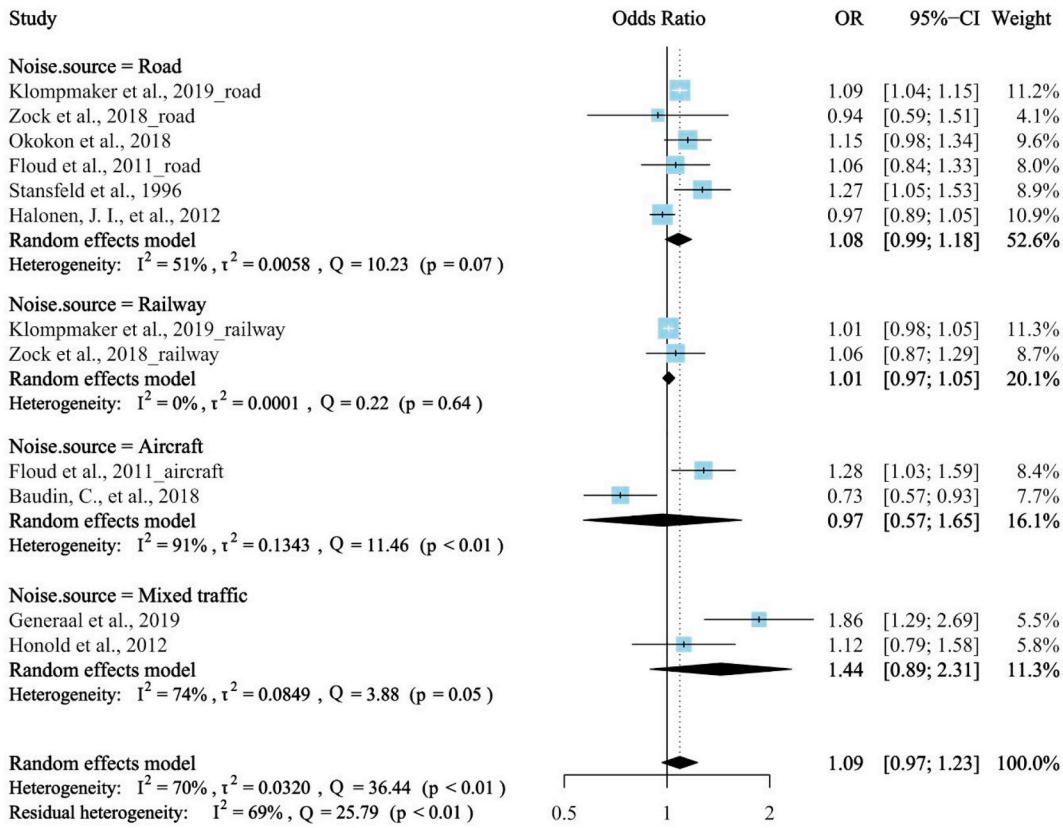

Fig. 3. Forest plot showing the effect of different noise source on anxiety with a 10 dB(A) increase in noise level (OR: odds ratio; CI: confidence interval).

noise was almost significant with OR = 1.08 (95% CI: [0.99; 1.18]), indicating that 8% higher odds of anxiety were associated with a 10 dB (A) increase in road traffic noise; the heterogeneity was moderate (I^2 = 51%). The association between railway noise and anxiety was negligible, with 1% higher odds with a 10 dB(A) increase; we found no indication of heterogeneity. For aircraft noise, the result revealed a slight (3%) mitigation effect on anxiety. However, pronounced heterogeneity (I^2 = 91%) resulted in a wide CI. Mixed traffic noise had the highest odds (44%) of anxiety, but the effect was not significant. The high heterogeneity (I^2 = 74%) could be explained by combining different measures of noise.

3.6.2. Publication bias

With all effect sizes included, the funnel plot (Fig. 4) revealed a roughly symmetric funnel shape. Studies with lower precision (at the base of the funnel plot) scattered more widely on both sides of pooled effect size; non-statistically significant studies were not missing (studies in the blank area in the bottom of Fig. 4). Thus, there was no evidence of serious publication bias, which was also indicated by the non-significant result of Egger's test (intercept = 0.816; 95% CI: [-0.752; 2.384]; t = 1.083; p = 0.304). Due to the limited number of effect sizes, publication bias for each noise type is difficult to judge.

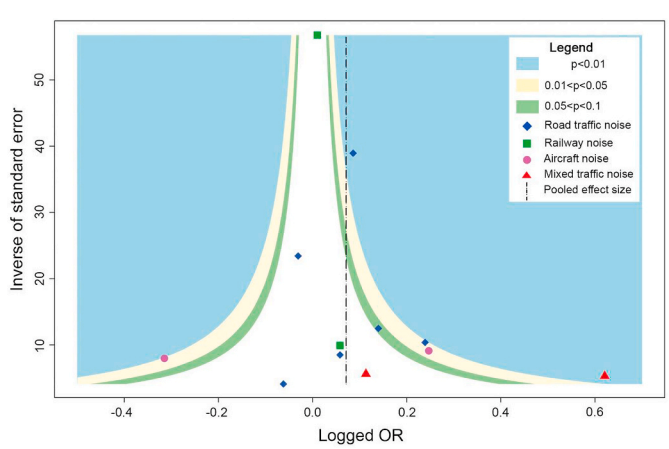

Fig. 4. Funnel plot to assess potential publication bias.

3.7. Quality of evidence according to GRADE

Since all studies were cross-sectional except one ecological study (Díaz et al., 2020), we started with a 'low' quality rating for all noise types (Table S7). For railway noise studies, this rating held since it was downgraded one level due to imprecision but upgraded by one level for 'exposure response', as most studies found an association between noise and anxiety. For road, aircraft and mixed traffic noise, the initial rating was downgraded twice due to inconsistent effects and imprecision in the ORs. The road, aircraft and mixed traffic noise studies were therefore judged overall as 'very low' quality.

4. Discussion

4.1. Main findings

In this systematic review and meta-analysis, we found that the overall effect of traffic noise was nearly significant with 9% higher odds of anxiety associated with a 10 dB(A) increase in L_{den}. Traffic noise was more likely to be significantly associated with more severe anxiety. Subgroup meta-analyses revealed discrepancies in the effects of different noise sources, but none of them were significant. Mixed traffic noise had the highest odds (44%) of anxiety, followed by road traffic noise (8%), and railway noise (1%). Aircraft noise revealed a slight mitigation effect on anxiety with 3% lower odds of anxiety associated with a 10 dB(A)

increase in L_{den}. No study was suspected to have a high risk of bias. However, the GRADE assessment indicated the current evidence is of poor quality.

4.2. Possible mechanisms

The relationship between noise and mental health may be explained by stress and behavioral processes. The stress-diathesis hypothesis suggests that transportation noise, as an environmental stressor, can increase physiological arousal and stress hormone secretion (e.g., adrenaline and cortisol) through repeated stimulation of the endocrine system and autonomic nervous system (Babisch, 2002; Hahad et al., 2019; Stansfeld and Clark, 2011, 2015). Prolonged activation of these responses may cause mental disorders including anxiety (Clark and Paunovic, 2018; McEwen, 1998). According to the behavioral mechanism, it emphasizes that people proactively deal with exposure to noise by adjusting their behavior in noisy conditions to reduce exposure through the appraisal of noise (in terms of danger, loss of quality, the meaning of the noise, challenges for environmental control, etc.) and coping strategies (Clayton, 2012; Stansfeld and Clark, 2011). As a result, actively coping with noise may be sufficient to mitigate the ill effects (Van Kamp, 1990).

4.3. Strengths and limitations

To our knowledge, this review provides the most comprehensive and up-to-date evidence on the transportation noise-anxiety association. We added seven more effect sizes in the meta-analysis compared with a previous comparable meta-analysis (Dzhambov and Lercher, 2019b), and the effects of different noise sources were examined separately using sub-group meta-analysis.

However, this review also has some limitations. First, the numbers of studies for each noise type were limited. This reduces confidence in the summary ORs. Second, all effect sizes included in the meta-analysis were from Europe, so the results may not represent other contexts worldwide. Third, our review was limited to English articles only. This prevented studies published in other languages from being included.

4.4. Implications for future research

First, all included studies are cross-sectional or ecological, both of which provide limited evidence for causality (Setia, 2016). Future research with stronger study designs (e.g., cohort study, case-control study) are needed to support causal statements (Noordzij et al., 2009).

Second, standardization of confounders is another suggestion for future studies. Current studies adjusted different sets of confounding factors, which limits the comparability among studies. Most included age, sex, education, marital status, and income as confounders, while a few additionally adjusted for smoking status (Floud et al., 2011; Klompmaker et al., 2019; Okokon et al., 2018), alcohol intake (Floud et al., 2011; Klompmaker et al., 2019; Okokon et al., 2018) and physical activity (Floud et al., 2011; Klompmaker et al., 2019; Okokon et al., 2018). Aligning confounders is useful to ensure comparisons across studies.

Third, virtually all studies considered exposure to noise based on where people live (Generaal et al., 2019; Klompmaker et al., 2019; Zock et al., 2018). Given that most people spent a significant proportion of the day not at home, such exposure assessments are likely biased and vulnerable to exposure misclassification (Helbich, 2018; Kwan, 2012, Kwan, 2018a, Kwan, 2018b). We believe progress can be made through noise exposure assessments based on people's daily mobility. To obtain information on people's mobility and noise exposures en route, geospatial technologies such as Global Positioning System-equipped smartphones and portable noise sensors can be used to capture location and exposure in real-time (Helbich, 2019;Kou et al., 2020; Ma et al., 2020). Besides, exposure duration is another largely disregarded issue because the non-auditory health effects of noise exposure are more likely to be indirect and chronic (Hahad et al., 2019). While studies have started to explore the effects of long-term noise exposure on depressive symptoms (Orban et al., 2016), cognitive impairment (Tzivian et al., 2016) and psychological functions (Tzivian et al., 2015), only one study included long-term exposures along (Stansfeld et al., 1996). Scientific progress could be made by including long-term exposures along with people's residential history, which allows us to address the duration and accumulation of exposures (Helbich, 2018).

5. Conclusions

Our findings suggest that traffic noise is positively associated with more severe anxiety. The effects of different noise sources were not significant. The GRADE assessment indicated the poor quality of evidence, which limits confidence in obtaining the same findings in high-quality studies. More studies are needed to confirm associations between transportation noise from various sources and anxiety, as well as to better understand underlying mechanisms.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envres.2020.110118.

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