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Association between aircraft, road and railway traffic noise and depression in a large case-control study based on secondary data

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ABSTRACT

Background: Few studies have examined the relationship between traffic noise and depression providing inconclusive results. This large case-control study is the first to assess and directly compare depression risks by aircraft, road traffic and railway noise.

Methods: The study population included individuals aged ≥40 years that were insured by three large statutory health insurance funds and were living in the region of Frankfurt international airport. Address-specific exposure to aircraft, road and railway traffic noise in 2005 was estimated. Based on insurance claims and prescription data, 77,295 cases with a new clinical depression diagnosis between 2006 and 2010 were compared with 578,246 control subjects.

Results: For road traffic noise, a linear exposure-risk relationship was found with an odds ratio (OR) of 1.17 (95% CI=1.10–1.25) for 24-h continuous sound levels ≥70 dB. For aircraft noise, the risk estimates reached a maximum OR of 1.23 (95% CI=1.19–1.28) at 50–55 dB and decreased at higher exposure categories. For railway noise, risk estimates peaked at 60–65 dB (OR=1.15, 95% CI=1.08–1.22). The highest OR of 1.42 (95% CI=1.33–1.52) was found for a combined exposure to noise above 50 dB from all three sources.

Conclusions: This study indicates that traffic noise exposure might lead to depression. As a potential explanation for the decreasing risks at high traffic noise levels, vulnerable people might actively cope with noise (e.g. insulate or move away).

1. Introduction

Traffic noise is an environmental risk factor for various diseases. A report of the World Health Organization (WHO) estimates that yearly at least one million disability adjusted life years (DALY) are lost from diseases (ischemic heart disease, cognitive impairment of children, sleep disturbance, tinnitus, annoyance) related to traffic noise in Western Europe (WHO, 2010). One illness that might be affected by traffic noise is depression: previous research shows that traffic noise induces various stress reactions and insomnia, and these factors as well as chronic noise itself have been shown to affect mental health and particularly depression (Stansfeld and Matheson, 2003; Baglioni et al., 2011). Depression is one of the most common mental disorders, and a leading cause of disability worldwide (WHO, 2015). However, the relation between traffic noise and depression is unclear. Early evidence for a relation between airport noise exposure and an increased submission to psychiatric units of hospitals in London (Abey-Wickrama et al., 1969) and Los-Angeles (Meecham and Smith, 1977) could not be confirmed in two later studies producing inconclusive evidence (Jenkins et al., 1981; Tarnopolsky et al., 1980). However, for residents living close to a military air base a positive dose-response relationship between aircraft noise and depressiveness was found (Hiramatsu et al., 1997). Furthermore, while Stansfeld et al. (1996) found no relation between road traffic noise and psychiatric disorders, Halpern (2014) reported a weak association. Some studies find positive relations between aircraft noise exposure and the prescription frequency and amount of tranquilizing, and sleep-inducing drugs, as well as antidepressants (Floud et al., 2011; Greiser et al., 2007). Additionally, two recent studies support a link between higher traffic noise exposure and higher depression risks: while Greiser and Greiser (2010) only found an association between aircraft noise and depression for women, Orban et al. (2016) reported a generally increased risk of

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depressive symptoms for road traffic noise. Moreover, there are several studies examining the relationship between traffic noise and self-reported mental health (e.g., Sygna et al., 2014; Kishikawa et al., 2009). Overall, the evidence pointing towards a positive relation between road and aircraft traffic noise and depressive disorders – mostly stemming from relatively small studies – is still inconclusive and requires further research. To date, there has been no study that specifically examined the relation between railway traffic noise and depression. More conclusive and thorough evidence on the relation between traffic noise and depression is of high interest to both the scientific community and policy makers: it could provide insights to the common mental illness depression by examining a potential environmental risk, as well as informing public debates about necessary measures to protect residents. With equal noise levels, a much larger proportion of people state to be highly annoyed by aircraft noise than by railway noise and road traffic noise. This might be partly explained by differences in the sound characteristics: While road traffic is accompanied by rather continuous background noise, aircraft and railway traffic noise are both characterised by more irregular, disruptive, louder single noise events. We therefore regard a separate analysis of the health effects of different types of traffic noise as important.

The aim of this study was to assess the relation between depressive disorders and all traffic noise combined as well as separately for aircraft, road, and railway traffic noise, in a large secondary-data based case-control study.

2. Methods

2.1. Study region and population

The study region of the NORAH (Noise-related Annoyance, Cognition, and Health) case-control study was located around the Frankfurt international airport (see Fig. 1). The study population consisted of all individuals living in the study area aged 40 years or older in 2010 and insured by one of three large statutory health insurance funds in the period between 2005 and 2010 (n=1,026,670). The study population includes about 23% of all people aged ≥40 living in the study region.

2.2. Noise exposure assessment

Acoustic exposure was estimated separately for each type of traffic noise (aircraft, railway and road traffic) for each individual residential address. For aircraft noise, average and maximum sound levels were calculated using historical radar data from the German flight safety operator (DFS) according to the guidelines for calculations of noise abatement zones (AzB) (Bundesregierung, 2008).These values were then verified by comparing them to measurements of local monitoring stations. For railway and road traffic noise, the sound levels were calculated by using estimates of traffic exposure and estimating sound reductions between the source of sound and the immission sites according to the methods for calculation (VBUS, VBUSCH) used for EU noise mapping (Bundesregierung, 2006; European Union, 2002). Traffic exposure at the source was measured through road traffic counts and information by the Federal Railway Authority and the German Railway environmental department. Sound reduction calculations were based on a digital landscape model including information on the landscape and the footprint of buildings, and on information regarding the position of noise barriers and walls along roads and railways. More detailed information on the acoustic models, exposure calculations, uncertainties and plausibility checks can be found elsewhere (Möhler et al., 2015, 2016).

2.3. Data linkage

The participating health insurance funds provided pseudonymized hospital and ambulatory diagnostic data (ICD-10 codes) and prescription data according to the Anatomical Therapeutic Chemical/Defined Daily Dose Classification (ATC) for each reporting year between 2005 and 2010. Traffic noise data and individuals' address data were linked by a Data Linkage Office in Bremen, or for one insurance fund by the health insurer. These data were then pseudonymized by substituting address data with the study ID and forwarded to the Data Linkage office in Dresden that linked the diagnostic data and the traffic noise data using the study ID. For a more detailed description see Seidler et al. (2016a, 2016b, 2016c).

2.4. Definition of cases and control subjects

Patients with at least two ambulant or at least one hospital diagnosis of unipolar depressive disorder in the study period who had been insured for more than twelve continuous months were defined as cases with clinically diagnosed depression (Table 1). Diagnoses were coded according to the international classification of diseases (ICD-10). Following the evidence-based national disease management guidelines (DGPPN et al., 2015), only unipolar depressive disorders, that is depressive episodes (F32), recurrent depressive disorder (F33), persistent mood [affective] disorders (only dysthymia, F34.1), and mixed anxiety and depression disorders (F41.2) were included as cases. Other affective disorders were not included in the case definition. Furthermore, only patients who received a new diagnosis of a depressive disorder between 2006 and 2010 were included as cases (i.e. that did not get diagnosed with depression in at least four quarters before the newly diagnosed depressive disorder). The case definition criteria were fulfilled by 85,180 individuals. Of these, 77,295 individuals (90.5%) could be linked to traffic noise data and were included as cases in the analysis. Individuals without depression diagnosis between 2005 and 2010 and who had been insured for more than twelve continuous months fulfilled the criteria for control subjects (n=637,487). Of these, 578,246 individuals (90.7%) could be linked to traffic noise data and were included in the analysis as control subjects.

2.5. Statistical analyses

Logistic regression analysis was performed to calculate odds ratios (ORs) and 95% confidence intervals for each type of traffic noise separately, and in a combined model. The continuous sound levels for each traffic noise source were grouped in 5 dB categories, with sound level exposure below 40 dB as the reference category. For aircraft noise, individuals with continuous sound levels below 40 dB but at least six maximum nightly levels above 50 dB formed a separate exposure category. The exposure-risk relationship was examined, applying a linear (included traffic noise term: $B_1 \times L_{pAeq,24h}$) or third-degree polynomial model (included traffic noise term: $B_1 \times L_{pAeq,24h} + B_2 \times L_{pAeq,24h}{}^2 + B_3 \times L_{pAeq,24h}{}^3$) to the 24-h continuous sound levels ($L_{pAeq,24h}$) with a starting point of 35 dB. In case of a difference between linear and third-degree model Akaike Information Criteria (AIC) of 5 or less, a linear model was regarded as statistically adequate.

In an additional analysis, exposure to different combinations of traffic noise sources was examined against a reference group with no exposure of 40 dB or more to traffic noise of any source.

We calculated interaction terms between sex and the single continuous traffic noise variables. In case of significant sex-noise interaction, the results were stratified by sex.

2.6. Confounders

All analyses were adjusted for age, sex, urban living environment, and the local proportion of people receiving unemployment benefits as an indicator of socio-economic status (SES). When available, the

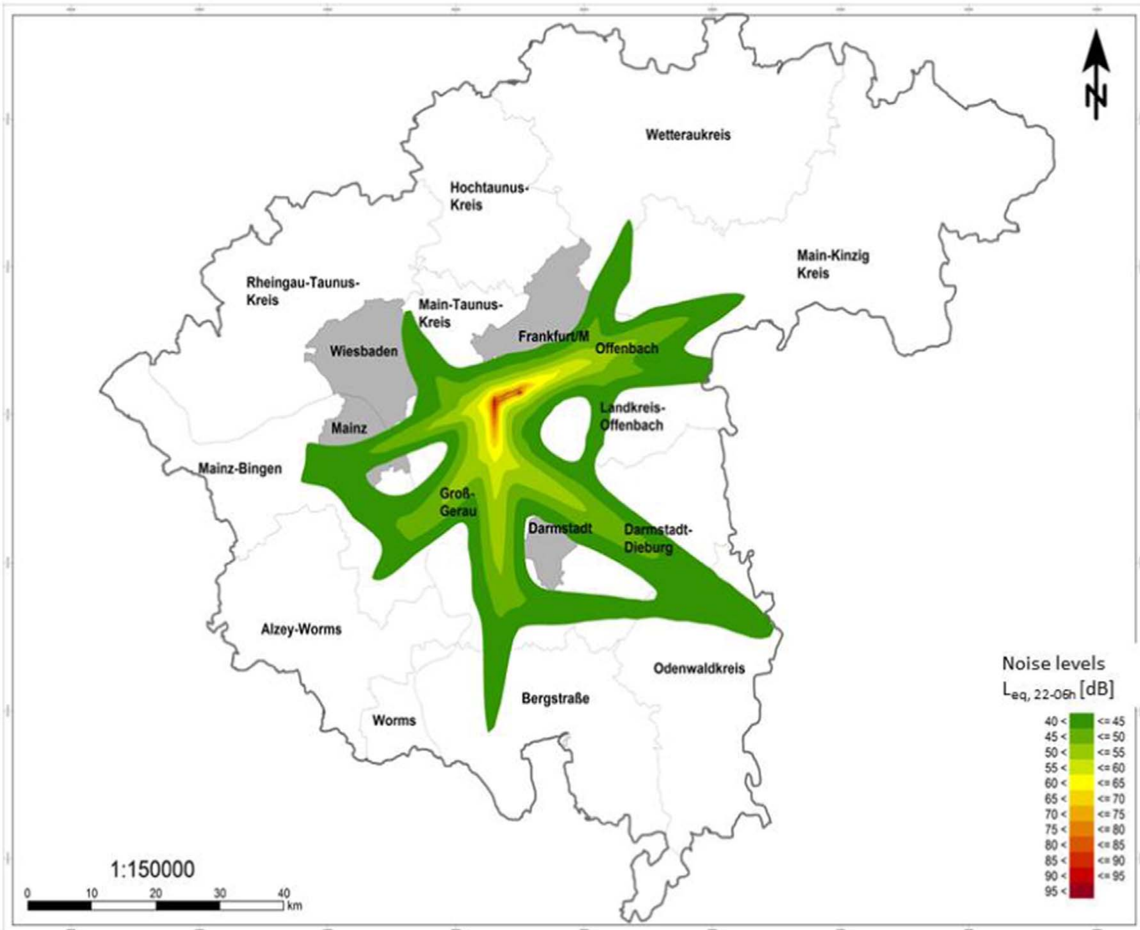

Fig. 1. Map of the study area* (surrounded by a grey line), and contours of continuous sound levels for nighttime aircraft noise exposure in 2005. *The study area included the administrative region Darmstadt, the districts Mainz-Bingen and Alzey-Worms, and the cities Mainz and Worms. Areas with nighttime (22–06 h) sound levels below 40 dB are shaded white.

Table 1
Definition of a depressive disorder

ICD-10 Classification		Depressive disorder[a]
F32	Depressive episode	1× hospital discharge diagnosis F32.-, F33.-, F34.1 or F41.2
F33	Recurrent depressive disorder	2× ambulatory secure diagnosis F32.-, F33.-, F34.1 or F41.2 (“g”=secure diagnosis) in two consecutive quarters
F34.1	Dysthymia	
F41.2	Mixed anxiety and depressive disorder	

[a] To fulfil the case definition of a depressive disorder, at least one of the criteria (line 1 and/or line 2) must be fulfilled.

analyses were further adjusted for individuals’ socio-economic status (education and job title). We aimed to take into account potential confounding by SES, as low socioeconomic status in childhood and adolescence has been found to be associated with a higher risk of depressive symptoms in adulthood (Elovainio et al., 2012, 2015). However, for 60% of the cases and 59% of control subjects this information was not available, mainly due to retirement or co-insurance with a family member. In an additional subgroup analysis, we therefore included only individuals with available individual socio-economic information (i.e. education or occupation). We included urban living environment (urban municipalities and suburban/rural districts as a dichotomous variable) as a potential confounder in our regression models, as a meta-analysis found that mood disorders are considerably more frequent in urban compared to rural living environments (Peen et al., 2010).

3. Results

The characteristics of cases with clinically diagnosed depression and control subjects are given in Table 2. There was no statistically significant sex-noise interaction on depression risk for any of the traffic noise sources (no table), we therefore did not stratify for sex. Fig. 2 shows results for each separate type of traffic noise (for the case numbers and for the risks of nightly traffic noise see Table 3); Fig. 3 gives the results restricting the analysis to individuals with known SES (for the case numbers and for the risks of nightly traffic noise see Table 4).

3.1. Aircraft noise and depression

Including noise as a third-degree polynomial increased the model fit, thus, the relation between aircraft noise and depression does not seem to be adequately represented by a linear model. Instead, the exposure-risk relation observed in the categorical analysis can be described as a reversed U-shape. For noise levels between 40 and 45 dB, the depression risk was already elevated to OR=1.13 (95% CI 1.10–1.15). The OR increased further for noise levels between 45 and 50 dB (OR=1.18; 95% CI 1.16–1.21) and peaked at 1.23 (95% CI 1.19–1.28) for noise between 50 and 55 dB. The OR then dropped to 1.09 (95% CI 1.02–1.16) for the 55 to 60 dB category, and further to 0.71 (95% CI 0.38–1.31) for > 60 dB, however, case numbers were low for this category. For the night time period (22–06 h) a comparable

Table 2
Characteristics of cases with depression and control subjects

	Cases		Control subjects	
	n	%	n	%
Total	77,295	100.0	578,246	100.0
Sex				
Males	24,914	32.2	292,239	50.5
Females	52,381	67.8	286,007	49.5
Age [yrs.]				
35 to < 45	9,486	12.3	93,554	16.2
45 to < 50	10,003	12.9	68,556	11.9
50 to < 55	9,965	12.9	59,127	10.2
55 to < 60	10,210	13.2	56,189	9.7
60 to < 65	7,225	9.3	52,716	9.1
65 to < 70	7,585	9.8	67,815	11.7
70 to < 75	7,492	9.7	64,420	11.1
75 to < 80	6,170	8.0	45,080	7.8
80 to < 65	4,897	6.3	35,587	6.2
≥85	4,262	5.5	35,202	6.1
Urban-rural living environment				
Urban	28,062	36.3	190,331	32.9
Rural	49,233	63.7	387,915	67.1
Education				
Primary/secondary education, no vocational education	6,811	8.8	45,556	7.9
Primary/secondary education with vocational education	12,618	16.3	102,748	17.8
Graduated from high school, no vocational education	344	0.4	3,272	0.6
Graduated from high school and vocational education	1,261	1.6	10,276	1.8
College degree	1,038	1.3	7,083	1.2
University degree	983	1.3	8610	1.5
Education unknown	54,240	70.2	400,701	69.3
Occupation according to Blossfeld				
AGR Agricultural occupations	189	0.2	2,426	0.4
EMB Unskilled manual occupations	3,232	4.2	26,513	4.6
QMB Skilled manual occupations	2,440	3.2	26,563	4.6
TEC Technicians	383	0.5	4,151	0.7
ING Engeneers	131	0.2	1,842	0.3
EDI Simple services	6,221	8,0	43,895	7.6
QDI Qualified services	1,392	1.8	7,714	1.3
SEMI Semiprofessionals	2,603	3.4	12,332	2.1
PROF Professionals	183	0.2	1,694	0.3
EVB Simple commercial and administrative occupations	2,700	3.5	16,450	2.8
QVB Qualified commercial and administrative occupations	5,676	7.3	40.977	7.1
MAN Managers	387	0.5	4,564	0.8
SONS Other	692	0.9	5,306	0.9
Unknown	51,006	66.1	383,819	66.4
Local proportion of persons receiving unemployment benefits (SGBII; quintiles)[a]				
≤6.7%	24,311	31.4	198,610	34.4
> 6.7 to ≤7.5%	12,675	16.4	91,422	15.8
> 7.5 to ≤8.7%	8,434	10.9	65,525	11.3
> 8.7 to ≤12.7%	23,998	31.0	166,747	28.8
> 12.7%	7,877	10.2	55,942	9.7

[a] Calculation of quintiles: frequent duplication of SGB II-values led to an uneven distribution.

reversed U-shape was found, with a significantly decreased OR of 0.72 (95% CI=0.56–0.93) in the highest exposure category (Table 3). In the analysis of nightly aircraft noise, a significantly increased risk (OR=1.07, 95% CI=1.05–1.09) was observed for individuals with nightly maximum sound levels above 50-dB (NAT 6) and continuous noise levels below 40-dB.

When the analysis was restricted to individuals with available individual SES information (Table 4), the reversed U-shape of the exposure-risk relation disappeared in the analysis of 24 h sound levels, but remained – albeit less distinct – in the analysis of the night time periods. The ORs were generally higher, reaching 1.35 (95% CI=1.28–1.43) in the 50 to < 55 dB noise category and 1.37 (95% CI 0.65–2.91) for noise levels above 60 dB.

3.2. Road traffic noise and depression

In the categorical analysis, an almost monotone risk increase was observed with increasing road traffic noise levels. For a continuous sound level of 40 to < 45 dB, the depression risk was 1.02 (95% CI 1.00–1.06), increasing up to 1.17 (95% CI 1.10–1.25) for the loudest road traffic noise category (≥70 dB). The exposure-risk relation was adequately represented by a linear model; thus, a risk increase of 3.7% (95% CI 2.8–4.6) per 10 dB road traffic noise was estimated. The ORs for the night time period (Table 3) were comparable to the ORs for 24-h continuous sound levels, increasing almost monotonically with increasing noise levels.

When only including cases with available SES information (Table 4), the ORs were substantially higher, with an OR of 1.25 (95% CI=1.17–1.33) for noise levels between 65 and < 70 dB. The continuous risk increased to 5.2% per 10 dB.

3.3. Railway traffic noise and depression

In the categorical analysis, we found the highest OR of 1.15 (95% CI=1.08–1.22) for 60 to < 65 dB continuous railway noise levels. For higher noise levels, the OR decreased again, with an OR of 0.93 (95% CI=0.82–1.06) for ≥70 dB. Including a third-degree polynomial improved the model fit, indicating non-linearity of risk. For the night time period a comparable reversed U-shape was found (Table 3).

In the sub-analysis with available SES information (Table 4), the exposure-risk relation was comparable to the one in the main analysis resembling a reversed U-shape. Just as for aircraft and road traffic noise, the ORs were higher than in the main analysis, reaching a maximum of 1.19 (95% CI=1.08–1.31) in the noise category 60 to < 65 dB.

3.4. Combined traffic noise exposure

When all types of traffic noise were included in one model, similar but slightly lower ORs and similar exposure-risk relations across the different exposure categories were found (no table).

Exposure to different combinations of traffic noise sources (Table 5) was examined against a reference group without any type of traffic noise exposure of 40 dB or more (and excluding maximum nightly aircraft noise of 50 dB or more). Resulting risk estimates were higher than for any of the separate traffic noise exposure categories: Combined exposure to all three types of traffic noise resulted in an OR of 1.42 (95% CI=1.33–1.52).

4. Discussion

This large case-control study is the first to assess and directly compare depression risks by aircraft, road traffic and railway noise. We observed a positive relation between “cross-sectional” 2005-traffic noise exposure and the risk of a newly diagnosed depression. The highest risk estimates were observed for individuals simultaneously exposed to high levels of all three types of traffic noise.

4.1. Strengths and limitations

This study uses individual traffic noise estimates and health-claims data to provide risk estimates for different noise levels of aircraft, road and railway traffic noise. It is the first study worldwide to directly compare the risk of depressive disorders for different sources of traffic noise. Traffic noise data was estimated precisely for each participant’s address using state-of the art calculations and digital landscape models. Our main analyses were based on “cross-sectional” 2005

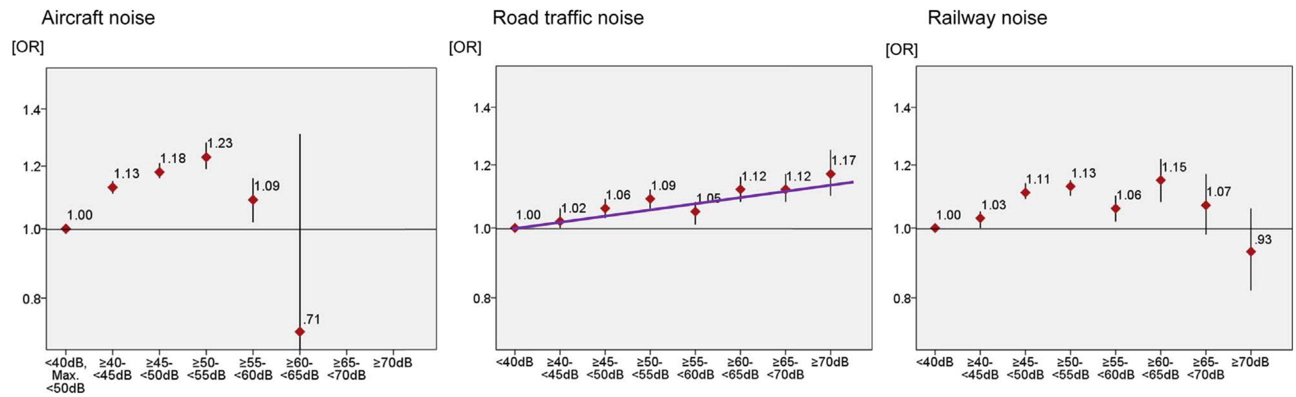

Fig. 2. Dose-response relationship between 24 h sound levels of traffic noise and depression. OR: Odds Ratio; adjusted for age, sex, education, and job title (when available), local proportion of persons receiving unemployment benefits; 95%-CI: 95%-confidence intervals.

exposure. Therefore, moves might have influenced the accuracy of exposures assessment and might have biased the results. For one health insurance company, the insurees' "address history" was known. If only individuals were included in the analysis that had not moved for 5 years or that had not moved for 10 years, this did not have a substantial effect on the risk estimations.

The use of secondary health-claim data reduced the possibility of a selection bias, since all insurees of the three participating health insurance funds were included in the analysis. In contrast, epidemiological studies based on primary data often achieve rather low response rates, being therefore open to substantial selection bias. As a further advantage, health insurance data are collected and stored on a legal basis, ensuring information about consultations, medication, diagnosis, date and duration of hospital stays in a comparable data structure. Therefore, secondary health insurance data allow for the monitoring of diagnosis, medical treatment and course of disease.

Complete longitudinal data are not available for individuals switching to other health insurance companies. However, in Germany rather few older people change their health insurance company, and about 90% of the population is covered by the statutory health insurance. As a fundamental disadvantage of epidemiologic studies based on health insurance data, these data are not collected for research purposes, but for the billing of medical services; therefore, these data only contain billing-relevant data, and persons with private health insurance were not included in the study. By far not all individuals that suffer from "typical" depressive symptoms seek medical help: in a large Norwegian survey including 92,000 individuals aged 20 to 89 years (Roness et al., 2005), only 13% of the individuals having clinical depression according to a self-rating scale (hospital depression and anxiety scale HADS) had sought professional help. Elevated depression "risks" in urban compared with rural living environments might at least partly reflect a higher proportion of "depressive people" seeking medical or psychological help. In principle, potential underdiagnosis or misdiagnosis of clinical depression might be regarded as a weakness of our case acquirement procedure based on physicians' diagnoses. However, potential socioeconomic and regional differences in health-seeking should not have influenced our results, as we adjusted our analyses for regional and individual SES status and for urban versus rural living

Table 3
Traffic noise ($L_{pAeq,24h}$, $L_{pAeq,night}$) and depression

	Aircraft noise				**Road traffic noise**				**Railway noise**			
Exposure	**Cases**	**Control subjects**	**OR**	**95%-CI**	**Cases**	**Control subjects**	**OR**	**95%-CI**	**Cases**	**Control subjects**	**OR**	**95%-CI**
24 h sound levels												
< 40 dB, Max. < 50 dB	28,687	233,178	1.00	–	7,728	62,733	1.00	–	40,213	314,545	1.00	–
< 40 dB, Max. ≥50 dB	4,647	37,668	1.01	0.98–1.04								
≥40 to < 45 dB	24,081	170,171	1.13	1.10–1.15	15,885	124,699	1.02	1.00–1.06	9,652	71,811	1.03	1.00–1.05
≥45 to < 50 dB	13,231	90,227	1.18	1.16–1.21	18,694	138,625	1.06	1.03–1.09	12,929	89,372	1.11	1.09–1.14
≥50 to < 55 dB	5,243	35,784	1.23	1.19–1.28	14,103	101,549	1.09	1.06–1.12	8,925	61,695	1.13	1.10–1.15
≥55 to < 60 dB	1,395	11,043	1.09	1.02–1.16	8,359	62,994	1.05	1.01–1.08	3,362	24,862	1.06	1.02–1.10
≥60 to < 65 dB	11	155	0.71	0.38–1.31	6,648	46,826	1.12	1.08–1.16	1,371	9,371	1.15	1.08–1.22
≥65 to < 70 dB	–	–	–	–	4,540	31,955	1.12	1.08–1.17	556	4,129	1.07	0.98–1.17
≥70 dB	–	–	–	–	1,338	8,865	1.17	1.10–1.25	287	2,461	0.93	0.82–1.06
Continuous (per 10 dB)*							1.037	1.028–1.046 p < 0.001				
Night time period 22–06 h												
< 40 dB, Max. < 50 dB	33,828	268,290	1.00	–	30,420	236,396	1.00	–	39,834	312,270	1.00	
< 40 dB, Max. ≥50 dB	20,990	152,047	1.07	1.05-1.09								
≥40 to < 45 dB	13,819	94,846	1.16	1.13–1.18	15,822	117,229	1.03	1.01–1.05	9,565	70,989	1.03	1.00–1.05
≥45 to < 50 dB	6,358	44,856	1.16	1.13–1.20	12,407	90,574	1.04	1.01–1.06	12,582	85,965	1.12	1.09–1.14
≥50 to < 55 dB	2,234	17,352	1.06	1.01–1.12	8,912	65,645	1.03	1.01–1.06	9,101	63,277	1.12	1.09–1.15
≥55 to < 60 dB	66	855	0.72	0.56-0.93	6,445	45,856	1.07	1.04–1.10	3,840	28,287	1.06	1.02–1.10
≥60 dB	0	0	–	–	3,289	22,546	1.11	1.06–1.15	2,373	17,458	1.07	1.02–1.17

OR: Odds Ratio; adjusted for age, sex, urban living environment, education, and job title (when available), local proportion of persons receiving unemployment benefits; 95%-CI: 95%-confidence intervals; Max.: maximum nightly aircraft noise (for the analysis of aircraft noise; for the analysis of road and railway traffic noise, the reference category included all individuals with 24 h continuous sound levels < 40 dB).
* ORs per 10 dB increase are only given if the linear model is statistically adequate.

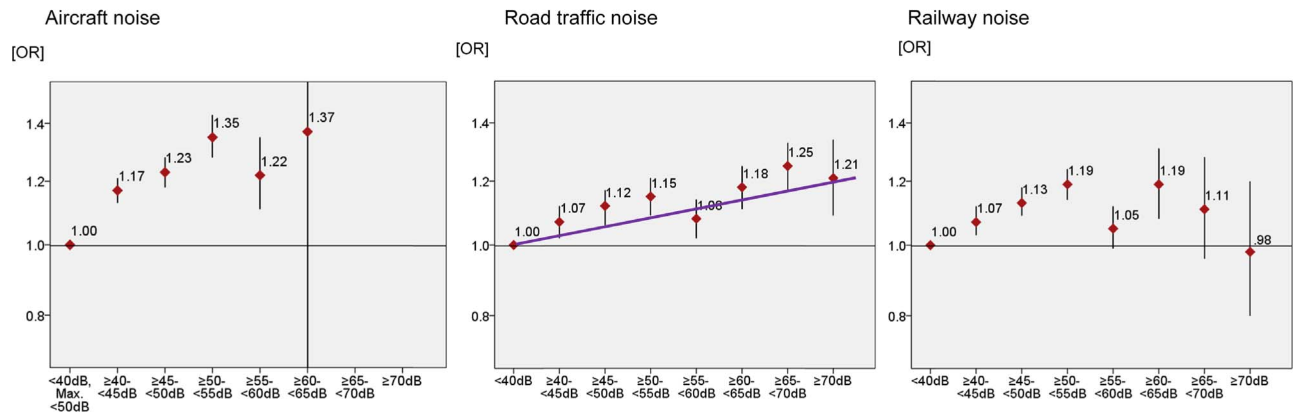

Fig. 3. Dose-response relationship between 24 h sound levels of traffic noise and depression, analysis restricted to persons for whom the individual socioeconomic status was known from the health insurance data (40% of cases, 41% of control subjects). OR: Odds Ratio; adjusted for age, sex, education, and job title (when available), local proportion of persons receiving unemployment benefits; 95%-CI: 95%-confidence intervals.

environment. Moreover, physicians' depression diagnoses might reflect more “objective” and more severe diseases than depression diagnoses according to self-rated scales. Furthermore, a physician's depression diagnosis is the precondition for sickness absence at work for more than three days. We consider our case definition more appropriately reflecting the (socioeconomic) consequences of depressive disorders than depression diagnoses according to self-rated scales would do.

Inclusion criteria for cases were rigid to avoid false positive diagnoses: only individuals who received at least two secure ambulant diagnoses or one hospital discharge diagnosis were included as cases. The utilization of health insurance data and secure diagnoses, determined without knowledge of the exposure-status, prevented the over-reporting of traffic-noise related outcomes often observed in self-reported studies (Stansfeld and Matheson, 2003).

The insurees included in our study represent about 23% of the population aged 40 or above in the study area, so external validity of the results might be questioned. We found that the insured clienteles differed considerably between the participating health insurance funds, particularly with respect to socioeconomic status. Despite these large differences in socioeconomic status, no systematic differences were found when data of the health insurance funds were analyzed separately. This supports the external validity of our results.

Considerable effort was made to test for undetected or residual confounding. An analysis including all three types of traffic noise was conducted to ensure there was no confounding through multiple traffic noise exposure. Furthermore, the analyses were repeated including only cases with available SES-information. The results in this sub-analysis were even more pronounced than in the main analysis, rendering an overestimation of the traffic noise risk by residual SES-confounding improbable. Some studies find different rates of depression among different cultural or ethnical groups. We had no information about cultural background and ethnicity in our dataset, so confounding in this respect cannot be totally excluded.

This study included precise information on exterior sound levels,

Table 4
Traffic noise ($L_{pAeq,24h}$, $L_{pAeq,night}$) and depression, analysis restricted to persons for whom the individual socioeconomic status was known from the health insurance data (40% of cases, 41% of control subjects)

	Aircraft noise				**Road traffic noise**				**Railway noise**			
Exposure	**Cases**	**Control subjects**	**OR**	**95%-CI**	**Cases**	**Control subjects**	**OR**	**95%-CI**	**Cases**	**Control subjects**	**OR**	**95%-CI**
24 h sound levels												
< 40 dB, Max. < 50 dB	10,072	86,378	1.00	–	2,675	23,386	1.00	–	14,593	118,928	1.00	–
< 40 dB, Max. ≥50 dB	1,676	14,350	1.02	0.96–1.08								
≥40 to < 45 dB	9,102	64,971	1.17	1.13–1.21	5,731	46,494	1.07	1.02–1.12	3,659	27,097	1.07	1.03–1.12
≥45 to < 50 dB	4,951	34,294	1.23	1.18–1.28	6,900	51,965	1.12	1.06–1.17	4,743	33,425	1.13	1.09–1.18
≥50 to < 55 dB	2,053	13,690	1.35	1.28–1.43	5,315	38,738	1.15	1.09–1.21	3,402	23,161	1.19	1.14–1.24
≥55 to < 60 dB	550	4,167	1.22	1.11–1.35	3,019	23,680	1.08	1.02–1.14	1,189	9,284	1.05	0.99–1.12
≥60 to < 65 dB	8	62	1.37	0.65–2.91	2,485	18,074	1.18	1.11–1.25	501	3,464	1.19	1.08–1.31
≥65 to < 70 dB	–	–	–	–	1,789	12,167	1.25	1.17–1.33	213	1,595	1.11	0.96–1.28
≥70 dB	–	–	–	–	498	3,408	1.21	1.09–1.34	112	958	0.98	0.80–1.20
Continuous (per 10 dB)[*]							1.052	1.038–1.068 p < 0.001				
Night time period 22–06 h												
< 40 dB, Max. < 50 dB	12,021	99,859	1.00	–	10,949	88,017	1.00	–	14,476	118,220	1.00	–
< 40 dB, Max. ≥50 dB	7,951	57,879	1.11	1.08–1.15								
≥40 to < 45 dB	5,064	35,918	1.18	1.14–1.22	5,868	44,590	1.03	0.99–1.06	3,628	26,806	1.07	1.03–1.11
≥45 to < 50 dB	2,512	17,303	1.27	1.21–1.34	4,583	34,145	1.04	1.01–1.08	4,655	32,230	1.14	1.20–1.18
≥50 to < 55 dB	831	6,588	1.11	1.02–1.20	3,313	24,948	1.05	1.00–1.09	3,357	23,499	1.16	1.11–1.20
≥55 to < 60 dB	33	365	0.90	0.63-1.30	2,472	17,623	1.11	1.05–1.16	1,440	10,602	1.11	1.05–1.18
≥60 dB	0	0	–	–	1,227	8,589	1.12	1.05–1.20	856	6,555	1.07	0.99–1.16

OR: Odds Ratio; adjusted for age, sex, urban living environment, education, and job title (when available), local proportion of persons receiving unemployment benefits; 95%-CI: 95%-confidence intervals; Max.: maximum nightly aircraft noise (for the analysis of aircraft noise; for the analysis of road and railway traffic noise, the reference category included all individuals with 24 h continuous sound levels < 40 dB).

[*] ORs per 10 dB increase are only given if the linear model is statistically adequate.

Table 5
Combined exposure to different sources of traffic noise

	24 h sound levels			
Exposure	**Cases**	**Controls**	**OR**	**95%-CI**
Aircraft, road and railway traffic noise				
< 40 dB traffic noise and Max. aircraft noise < 50 dB	3,994	33,632	1.00	–
≥40 dB at least one source of traffic noise or Max. aircraft noise ≥50 dB	29,408	225,394	1.09	1.05–1.13
≥50 dB aircraft traffic noise, other sources < 50 dB	2,092	15,428	1.15	1.08–1.22
≥50 dB road traffic noise, other sources < 50 dB	25,227	185,502	1.12	1.08–1.16
≥50 dB railway traffic noise, other sources < 50 dB	5,737	44,365	1.08	1.04–1.13
≥50 dB aircraft noise and ≥50 dB road traffic noise	2,073	15,772	1.12	1.06–1.19
≥50 dB aircraft noise and ≥50 dB railway noise	1,076	6,280	1.28	1.19–1.38
≥50 dB road traffic noise and ≥50 dB railway noise	6,280	42,371	1.21	1.16–1.26
≥50 dB aircraft noise and ≥50 dB road traffic noise and ≥50 dB railway noise	1,408	8,544	1.42	1.33–1.52

OR: Odds Ratio; adjusted for age, sex, urban living environment, education, and job title (when available), local proportion of persons receiving unemployment benefits; 95%-CI: 95%-confidence intervals; Max. aircraft noise: maximum nightly aircraft noise.

however, "real" personal noise exposure does not only depend on exterior sound levels, but also on noise insulation, window opening practices, and daily duration and time of stays at home as well as occupational and leisure time noise. While this might limit the assessment of direct effects of traffic noise on health, the exterior noise levels are more easily influenced by public noise protection measures, and, vice versa, establishment of new traffic noise sources such as airport runways, motorways, or railways.

In the literature, there is some evidence for an effect of ambient air pollution on depression (Tzivian et al., 2015; Kim et al., 2016; Pun et al., 2016). Unfortunately, in our study we had no information about air pollution; we therefore were not able to adjust for it. However, while air pollution is considerably associated with road traffic, this is rarely the case for aircraft and railway traffic. We therefore regard lack of adjustment for air pollution as an improbable explanation for our elevated depression risks for aircraft noise and railway traffic noise.

Our dataset did not include any lifestyle variables. Recently published results of the Women's Health Study (Chang et al., 2016) found (in accordance with other studies) an association of depression risks with smoking, lack of activity, and (less clearly) body mass index. However, to constitute a confounder, risk factors must have a causal influence on the exposure level. It is, for example, rather improbable that smoking behavior should have considerably influenced the choice of the home area. Moreover, pre-depressive symptoms as well as previous depressive episodes might have had an influence on lifestyle, so adjustment for these lifestyle factors could have introduced a "cause-and-effect" bias: Adjustment for these consequences of disease could lead to an attenuation of real effects.

4.2. Comparing different types of traffic noise

While the risk-exposure relation for aircraft and railway noise is shaped like a reversed U, the risk estimates for road traffic noise follow a linear trend. This might at least partly be explained by different acoustic qualities of road traffic noise that may only start to have a comparable effect at a higher exposure level – a reversed U-shape might have been found for even higher exposure categories. This is supported by the different steepness of increase in risk estimates: for road traffic noise, the increase of risk with exposure increments is comparably smaller in the lower exposure categories. Depression risk in the highest exposure category of road traffic noise (≥70 dB) is comparable to the risk for aircraft noise at 50 to < 55 dB (the exposure category with the highest risk estimates). Moreover, there are several acoustic and psychosocial variables that influence peoples' perceptions of and reactions to noise beyond amplitude (Marquis-Favre et al., 2005). Two of these factors are fluctuation strength and regularity of noise (Fastl, 1997). More regular, continuous sounds such as road traffic noise lead to stronger habituation, and are experienced as less disruptive compared to railway and aircraft noise, both characterised by more irregular, disruptive events. Each new train or plane coming along might create a new, consciously perceived disturbance (Marquis-Favre et al., 2005; Fastl, 1997; Namba et al., 1996). This is supported by the finding of increased risk estimates for the subcategory of individuals with a continuous nightly aircraft noise exposure below 40 dB, but more than five single noise events above 50 dB at night time: in this category the exposure to few, disruptive events leads to higher risk estimates.

Another potential explanation for the decreasing risks at high traffic noise levels is that vulnerable people might start to actively react to noise exposure at higher noise levels (Stansfeld and Matheson, 2003). Thus, there might be a threshold of noise exposure at which people affected in ways that might lead to depression (e.g. who suffer from insomnia or higher annoyance and stress levels) choose active strategies to minimise their exposure to this noise. These active strategies could be measures such as reducing the interior noise through better insulation, or even moving away from highly noise-exposed areas or not moving there in the first place. This might be called a "healthy resident effect", similar to the healthy worker effect that has frequently been reported in occupational medicine (Checkoway et al., 2004): only people who are not overly sensitive to noise and not prone to experience noise-induced stress, annoyance and insomnia remain in the highest-noise exposure areas. Avoiding negative outcomes by active coping might be less common for road noise for two reasons. Firstly, the threshold beyond which an individual choses to actively cope might be higher since the quality of road traffic noise might be perceived as less disturbing. Secondly, disruptions through road traffic noise might be less noticeable due to its continuous qualities (Marquis-Favre et al., 2005).

4.3. Combination of different types of traffic noise

The highest risk estimates were found for a combined exposure to different sources of traffic noise. This is very relevant for public health, since many people are exposed to a combination of different sources of traffic noise. These results are also relevant with respect to "environment justice": only 0.3% of individuals in the areas with the lowest unemployment rates, but 3.9% of individuals in the "poorest" areas were simultaneously exposed to high levels (≥50 dB) of all three types of traffic noise. According to our study results, people simultaneously exposed to high levels of aircraft, road and railway traffic noise have a considerably increased risk of depression. This disproportionate exposure to combined traffic noise, may be exposing an already vulnerable population (due to their limited socioeconomic resources) at further risk of depression.

This result also raises the issue of autocorrelation which so far has not been examined extensively in environmental noise studies, including our work. Even though limitations of the approach have been described (Paciorek, 2010; Lee and Sarran, 2015), future studies should aim to employ methods to account for spatial autocorrelation as this may help to reduce residual confounding. As our dataset was anonymized after linking individual health insurance data with address-specific noise data, we were not able to control for spatial autocorrelation.

4.4. Potential pathophysiological pathways

There are several potential pathways that could explain the observed association between traffic noise exposure and risk of an incident depressive disease. One possible mechanism is a reduction in quantity and quality of sleep. Traffic noise has previously been linked to insomnia symptoms (i.e., difficulties falling asleep, waking up frequently, waking up too early, nonrestorative sleep) and physiological symptoms of decreased sleep quality (Halonen et al., 2012), and sleep disturbance has been linked to depression risks (Franzen and Buysse, 2008). Additionally, there is ample evidence showing that traffic noise exposure causes annoyance and heightened biological stress reactions (Marquis-Favre et al., 2005; Vallet et al., 1983; Babisch, 2002). These outcomes in turn have been linked to depression (Wager-Smith and Markou, 2011; Anisman and Merali, 2002). Particularly stress that is perceived as non-controllable (which is the case for stress caused by traffic noise) might have aversive effects on mental health (Johnson and Sarason, 1978; Ghorbani et al., 2008).

For aircraft noise, we considered maximum nightly sound pressure levels separately from continuous sound pressure levels. This is in line with the hypothesis of sleep disturbance constituting an important pathophysiological link between aircraft traffic noise and depression risk. Analyzing the night time period, we found a significantly increased depression risk (OR=1.07; 95% CI 1.05–1.09; Table 3) in this newly formed category. The depression risk further increased to 1.11 (95% 1.08–1.15; Table 4) when the analysis was restricted to individuals with known socioeconomic status. These results suggest that – comparable to our results for heart failure and hypertensive heart disease (Seidler et al. 2016c) – nightly maximum sound pressure levels exceeding 50 dB lead to increased depression risks from aircraft noise even if continuous sound pressure levels are below 40 dB.

4.5. Relevance and implications

Depression constitutes a major public health burden (Ferrari et al., 2013), and high proportions of the population are exposed to traffic noise. Thus, the – albeit relatively small – increases in relative risks translate into a large absolute number of additional depression cases attributable to traffic noise exposure, assuming the observed association is causal. Therefore, the findings are of high public-health relevance. Traffic noise protection is a public duty, and the present findings indicate that better noise protection could help to reduce the incidence of depressive disorders. According to our study results, particular attention should be paid to residential areas with combined sources of traffic noise.

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Conflict of interest

None.

Approval of ethics committee and data protection commissioners

We complied with the comments of the Ethics Committee of the Medical Faculty, TU Dresden (AZ: EK328102012; 21 February 2013 and 22 April 2014). The Federal Commissioner for Data Protection and Freedom of Information (AZ: III-320/010#0011; reply of 11 June 2012) and the Data Protection Commissioners of the German states Hesse (AZ: 43.60-we; reply of 13 March 2012; amendments 7 February 2014) and Rhineland-Palatinate (AZ: 6.08.22.002; reply of 7 May 2012; amendments 4 February 2014) approved the study concept. These authorities confirmed, that the research project is, in principle, in accordance with data protection regulations.

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References

Abey-Wickrama, I., A'Brook, M., Gattoni, F., Herridge, C.F., 1969. Mental-hospital admissions and aircraft noise. Lancet 294, 1275–1277.

Anisman, H., Merali, Z., 2002. Cytokines, stress, and depressive illness. Brain Behav. Immun. 16, 513–524.

Babisch, W., 2002. The noise/stress concept, risk assessment and research needs. Noise Health 4, 1–11.

Baglioni, C., Battagliese, G., Feige, B., Spiegelhalder, K., Nissen, C., Voderholzer, U., Lombardo, C., Riemann, D., 2011. Insomnia as a predictor of depression: a meta-analytic evaluation of longitudinal epidemiological studies. J. Affect. Disord. 135, 10–19.

Bundesregierung, 2006. Verordnung zur Durchführung des Bundes-Immissionsschutzgesetzes (Verordnung über die Lärmkartierung). Bundesanzeiger 154a, 6, vom 6. März 2006 (34. BImSchV (2006)).

Bundesregierung, 2008. Erste Verordnung über die Datenerfassung und das Berechnungsverfahren für die Festsetzung von Lärmschutzbereichen vom 27. Dezember 2008 (1. FlugLSV). Anlage 2: Anleitung zur Berechnung von Lärmschutzbereichen (AzB). Bundesanzeiger 195a, 2.

Chang, S.C., Pan, A., Kawachi, I., Okereke, O.I., 2016. Risk factors for late-life depression: a prospective cohort study among older women. Prev. Med. 91, 144–151.

Checkoway, H., Pearce, N., Kriebel, D., 2004. Research Methods in Occupational Epidemiology. Oxford University Press, USA.

DGPPN, BÄK, KBV, AWMF, AkdÄ, BPtK, BApK, DAGSHG, DEGAM, DGPM, DGPs, DGRW, 2015. S3-Leitlinie/Nationale Versorgungsleitlinie Unipolare Depression–Langfassung, 2nd edition. Version 3. ⟨www.depression.versorgungsleitlinien.de⟩ (last accessed on 29.08.16.).

Elovainio, M., Pulkki-Råback, L., Hakulinen, C., Ferrie, J., Jokela, M., Hintsanen, M., Raitakari, O., Keltikangas-Järvinen, L., 2015. Childhood and adolescence risk factors and development of depressive symptoms: the 32-year prospective Young Finns follow-up study. J. Epidemiol. Commun. Health 69, 1109–1117.

Elovainio, M., Pulkki-Råback, L., Jokela, M., Kivimäki, M., Hintsanen, M., Hintsa, T., Viikari, J., Raitakari, O., Keltikangas-Järvinen, L., 2012. Socioeconomic status and the development of depressive symptoms from childhood to adulthood: a longitudinal analysis across 27 years of follow-up in the Young Finns study. Soc. Sci. Med. 74, 923–929.

European Union, 2002. Directive 2002/49/EC of the European Parliament and of the Council of 25 June 2002 relating to the assessment and management of environmental noise – Declaration by the Commission in the Conciliation Committee on the Directive relating to the assessment and management of environmental noise. ⟨http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32002L0049⟩ (last accessed on 29.08.16.).

Fastl, H., 1997. The psychoacoustics of sound-quality evaluation. Acta Acust. United Acust. 83, 754–764.

Ferrari, A.J., Charlson, F.J., Norman, R.E., Patten, S.B., Freedman, G., Murray, C.J., Vos, T., Whiteford, H.A., 2013. Burden of depressive disorders by country, sex, age, and year: findings from the global burden of disease study 2010. PLoS Med. 10, e1001547.

Floud, S., Vigna-Taglianti, F., Hansell, A., Blangiardo, M., Houthuijs, D., Breugelmans, O., Cadum, E., Babisch, W., Selander, J., Pershagen, G., Antoniotti, M.C., Pisani, S., Dimakopoulou, K., Haralabidis, A.S., Velonakis, V., Jarup, L., HYENA Study Team, 2011. Medication use in relation to noise from aircraft and road traffic in six European countries: results of the HYENA study. Occup. Environ. Med. 68, 518–524.

Franzen, P.L., Buysse, D.J., 2008. Sleep disturbances and depression: risk relationships for subsequent depression and therapeutic implications. Dialogues Clin. Neurosci. 10, 473–481.

Ghorbani, N., Krauss, S.W., Watson, P.J., LeBreton, D., 2008. Relationship of perceived stress with depression: Complete mediation by perceived control and anxiety in Iran and the United States. Int. J. Psychol. 43, 958–968.

Greiser, C., Greiser, E., 2010. Risikofaktor nächtlicher Fluglärm: Abschlussbericht über eine Fall-Kontroll-Studie zu kardiovaskulären und psychischen Erkrankungen im Umfeld des Flughafens Köln-Bonn. Umweltbundesamt, Anlagenband.

Greiser, E., Greiser, C., Janhsen, K., 2007. Night-time aircraft noise increases prevalence of prescriptions of antihypertensive and cardiovascular drugs irrespective of social class—the Cologne-Bonn Airport study. J. Public Health 15, 327–337.

Halonen, J.I., Vahtera, J., Stansfeld, S., Yli-Tuomi, T., Salo, P., Pentei, J., Kivimäki, M.,

Lanki, T., 2012. Associations between nighttime traffic noise and sleep: the Finnish public sector study. Environ. Health Perspect. 120, 1391–1396.
Halpern, D., 2014. Mental Health and the Built Environment: More than Bricks and Mortar?. Routledge, London & New York.
Hiramatsu, K., Yamamoto, T., Taira, K., Ito, A., Nakasone, T., 1997. A survey on health effects due to aircraft noise on residents living around Kadena air base in the Ryukyus. J. Sound Vib. 205, 451–460.
Jenkins, L., Tarnopolsky, A., Hand, D., 1981. Psychiatric admissions and aircraft noise from London Airport: four-year, three-hospitals' study. Psychol. Med. 11, 765–782.
Johnson, J.H., Sarason, I.G., 1978. Life stress, depression and anxiety: Internal-external control as a moderator variable. J. Psychosom. Res. 22, 205–208.
Kim, K.N., Lim, Y.H., Bae, H.J., Kim, M., Jung, K., Hong, Y.C., 2016. Long-term fine particulate matter exposure and major depressive disorder in a community-based urban cohort. Environ. Health Perspect. 124, 1547–1551.
Kishikawa, H., Matsui, T., Uchiyama, I., Miyakawa, M., Hiramatsu, K., Stansfeld, S.A., 2009. Noise sensitivity and subjective health: questionnaire study conducted along trunk roads in Kusatsu, Japan. Noise Health 11, 111–117.
Lee, D., Sarran, C., 2015. Controlling for unmeasured confounding and spatial misalignment in long-term air pollution and health studies. Environmetrics 26, 477–487.
Marquis-Favre, C., Premat, E., Aubrée, D., 2005. Noise and its effects–a review on qualitative aspects of sound. Part II: noise and annoyance. Acta Acust. United Acust. 91, 626–642.
Meecham, W., Smith, H., 1977. Effects of jet aircraft noise on mental hospital admissions. Br. J. Audiol. 11, 81–85.
Möhler, U., Liepert, M., Mühlbacher, M., Beronius, A., Nunberger, M., Braunstein, G., Brillé, G., Schaal, J., Bartel, R., 2015. Dokumentation zur Berechnung der akustischen Daten in NORAH. NORAH (Noise-related annoyance, cognition and health): verkehrslärmwirkungen im Flughafenumfeld Endbericht. Band 2⟨http://www.laermstudie.de/fileadmin/files/Laermstudie/Akustik_Wiss_Ergebnisbericht.pdf⟩, (last accessed on 29.08.16).
Möhler, U., Liepert, M., Mühlbacher, M., Klatte, M., Vogelgesang, B., Thomann, G., 2016. The acoustic basis of the NORAH field studies. 45th International Congress on Noise Control Engineering (INTER-NOISE 2016). ⟨http://www.norah-studie.de//de/publikationen.html?file=files/norah-studie.de/Downloads/Moehler_etal_2016_Acoustic%20basis_IN2016-288.pdf⟩ (last accessed 29.10.16.).
Namba, S., Kuwano, S., Katorade, T., 1996. Trade-off effect between the number of events and LAEQ in the evaluation of train noise. Int. Congr. Noise Control Eng., (INTERNOISE 1996).
Orban, E., McDonald, K., Sutcliffe, R., Hoffmann, B., Fuks, K.B., Dragano, N., Viehmann, A., Erbel, R., Jöckel, K.H., Pundt, N., Moebus, S., 2016. Residential road traffic noise and high depressive symptoms after five years of follow-up: results from the heinz nixdorf recall study. Environ. Health Perspect. 124, 578–585.
Paciorek, C.J., 2010. The importance of scale for spatial-confounding bias and precision of spatial regression estimators. Stat. Sci. 25, 107–125.
Peen, J., Schoevers, R., Beekman, A., Dekker, J., 2010. The current status of urban-rural differences in psychiatric disorders. Acta Psychiatr. Scand. 121, 84–93.
Pun, V.C., Manjourides, J., Suh, H., 2016. Association of ambient air pollution with depressive and anxiety symptoms in older adults: results from the NSHAP study. Environ. Health Perspect..
Roness, A., Mykletun, A., Dahl, A.A., 2005. Help-seeking behavior in patients with anxiety disorder and depression. Acta Psychiatr. Scand. 111, 51–58.
Seidler, A., Wagner, M., Schubert, M., Dröge, P., Pons-Kühnemann, J., Swart, E., Zeeb, H., Hegewald, J., 2016b. Myocardial infarction risk due to aircraft, road, and rail traffic noise. Dtsch. Arztebl. Int. 113, 407–414.
Seidler, A., Wagner, M., Schubert, M., Dröge, P., Römer, K., Pons-Kühnemann, J., Swart, E., Zeeb, H., Hegewald, J., 2016c. Aircraft, road and railway traffic noise as risk factors for heart failure and hypertensive heart disease – a case-control study based on secondary data. Int. J. Hyg. Environ. Health. http://dx.doi.org/10.1016/j.ijheh.2016.09.012.
Seidler A., Wagner M., Schubert M., Dröge P., Hegewald J., 2016a. Sekundärdatenbasierte Fallkontrollstudie mit vertiefender Befragung. NORAH (Noise-related annoyance, cognition and health): Verkehrslärmwirkungen im Flughafenumfeld. Endbericht, Band 6, 2nd ed., ⟨http://www.laermstudie.de/fileadmin/files/Laermstudie/NORAH-Fallkontrollstudie_Endbericht_16mai12_Auflage_2.pdf⟩ (last accessed on 25.09.16)
Stansfeld, S.A., Matheson, M.P., 2003. Noise pollution: non-auditory effects on health. Br. Med. Bull. 68, 243–257.
Stansfeld, S.A., Gallacher, J., Babisch, W., Shipley, M., 1996. Road traffic noise and psychiatric disorder: prospective findings from the Caerphilly study. Br. Med. J. 313, 266–267.
Sygna, K., Aasvang, G.M., Aamodt, G., Oftedal, B., Krog, N.H., 2014. Road traffic noise, sleep and mental health. Environ. Res. 131, 17–24.
Tarnopolsky, A., Watkins, G., Hand, D.J., 1980. Aircraft noise and mental health: I. Prevalence of individual symptoms. Psychol. Med. 10, 683–698.
Tzivian, L., Winkler, A., Dlugaj, M., Schikowski, T., Vossoughi, M., Fuks, K., Weinmayr, G., Hoffmann, B., 2015. Effect of long-term outdoor air pollution and noise on cognitive and psychological functions in adults. Int. J. Hyg. Environ. Health 218, 1–11.
Vallet, M., Gagneux, Clairet, J., Laurens, J., Letisserand. D., 1983. Heart rate reactivity to aircraft noise after a long term exposure. Noise as a Public Health Problem. In: G. Rossi (Ed.), Milan: Centro Recherche e Studio Amplifon, pp. 965–975.
Wager-Smith, K., Markou, A., 2011. Depression: a repair response to stress-induced neuronal microdamage that can grade into a chronic neuroinflammatory condition? Neurosci. Biobehav. Rev. 35, 742–764.
WHO, 2010. Burden of Disease from Environmental Noise: Quantification of Healthy Life Years Lost in Europe. WHO: European Centre for Environment and Health, Bonn.
WHO, 2015. Depression fact sheet 5 2015. ⟨http://www.who.int/mediacentre/factsheets/fs369/en/⟩ (last accessed on 29.08.16.).