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Associations between road, rail and aircraft traffic noise with cognitive function in the UK Biobank cohort

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ABSTRACT

Background: Studies on the associations between noise pollution and cognitive function in adults remain few. Here, we examine the cross-sectional associations between traffic noise with cognitive performance in the UK Biobank cohort.

Methods: Mid-to-older aged adults recruited during 2006–2010 were included in the analyses for road (N = 499,717), rail (N = 228,079) and aircraft (N = 105,768) noise exposures. Address-level average road noise for 2013 from minor and major roads were modelled using an enhanced CNOSSOS-EU framework; address-level average rail noise estimates for 2011 from major rail corridors were obtained from the government Department for Environment, Food and Rural Affairs; average aircraft noise estimates at postcode-level for 2011 were modelled using the ANCON model by Civil Authority Aviation for participants residing in one of 44 districts partially or wholly encompassed with weighted 24-hour day-evening-night (L_{den}) aircraft noise contours. Reaction time, visuospatial memory, verbal-numerical reasoning, and prospective memory were self-administered via touchscreen at baseline. Regression models were used to test associations between each traffic noise (L_{den} and L_{night}) and each cognitive domain, allowing for covariates adjustment and correction for multiple testing.

Results: Exposure to higher L_{night} aircraft (≥55 dB vs. <45 dB), and L_{den} aircraft (≥60 dB vs. <50 dB) was associated with 133 % (95 %CI: 66 %–229 %) and 48 % (95 %CI:17 %–86 %) higher error rate respectively in the visuospatial memory test; for rail L_{night} and L_{den}, the respective figures were 68 % (95 %CI: 29 %–118 %) and 39 % (95 %CI: 10 %–75 %). There were no convincing associations of road traffic noise with visuospatial memory performance. No associations were found of reaction time, verbal-numerical reasoning, or prospective memory with any traffic-source noise.

Conclusion: Higher aircraft and rail traffic noise exposure was associated with poorer visuospatial memory test performance, indicating a potential role in cognitive impairment in adults.

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1. Introduction

Road vehicles, trains and aircrafts are major sources of traffic noise. Exposure to traffic noise, both in short- and long-term timeframes, can contribute to a range of adverse health effects, including sleep disturbances (Gong et al., 2024), cardiovascular diseases (Münzel et al., 2024), impaired cognition (Hahad et al., 2022) and Alzheimer's disease (Cantuaria et al., 2021; Havyarimana et al., 2025). It is generally regarded that noise exposure, through both direct and indirect pathways, can lead to a stress response that activates the autonomic and endocrine systems (Babisch, 2003). The indirect pathway in particular can trigger cognitive and emotional responses in some regions of the brain via disturbances of activities, sleep, communication and annoyance (Hahad et al., 2022).

Only a few studies have investigated the association between traffic noise exposure and cognitive impairment in adult populations. These studies have sample sizes ranging from approximately 1600 to 10,800, reporting mixed results. Cross-sectional analyses from Heinz Nixdorf Recall Study in Germany found that higher exposure to road traffic noise was associated with mild cognitive impairment (MCI, a stage between normal cognitive changes in ageing and early dementia) (Tzivian et al., 2016a,b), but no association was found in the longitudinal analyses with cognitive decline (Ogurtsova et al., 2023). Similarly, another study in Chicago reported a negative, cross-sectional association between community noise exposures and global cognitive performance but no association was found with cognitive decline in longitudinal analyses (Weuve et al., 2021). Some studies have reported that air pollution and road traffic noise may act synergistically to impact cognitive function (Tzivian et al., 2017; Yu et al., 2023), whilst a study in Ireland found the association between road traffic noise and cognitive function was largely confounded by air pollution (Mac Domhnaill et al., 2021). To date, the evidence on the impact of road traffic noise on cognitive function remains weak and we know little about how rail and aircraft traffic noise affects cognitive function in adult populations (Wu et al., 2024).

We previously reported that higher exposure to road traffic noise was associated with incident Alzheimer's disease in the UK Biobank (Havyarimana et al., 2025) whilst another study using the same cohort reported that cognitive task performance at baseline is of predictive value of incident dementia (Calvin et al., 2019). This cross-sectional study aims to investigate the impact of traffic noise pollution, specifically road, rail and aircraft, on cognitive function in middle-aged and older adults participating in the UK Biobank cohort. Noise from road, rail and aircraft traffic differs by sound characteristics in terms of the rise time, frequency and duration (Basner et al., 2011). Each traffic noise may differentially cause annoyance, stress response and noise perception to various degrees, leading to varying strengths of epidemiological associations with different health outcomes (Brink et al., 2019). We therefore hypothesised that different traffic noise sources have differential impacts on different cognitive domains.

2. Methods

2.1. Study population

This study used baseline data of participants recruited in UK Biobank. Following postal invitations to over 9.2 million individuals who registered with National Health Service (NHS), over 500,000 men and women aged 40–69 years living within 25 miles of one of the 22 assessment centres across England, Scotland and Wales were recruited (5.5 % response rate) between 2006 and 2010 (Sudlow et al., 2015; Fry et al., 2017). Upon recruitment, participants completed touchscreen questionnaires (i.e., participants directly interacted with the dedicated computer screen to answer the questions at the assessment centre) to provide information on their sociodemographic, lifestyle factors and medical history, and to perform cognitive function tests. Physical measurements such as height, weight and blood pressure as well as blood samples were also collected by trained and certified registered nurses at the assessment centre. All participants provided informed consent at the time of recruitment.

This study was conducted using UK Biobank ethical approval (REC: 21/NW/0157) under approved project application 59129.

2.2. Cognitive function assessment

The cognitive function assessment at baseline in the UK Biobank was designed to assess cognitive function of large populations. The assessment was self-administered using a touchscreen questionnaire, with the scoring being generated automatically (Supplementary Table 1). Almost all participants completed the Reaction time test and Visuospatial memory test. The Verbal-numerical reasoning test and Prospective memory test were added to the battery test part way through the recruitment period, therefore only a subset of the participants completed these two tests (Campbell and Cullen, 2023). For this reason, sample size varied for different cognitive domains in this study. We used Reaction time test and Visuospatial memory test as the main outcomes given the relative completeness of data.

Reaction time – this was calculated using the mean duration to first press the snap-button when both cards matched. The mean reaction time was recorded in milliseconds (ms) and rounded to the nearest whole number; higher values denote longer reaction time.

Visuospatial memory – this was assessed using a pairs matching game, whereby the number of errors made by participants was recorded, therefore higher error counts indicate worse visuospatial memory.

Verbal-numerical reasoning – a participant's ability to solve verbal and numerical reasoning problems was quantified using a fluid intelligence score on a scale ranging from 0 to 13, where a higher score indicates better performance.

Prospective memory – this was measured by asking participants to remember and act on an instruction after a filled delay. For the purpose of analyses, the results for prospective memory were categorised dichotomously as 0 for instruction not recalled, either skipped or incorrect, and 1 for correct response on the first or second attempt.

2.3. Noise exposure assessment

For all participants in England, Scotland and Wales, the annual average road traffic noise estimates were modelled at baseline home addresses for 2013 using the Common Noise Assessment Methods in Europe (CNOSSOS-EU) in accordance to European Noise Directive 2002/49/EC. In our study, CNOSSOS-EU model was enhanced by considering traffic flow on both major and minor roads based on the protocol described in (Morley et al., 2015; Morley and Gulliver, 2016). Annual mean A-weighted sounds pressure level in decibels (dB[A]) were modelled using Annual Average Daily Traffic (AADT) counts and traffic speeds for major (heavily trafficked) roads within 1 km and minor (connection and residential) roads within 100 m. The ray-traced source to receiver sound propagation paths empirically adjusted for the surface roughness of land cover, building heights, and annual-average meteorology (wind profiles and air temperatures).

Annual average rail noise metrics in 2011 were only available for participants in England who lived along major railway corridors, defined as railway lines running more than 30,000 passenger vehicle trips per year. Baseline home addresses points were spatially joined to the appropriate cells from the 10 m rail noise exposure map by Extrium on behalf of the UK Government Department for Environment, Food and Rural Affairs (DEFRA) (Extrium, 2015).

The Civil Aviation Authority (CAA) provided modelled aircraft noise estimates in 2011 from version 2 of the Aircraft Noise CONtour (ANCON) model as noise contours for four major English airports: Birmingham (BHX), London Heathrow (LHR), London Gatwick (LGW) and Manchester (MAN). ANCON considers the average flight path heights and speed profiles, aircraft performance characteristics, and take-off/

landing weights (Ollerhead et al., 1999). Point locations from the ANCON model were converted to mapped surfaces with a resolution of 100 m for the four airports (Topriceanu et al., 2025). These 100 m aircraft noise raster surfaces around the four airports were then intersected by the postcode centroids of a subset of participants residing in one of 44 districts partially or wholly encompassed with CAA weighted 24-hour day-evening-night aircraft noise contours.

For each traffic noise source, we used noise indicators L_{den} (day-evening-night equivalent level; A-weighted average 24-hour noise sound level, with a 5 dB penalty for evening hours 19:00–23:00 and 10 dB penalty for night hours 23:00–7:00) and L_{night} (night equivalent level; average sound pressure level overnight 23:00–07:00) for the analyses. For aircraft noise, CAA noise contours were provided for L_{night} levels ≥45 dB and L_{den} levels ≥50 dB, with continuous measures available above these thresholds. Continuous values below 45 dB (L_{night}) and 50 dB (L_{den}) were not provided by CAA due to concern of accuracy of lower noise levels and were therefore grouped into <45 dB and <50 dB categories, respectively.

2.4. Study covariates

Covariates were selected based on a direct acyclic graph (Supplementary Fig. 1). The covariates included age, sex, socio-economic status (SES), ambient air pollution and greenspace (for further details, see Supplementary Table 2). Age was calculated based on a participant's date of birth and the first date of visit to the assessment centre and was recorded in whole years. Sex was self-reported as male or female or acquired from the NHS central registry at recruitment.

Area-level SES was derived from the Townsend deprivation index, which was calculated by UK Biobank immediately prior to participant joining UK Biobank. The score was based on the previous national census output areas, with each participant having an assigned score depending on their residential postcode. Townsend indices were reported from 1 (least deprived) to 5 (most deprived). Individual-level SES included level of education (degree level or not), employment status (paid employment or not) and household income (<£31,000 vs.≥ £31,000), which were all self-reported via questionnaire at baseline assessment and were dichotomised in the analysis. Sleep duration was categorised as <6 h, ≥6–8 h, and ≥8 h. Sleeplessness was categorised (never/rarely, sometimes, usually) based on the answer on the question whether the study participant had trouble falling asleep at night or woke up in the middle of the night.

Particulate matter with a diameter of <2.5 μm ($PM_{2.5}$), and light absorbance of $PM_{2.5}$ ($PM_{2.5abs}$) – as a proxy of black carbon – were modelled for each address for 2010 using Land Use Regression (LUR) model developed for the European Study of Cohorts for Air Pollution Effects (ESCAPE) (Eeftens et al., 2012). Land use data for greenspace percentage were obtained from Generalised Land Use Database for England (GLUD) for the year 2005. For this study, the percentage of UKB home locations which are within 300 m of a greenspace covering more than 20 % of land-use was used as cut-off point to define closeness to greenness. The use of 20 % as cut-off point was informed by previous literature (Cantuaria et al., 2021).

Higher exposure to traffic noise is associated with poor cardiovascular health (Huang et al., 2023; Kupcikova et al., 2021; van Kempen et al., 2018; Hansell et al., 2013; Saucy et al., 2021), which is also a strong predictor of cognitive impairment (van Nieuwkerk et al., 2023). Using the available data in UK Biobank, we constructed the cardiovascular risk score (CRS) which was originally proposed by the American Heart Association in 2010 for primordial prevention of cardiovascular disease(Lloyd-Jones et al., 2010). Since then, CRS (or the Life's Simple 7) is a widely-used metric to define one's cardiovascular health and studies have shown that CRS was associated with dementia incidence (Sabia et al., 2019). CRS integrates seven risk factors (i.e., individual metric) for cardiovascular disease: smoking, body mass index (BMI) (kg/m^2), physical activity (min/week), healthy diet score, total cholesterol (mmol/L), blood pressure (mmHg) and blood glucose (mmol/L). Each individual metric was scored as ideal (2 points), intermediate (1 point) or poor (0 points) (Supplementary Table 3) (Lloyd-Jones et al., 2010). The total CRS is the sum of individual metric point values which ranged from 0 to 14 (14-scale CRS). The overall CRS was reported as high (11–14 points), moderate (7–10 points), or low (0–6 points), with higher CRS indicating more optimal cardiovascular health.

2.5. Statistical analysis

Descriptive analyses on population characteristics, cognitive function and noise distributions were conducted.

Both road and rail traffic noise metrics were analysed on a continuous (per 10 dB) and categorical scale (assuming linear and non-linear effects). Aircraft noise was only analysed on a categorical scale as continuous values below 45 dB (L_{night}) and 50 dB (L_{den}) were not provided by CAA as previously described. For all three traffic noise sources, noise exposure was categorised as <50 dB (reference), ≥50–55 dB, ≥55–60 dB, and ≥60 dB for L_{den} and <45 dB (reference), ≥45–50 dB, ≥50–55 dB, and ≥55 dB for L_{night}. We set 50 dB as reference for L_{den} because this may be the potential threshold above which adverse health impacts of noise start to be observed (Vienneau et al., 2015). We set the 45 dB as reference for L_{night}, following the WHO environmental noise guidelines (WHO, 2018).

Regression analyses were used to examine the cross-sectional associations between the different noise exposure metrics and cognitive function outcomes at baseline. To investigate the association between traffic noise exposure and reaction time or verbal-numerical reasoning, multiple linear regression analyses were performed, with results expressed as a beta (β) coefficient (as change in reaction time in milliseconds or change in fluid intelligence scores per 10 dB higher annual average road or railway noise exposure). Given the discrete nature of the visuospatial memory test, a Poisson regression was used to investigate an association with each of the traffic noise metric and the results are reported as prevalence rate ratio (PRR). Logistic regression was used to determine the association between noise exposures and prospective memory (those who could not recall the instructions, skipped or had incorrect responses are compared with those had correct responses at the first or second attempt as a reference group) due to the binary nature of test score, and results are reported as odds ratio (OR) – higher ORs indicate poorer performance of test. Furthermore, to compare the results from the different cognitive function test scores, raw scores were converted to z-score, except for prospective memory as it was captured as a categorical variable. The mean score is approximately zero and the standard deviation approximately one, with higher z-score indicating longer reaction time, worse visuospatial memory and better fluid intelligence.

Three models were generated for each noise exposure-outcome pair. Each noise exposure was modelled separately (single noise exposure model). The first model was the unadjusted model, followed by model 1 adjusted for age, sex, socioeconomic status and cardiovascular risk score (14-scale CRS) and the main model which adjusted for model 1 variables, $PM_{2.5}$ and greenspace. All covariables included in the modelling stage were within reasonable range of multicollinearity (variation inflation factor (VIF) = 1.0–1.9). Since there were a total of 60 regression models that covered four cognitive function domains (both continuous and categorical noise threshold comparisons), *p*-values (two-tail) from the traffic noise coefficient were adjusted for multiple comparison using the Simes-Benjamini-Hochberg (SBH) false discovery rate (FDR) method (Benjamini and Hochberg, 1995). SBH method is a less conservative approach in balancing the control for the rate of false discoveries while allowing for detection of effects that are truly significant. It is also a flexible approach for the large number of hypotheses being tested as in our study. Statistical significance was set at FDR p-value of 0.05.

We conducted several sensitivity analyses on the statistically significant associations from the main model to test their robustness. First, given the strong evidence indicating the negative impacts of both stroke and dementia on cognitive function, all regression analyses were repeated in two separate sensitivity analyses: participants with a history of stroke were excluded while retaining those with dementia; participants with dementia were excluded while retaining those with stroke. Second, we re-ran the main model without including CRS as a covariate given that CRS is potentially on the causal pathway between noise exposure and cognitive function. Third, we adjusted for $PM_{2.5abs}$ in the main model instead of $PM_{2.5}$ given that $PM_{2.5abs}$ is often used as a proxy of traffic-related air pollution. Fourth, we repeated the analyses by restricting to participants who had data for all four cognitive function tests. Finally, to avoid potential selection bias, we repeated all the analyses relating to rail noise by including participants whose residential addresses are not covered by the Extrium rail noise exposure map, assuming they have zero rail noise exposures. Similarly, we repeated all the analyses relating to aircraft noise by including participants whose residential addresses are not covered by the ANCON model, assuming these participants do not live near airport or under flight path and hence has <45 dB for L_{night} and <50 dB for L_{den} aircraft noise.

To address the issue of missing data, we also conducted complete-case analyses, i.e., restricting the study sample to those who had complete data on noise exposure, covariates and cognitive function outcomes and by re-running all the models.

For the cognitive tests that were statistically significantly associated with the respective traffic noise source, subgroup analyses were carried out stratifying by age (<60, ≥60 years old), sex, tertiles of $PM_{2.5}$(<9.9, ≥9.9–10.6, and ≥10.6 μg/m^3), closeness to greenspace (<20 %, and ≥20 %).

All data analyses were performed using Stata (18.0, StataCorp LLC, College Station, TX) and RStudio (2023.03.0 + 386).

3. Results

3.1. Study population characteristics

Of the full cohort (n = 502,416), 98.5 % of participants had at least one modelled noise estimate and one complete cognitive function test. As shown in Fig. 1, the final analytical sample varied depending on data availability for each traffic noise metric and cognitive domain. 499,717 (99 %) had modelled estimates for road traffic noise, whilst only 228,079 (45 %) and 105,768 (21 %) participants had data on rail and aircraft noise, respectively. Over 98 % of the cohort completed the reaction time and visuospatial memory tests, whilst 33 % and 34 % of the cohort also additionally completed the verbal-numeric reasoning and prospective memory test, respectively. Those who had data on all four cognitive function tests tended to have higher education levels, higher household income and better cardiovascular health, compared with those who had incomplete data of the four tests (Supplementary Table 4).

The mean age was 57 years old (SD 8.1) with 54 % of participants being female (Table 1). The majority of study participants were of White ethnicity (94.6 %), and in employment (96.8 %). 10 % of the cohort were current smokers with 35 % of study participants being classified to have a poor cardiovascular health (CRS <6 points) and an overall 1.1 % prevalence of stroke and 1.5 % of all-cause dementia.

Among the study population who completed the tests on reaction time and visuospatial memory (>98 % of the full cohort), the median road noise exposure was 52 dB for L_{den} and 45 dB for L_{night}. The respective figures for aircraft noise were 54 dB for L_{den} and 49 dB for L_{night}. The median rail noise (L_{den} and L_{night}) exposures were much lower (31 dB for L_{den} and 24 dB for L_{night}) compared to those of road and aircraft noise (Table 1). Median annual-average concentration was 9.8 μg/m^3 for $PM_{2.5}$.

3.2. Associations between traffic noise (L_{night} and L_{den}) and reaction time test score

In the main model, there were no associations of road traffic noise, or rail noise (L_{night} and L_{den}), with reaction time scores (Table 2).

Higher levels of exposure to aircraft noise, L_{night} and L_{den}, were associated with a worse (i.e., longer) reaction time. Exposure to night-time aircraft noise (L_{night} aircraft) of ≥50–55 dB, as compared to <45 dB, was associated with 21.7 ms (95 % CI: 6.874–36.526) longer reaction time (Table 2) in the main model. Exposure to L_{den} aircraft ≥60 dB vs. <50 dB, was associated with 21.2 ms (95 % CI: 5.840–36.639) longer reaction time (Table 2).

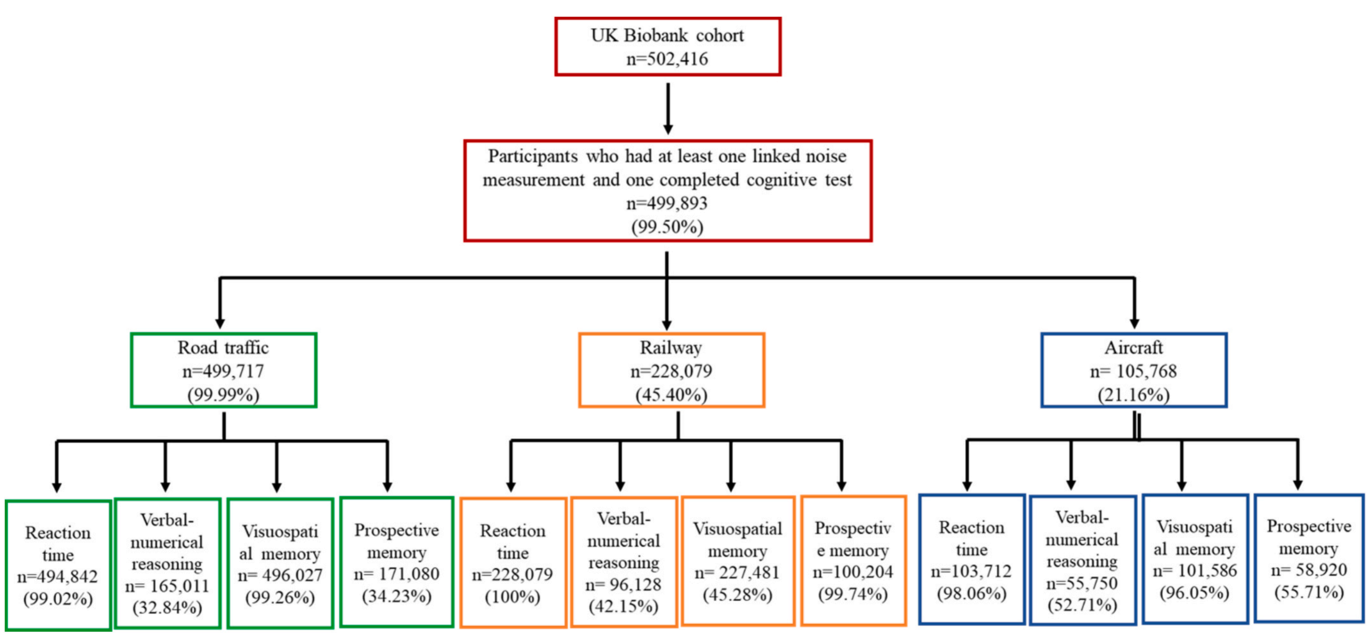

Fig. 1. The number of participants from the UK Biobank cohort who were included in the analyses, categorised into the type of transport noise they were exposed to and the cognitive test they completed at baseline assessment.

Table 1
Baseline characteristics of UK Biobank participants with a complete cognitive function test and at least one linked noise exposure data.

	Reaction time (n = 494,842)	Visuospatial memory (n = 496,027)	Verbal-numeric reasoning (n = 165,011)	Prospective memory (n = 171,080)
Age, mean (SD)	57.03 (8.10)	57.05 (8.10)	57.19 (8.15)	57.25 (8.16)
Sex, n (%)	225,460 (45.6)	226,106 (45.6)	75,067 (45.5)	77,967(45.6)
Ethnicity, n (%)				
White British	46,7697 (94.9 %)	467,401 (94.6 %)	152,132 (92.6 %)	155,691 (91.4 %)
Education level, n (%)				
Degree, yes	56,074 (11.5 %)	56,576 (11.5 %)	20,706 (12.7 %)	21,320 (12.6 %)
Townsend deprivation index, n (%)				
1 (Least deprived)	99,917 (20.2 %)	99,731 (20.1 %)	27,376 (16.6 %)	27,831 (16.3 %)
5 (Most deprived)	97,126 (19.7 %)	98,329 (19.8 %)	33,356 (20.2 %)	35,891 (21.0 %)
Smoking status, n (%)				
Never	269,657 (54.7)	270,598 (54.8)	90,533 (55.0)	94,136 (55.3)
Previous	171,214 (34.7)	171,183 (34.7)	57,517 (35.0)	58,972 (34.6)
Current	52,009 (10.6)	52,173 (10.6)	16,410 (10.0)	17,222 (10.1)
Alcohol consumption, n (%)				
Current	45,5172 (92.1)	45,5203 (91.9)	15,1532 (91.9)	15,5807 (91.2)
Employment status, n (%)	284,088 (96.9)	283,484 (96.8)	92,654 (95.8)	94,798 (95.6)
Average total household income before tax, n (%)				
<£31 k	202,082 (48.04)	204,551 (48.27)	67,440 (47.27)	70,327 (48.04)
≥£31 k	218,544 (52.96)	219,207 (51.73)	75,222 (52.73)	76,055 (51.96)
Sleep duration, n (%)				
<6 h	121,299 (24.7)	121,859 (24.7)	41,462 (25.3)	43,316 (25.5)
≥6–8 h	332,765 (67.7)	332,907 (67.6)	110,544 (67.3)	113,851 (67.0)
≥8 h	37,644 (7.7)	37,933 (7.7)	121,62 (7.4)	12,805 (7.5)
Cardiovascular risk score, n (%)				
Poor	177,869 (35.9)	176,317 (35.5)	56,857 (34.5)	59,453 (34.8)
Intermediate	282,487 (57.1)	284,943 (57.4)	95,782 (58.0)	98,935 (57.8)
Optimal	34,486 (7.0)	34,767 (7.0)	12,372 (7.5)	12,692 (7.4)
Stroke, n (%)	5,263 (1.1)	5,289 (1.1)	14,96 (0.9)	1,623 (0.9)
All-cause dementia, n (%)	7,408 (1.5)	7,498 (1.5)	1,936 (1.2)	2,172 (1.3)
Road Lden, dB				
Median, IQR	52.02 (49.69, 55.22)	52.03 (49.69, 55.24)	52.00 (49.72, 55.42)	52.01 (49.74, 55.45)
Noise category				
<50 dB	137,239 (27.7)	137,426 (27.7)	45,427 (27.5)	46,935 (27.4)
≥50–55 dB	229,035 (46.3)	229,353 (46.2)	75,584 (45.8)	78,328 (45.8)
≥55–60 dB	71,333 (14.4)	71,689 (14.5)	23,367 (14.2)	24,272 (14.2)
≥60 dB	57,235 (11.6)	57,559 (11.6)	20,633 (12.5)	21,545 (12.6)
Road Lnight, dB				
Median, IQR	45.32 (43.13, 48.45)	45.32 (43.13, 48.47)	45.30 (43.16, 48.64)	45.31 (43.17, 48.67)
Noise category				
<45 dB	137,239 (27.7)	137,426 (27.7)	45,427 (27.5)	46,935 (27.4)
≥45–50 dB	229,035 (46.3)	229,353 (46.2)	75,584 (45.8)	78,328 (45.8)
≥50–55 dB	71,333 (14.4)	71,689 (14.5)	23,367 (14.2)	24,272 (14.2)
≥55 dB	57,235 (11.6)	57,559 (11.6)	20,633 (12.5)	21,545 (12.6)
Rail Lden, dB				
Median, IQR	31.0(23.6, 38.1)	31.7(24.5, 38.6)	31.0(23.6, 38.2)	31.7(24.6, 38.6)
Noise category				
<50 dB	216,638 (94.99)	216,075 (94.99)	90,772 (94.43)	94,626 (94.44)
≥50–55 dB	5,705 (2.50)	5,695 (2.50)	2,674 (2.78)	2,776 (2.77)
≥55–60 dB	3,218 (1.41)	3,209 (1.41)	1,525 (1.59)	1,584 (1.58)
≥60 dB	2,499 (1.10)	2,483 (1.09)	1,152 (1.20)	1,214 (1.21)
Rail Lnight, dB				
Median, IQR	23.6[16.3, 30.8]	23.6[16.3, 30.8]	24.3[17.2, 31.2]	24.3[17.3, 31.2]
Noise category				
<45 dB	212,969 (96.39)	212,352 (96.39)	90,299 (96.03)	94,149 (96.02)
≥45–50 dB	4,218 (1.91)	4,209 (1.91)	1,988 (2.11)	2,065 (2.11)
≥50–55 dB	2,220 (1.00)	2,217 (1.01)	1,043 (1.11)	1,096 (1.12)
≥55 dB	1,544 (0.70)	1,529 (0.69)	704 (0.75)	737 (0.75)
Aircraft Lden, dB				
Median, IQR	54.09 (51.54, 57.73)	54.03 (51.50, 57.61)	54.51 (51.98, 58.47)	54.57 (52.00, 58.59)
Noise category				
<50 dB	75,430 (72.73)	74,117 (72.96)	34,543 (61.96)	36,441 (61.85)
≥50–55 dB	16,433 (15.84)	16,036 (15.79)	12,105 (21.71)	12,658 (21/48)
≥55–60 dB	7,847 (7.57)	7,560 (7.44)	5,801 (10.41)	6,086 (10.33)
≥60 dB	4,002 (3.86)	3,873 (3.81)	3,301 (5.92)	3,736 (6.34)
Aircraft Lnight, dB				
Median, IQR	49.01 (46.35, 52.53)	49.03 (46.37, 52.57)	49.62 (46.71, 52.84)	49.69 (46.76, 53.06)
Noise category				
<45 dB	91,745 (88.46)	90,243 (88.83)	46,988 (84.28)	49,465 (83.95)
≥45–50 dB	7,230 (6.97)	6,805 (6.70)	4,776 (8.57)	5,071 (8.61)
≥50–55 dB	3,409 (3.29)	3,211 (3.16)	2,917 (5.23)	3,103 (5.27)

(continued on next page)

Table 1 (*continued*)

	Reaction time (n = 494,842)	Visuospatial memory (n = 496,027)	Verbal-numeric reasoning (n = 165,011)	Prospective memory (n = 171,080)
≥55 dB	1,328 (1.28)	1,327 (1.31)	1,069 (1.92)	1,282 (2.18)
Particulate matter ($PM_{2.5}$) 2010 (median IQR), μg/m^3	9.93(9.29, 10.56)	9.93(9.29, 10.56)	9.88(9.30, 10.42)	9.89(9.31, 10.43)
$PM_{2.5}$ absorbance, 2010 (median IQR), (10^{-5} m^{-1})	1.13(1.00, 1.30)	1.13(1.00, 1.30)	1.17(1.00, 1.35)	1.17(1.00, 1.35)
Proximity to greenness, n (%)				
≥20 %	360,135 (72.8)	361,170 (72.8)	107,176 (65.0)	110,683 (64.7)

3.3. Associations between traffic noise (L_{night} and L_{den}) and visuospatial memory test score

For L_{night} road traffic noise, in the main model, there were positive but non-significant associations with visuospatial memory (Table 3) but no progression across increasing noise categories. For L_{den} road traffic noise, compared with L_{den} road of <50 dB, the error rate was higher by 7.3 % (PRR1.073, 95 %CI: 1.024–1.124), 7.7 % (PRR: 1.077, 95 %CI: 1.012–1.145), and 7.0 % (PRR:1.070, 95 %CI:0.999–1.146) among those exposed to ≥50–55 dB, ≥55–60 dB and ≥60 dB , respectively (Table 3)

For L_{night} rail traffic noise, in the main model, compared with L_{night} rail of <45 dB, exposure to L_{night} rail of ≥50–55 dB and ≥55 dB was linked with 39.9 % (PRR: 1.399, 95 %CI: 1.086–1.801) and 67.7 % (PRR: 1.677, 95 %CI: 1.288–2.183) higher error rate (Table 3). For L_{den} rail traffic noise, compared with L_{den} rail of <50 dB, exposure to L_{den} rail of ≥55–60 dB and ≥60 dB was linked with 30.3 % (PRR: 1.303, 95 %CI: 1.059–1.604) and 38.7 % (PRR: 1.387, 95 %CI:1.101–1.748) higher error rate (Table 3).

For L_{night} aircraft noise, in the main model, exposure to ≥55 dB vs. <45 dB was associated with a 133 % higher error rate in the visuospatial memory test (PRR: 2.334, 95 % CI: 1.657–3.288) (Table 3). For L_{den} aircraft, exposure to ≥60 dB vs. <50 dB, error rate was higher by 48 % (PRR: 1.480 (95 %CI: 1.175–1.863) (Table 3).

Subgroup and sensitivity analyses

We only performed subgroup and sensitivity analyses on L_{night} for rail and aircraft noise, which had highest and most convincing associations.

For L_{night} rail, comparing ≥55 dB with <45 dB, higher effect sizes were seen among females, those exposed to the lowest tertile of $PM_{2.5}$ (<9.9 μg/m^3), and with higher residential greenness exposure (Supplementary Table 5a).

For L_{night} aircraft, comparing ≥55 dB with <45 dB, higher effect sizes were seen among those less than 60 years of age, exposed to higher $PM_{2.5}$ exposure, exposed less to residential greenness, and had complete data for four cognitive tests (Supplementary Table 5b).

After excluding participants who had dementia or stroke, the associations with L_{night} rail remained similar while effect size for aircraft noise ≥55 dB vs. <45 dB became much higher (PRR: 3.175, (95 %CI: 2.253–4.474) (Supplementary Table 6a). We saw the same pattern in other sensitivity analyses (excluding CRS in the main model, adjusted for $PM_{2.5abs}$ rather than $PM_{2.5}$, and complete-case analyses) (Supplementary Table 6b and 6c). By converting cognitive function raw scores to z-scores and repeating the analyses, main findings still remain (Supplementary Table 6d and 6e). By including all participants in the analyses of rail (Supplementary Table 6f and 6g) and aircraft noise (Supplementary Table 6h and 6i), the strength of associations with visuospatial memory remained generally similar.

3.4. Associations of traffic noise (L_{night} and L_{den}) with prospective memory test performance, or verbal-numeric reasoning

None associations were found between traffic noise metrics and either prospective memory (Supplementary Table 7), or verbal-numeric reasoning test (Supplementary Table 8).

4. Discussion

In this large cross-sectional study, we examined the associations of traffic noise from road, rail and aircraft with cognitive function test performance in the UK Biobank participants. We found associations between higher exposure to rail or aircraft noise (both L_{den} and L_{night}) with higher error rate in the visuospatial memory test, which remained statistically significant after multiple testing correction. Exposure to higher L_{night} rail and L_{night} aircraft (≥55 dB vs <45 dB) was significantly linked with 68 % and 133 % higher error rate respectively in the visuospatial memory test. The respective figures for L_{den} rail and L_{den} aircraft (≥60 dB vs <50 dB) was 39 % and 48 %. Additionally for night-time aircraft noise, we also observed stronger associations among those who were exposed to higher $PM_{2.5}$ or less residential greenness. Overall, these novel findings have both clinical and public health implications, highlighting the importance of considering higher exposures to rail and aircraft noise as potential risk factors for cognitive impairment in middle-to-older aged adults.

Few previous studies that have reported on traffic noise, or community noise, and cognitive function vary in design including noise metrics studied and cognitive assessments performed (Mac Domhnaill et al., 2021; Ogurtsova et al., 2023; Tzivian et al., 2017, 2016a,b; Weuve et al., 2021; Wu et al., 2024; Yu et al., 2023). To our knowledge, there is only one previous study investigating the effects of rail and aircraft noise on cognitive function (Wu et al., 2024). The study investigated 2594 dementia-free participants aged ≥60 years old from Stockholm, Sweden, and showed that rail noise (per 10 dB L_{den} of 10-year weighted average noise preceding baseline recruitment) was associated with 0.041 points/SD reduction in the global cognitive function score at baseline. In their study, the global cognitive score covered domains of episodic memory, semantic memory, perceptual speed, and verbal fluency. The authors also reported that aircraft noise at the highest tertile level (>46.5 dB L_{den}) had a small but non-significant association with a lower global cognitive score (Wu et al., 2024). In another longitudinal study of older aged people (≥65 years old) residing in Chicago, it was reported that a 10 dB higher annual exposure to noise (mainly road noise) five years preceding cognitive assessment was linked with 0.09 standard deviation reduction in perceptual speed (Weuve et al., 2021).

Tzivian et al. (2016a) showed in a cross-sectional study of 4,086 middle-to-older aged adults (mean age 64.29 (SD ± 7.64)) from a highly urbanised population in Germany Ruhr area, that higher ambient noise (road noise, aircraft noise and industrial noise combined) exceeding 60 dB was linked with lower global cognitive function score. However, results from specific cognitive domains (verbal fluency test, immediate recall test, delayed recall test, labyrinth test and clock drawing test) did not show any association with traffic noise (Tzivian et al., 2016a). The clock-drawing test is the equivalent to the visuospatial memory test reported here in our study. Another study reporting the effect of transport noise on specific cognitive domain in adults showed that road traffic noise may be associated with lower executive function, as measured by the Animal Naming Test (Mac Domhnaill et al., 2021). However, this association appeared to be confounded by air pollution. There are differences between our study and (the small number of) previous studies, relating to the window of traffic noise exposure assessment, approaches to cognitive function assessment and reporting, and population

Table 2
Linear and categorical analyses of associations between night-time (L_{night}) and 24-hour (L_{den}) transport noise (road, rail and aircraft) with reaction time in UK Biobank.

			Unadjusted		Model 1[a]		Main model[b]	
Exposure	**Noise categ.**	**Cases (n)**	**β-coef. 95 % CI**	**pFDR**	**β-coef. 95 % CI**	**pFDR**	**β-coef. 95 % CI**	**pFDR**
Road noise L_{night}	per 10 dB	494,842	**1.50 (0.968,2.029)**	0.000	−0.66 (−2.319,0.989)	0.723	−0.30 (−2.059,1.463)	0.878
Noise category								
	<45 dB	227,795	Ref		Ref		Ref	
	≥45–50 dB	169,979	**0.994 (0.253–1.736)**	0.016	0.184 (−2.086–2.454)	0.952	0.361 (−2.042–2.764)	0.878
	≥50–55 dB	53,215	**1.753 (0.640–2.867)**	0.005	−1.185 (−4.566–2.196)	0.723	−0.989 (−4.565–2.587)	0.878
	≥55 dB	43,853	**2.288 (1.082–3.494)**	0.000	−1.734 (−5.506–2.038)	0.693	−0.963 (−5.017–3.091)	0.878
Rail noise L_{night}	per 10 dB	220,970	0.005 (−0.468–0.478)	0.984	0.635 (−0.810–2.079)	0.795	0.712 (−0.760,2.185)	0.872
Noise category								
	<45 dB	212,988	Ref		Ref		Ref	
	≥45–50 dB	4,218	0.534 (−3.185–4.253)	0.845	0.561 (−10.488–1.610)	0.989	0.658 (−10.575–11.891)	0.986
	≥50–55 dB	2,220	−4.162 (−9.265–0.941)	0.206	7.580 (−9.430–24.590)	0.795	8.118 (−9.152–25.387)	0.873
	≥55 dB	1,544	**−6.274 (−12.384; 0.165)**	0.092	1.765 (−17.646–1.176)	0.989	3.688 (−16.093–23.468)	0.949
Aircraft noise† L_{night}	<45 dB	91,745	Ref		Ref		Ref	
	≥45–50 dB	7,230	**8.556 (5.659–11.454)**	0.000	5.130 (−5.218–15.479)	0.795	5.238 (−5.347–15.825)	0.872
	≥50–55 dB	3,409	**9.439 (5.380–13.498)**	0.000	**22.748 (8.420–37.075)**	0.035	**21.700 (6.874–36.526)**	0.070
	≥55 dB	1,328	**27.152 (21.090–33.215)**	0.000	−4.335 (−32.630–23.959)	0.989	−5.256 (−33.648–23.136)	0.949
Road noise L_{den}	per 10 dB	494,842	1.484(0.964,2.004)	0.560	−0.625 (−2.245,0.994)	0.723	−0.269 (−1.994,1.456)	0.878
Noise category								
	<50 dB	137,239	Ref		Ref		Ref	
	≥50–55 dB	229,035	1.303 (0.513–2.092)	0.219	0.122 (−2.302–2.545)	0.952	−0.023 (−2.59–2.543)	0.986
	≥55–60 dB	71,333	1.231 (0.163–2.298)	0.350	−0.204 (−3.444–3.037)	0.952	0.383 (−3.035–3.800)	0.911
	≥60 dB	57,235	2.609 (1.459–3.760)	0.000	−2.043 (−5.615–1.529)	0.671	−1.715 (−5.543–2.112)	0.839
Railway noise L_{den}	per 10 dB	220,970	1.003(1.002,1.006)	0.004	1.001(0.992,1.010)	0.056	1.00(0.991,1.009)	0.966
Noise category								
	<50 dB	216,657	Ref		Ref		Ref	
	≥50–55 dB	5,705	0.830 (−2.375–4.035)	0.690	−1.207 (−10.654–8.241)	0.989	−1.384 (−10.981–8.214)	0.991
	≥55–60 dB	3,218	−3.973 (−8.217–0.270)	0.104	−1.964 (−15.518–11.591)	0.989	−1.483 (−15.261–12.294)	0.991
	≥60 dB	2,499	−4.191 (−8.999–0.616)	0.136	11.293 (−4.239–26.826)	0.608	12.893 (−2.952–28.737)	0.488
Aircraft noise† L_{den}	<50 dB	75,430	Ref		Ref		Ref	
	≥50–55 dB		7.309 (5.305–9.313)	0.000	−3.715 (−9.900–2.468)	0.751	−3.737 (−9.991–2.517)	0.757
	≥55–60 dB	7,847	5.497 (2.709–8.286)	0.000	−0.004 (−9.837–9.831)	0.999	−0.184 (−10.201–9.834)	0.991
	≥60 dB	4,002	**18.780 (15.158–22.402)**	0.000	**22.271 (7.469–37.074)**	0.044	**21.240 (5.840–36.639)**	0.088

a – adjusted for age, sex, individual-level SES (education, household income, current employment status) and area-level SES (Townsend), cardiovascular risk score (CRS), b – Model 1 + $PM_{2.5}$ and proximity to greenness † – aircraft noise exposure estimates were provided as noise contours (categorical data) with those exposed to less than 50 dB for L_{den} and 45 dB for L_{night} being labelled <50 dB and 45 dB, respectively.

characteristics. These differences, particularly on various reported cognitive function domains, make it difficult to directly compare our results with the previous literature.

For the first time, our findings revealed that exposure to higher aircraft noise or rail noise, was significantly associated with poorer performance of visuospatial memory test. Effect sizes of night-time noise (L_{night}) were larger as compared to 24-hour noise (L_{den}), especially for night-time aircraft noise. We suspect that noise-induced sleep disturbance could be a potential reason in explaining this. In a previous UK Biobank study of 22,102 participants, it was reported that L_{night} aircraft noise (≥55 dB vs. <45 dB) was significantly associated with higher levels of objectively-measured sleep disturbance such as lower relative amplitude, poorer inter-daily stability and greater intra-daily variability (Gong et al., 2024). Sleep disturbance has also been linked with impaired cognitive function in adults (Kong et al., 2023). As shown in the DAG (Supplementary Fig. 1), theoretically, sleep disturbance could serve as a mediator between the link of night-time traffic noise and cognitive function. However, a formal causal mediation analysis cannot be performed to confirm this due to the cross-sectional nature of our study. We also found that the effect sizes of aircraft noise were larger than those of rail noise. Aircraft noise is generally seen to cause more annoyance than road or rail noise if standardised at the same noise level (Lefèvre et al., 2020), therefore may trigger stronger stress response in the brain, which can partly explain the stronger negative impacts of aircraft noise on visuospatial memory. The association of road traffic noise, both L_{night} and L_{den}, with visuospatial memory appeared to be confounded by $PM_{2.5}$ in our study, which is in line with the Irish study (Mac Domhnaill et al., 2021).

Among the four cognitive domains, the associations between traffic noise exposures and visuospatial memory performance appeared to be more evident. This epidemiological link is supported by plausible biological mechanisms. As an environmental stressor, long-term exposure to traffic noise can cause neurohormonal response through activations of sympathetic nervous system, hypothalamic–pituitary-adrenal axis, and endocrine systems (Hahad et al., 2022). Such noise stress response ultimately leads to excess production of reactive oxygen species (ROS) and inflammation, which alter release of different neurotransmitters on the limbic surfaces of the brain as well as structural and functional changes in brain morphology, including hippocampus (Cui et al., 2015; Hahad et al., 2022; Hayes et al., 2019). The hippocampus has a key role in supporting normal function of short- and long-term memory, visuospatial memory, verbal memory and learning (Bird and Burgess, 2008). It is therefore biologically plausible that noise-induced changes in the hippocampus, if sustained, may have negative impacts on its normal cognitive functionality, leading to impairment in memory and learning. Interestingly, recent evidence suggests that impairment of the visuospatial memory is a potential cognitive marker of Alzheimer's disease as it affects p-tau levels and is linked with the temporal lobe atrophy (Seo et al., 2021). Traffic noise from road, rail and aircraft has been linked with dementia, particularly Alzheimer's disease (Cantuaria et al., 2021; Havyarimana et al., 2025). The other three cognitive domains involve other different regions of the brain, mainly in frontal lobe and parietal lobe (Burgess et al., 2001; Mole et al., 2021; Parker et al., 2013). The impact of long-term traffic noise exposure on these brain regions is still largely unknown, which warrants further studies investigating the association between traffic noise and brain structure phenotypes.

Our study has some notable strengths. The sample size is much larger than previous studies, with comprehensive information on relevant confounders and mediators being considered. Such a large sample size also enabled investigations of subgroup analyses, identifying vulnerable groups of noise effects on cognitive impairment. Road traffic noise estimates used in the present study have been improved by enhanced modelling, which had high spatial resolution covering noise exposure levels from both major and minor roads. Rail noise from major railway corridors and aircraft noise from airports near large urban areas of the UK have also for the first time been linked to a major UK cohort for epidemiological analyses.

A number of limitations should be considered when interpreting our findings. This is a cross-sectional study by design, meaning it is not possible to infer causality. The noise exposure metrics were modelled at later years (2011 for rail and aircraft noise and 2013 for road noise) after the baseline cognitive assessment (2006–2010). It is generally assumed that, in a setting such as the UK, there is little temporal change in road, rail and aircraft traffic noise exposure over a few years of time. For example, road traffic noise levels remained stable between 2003 and 2010 in a London study (Fecht et al., 2016). We have provided detailed information regarding the temporal changes of road traffic volumes in England, Scotland and Wales from 2006 to 2013, comparisons of aircraft noise between 2006 and 2011, and rail noise between 2011 and 2017 (Supplementary file). All three noise exposures show a high degree of temporal stability during these years, i.e., for almost all study areas there were rarely a change of 3 dB (a minimum change in sound level that becomes perceptible to the human ear) over time, which supports our assumption. We used highest spatial resolution data possible for each noise exposure. Aircraft noise exposure is at a slightly lower spatial resolution (at postcode) than road and rail (at home address or 10 m within address). UK postcodes are small geographical areas and comprise on average 12–15 households. Further and importantly, aircraft noise is generated above houses, so it is less prone to encounter surfaces in-between generation point and the house that may affect noise levels, unlike the situation for road and rail. Nevertheless, these modelled noise estimates at or near residential address were subject to exposure-misclassification, especially during the daytime when people likely spend majority of their day in different microenvironments such as workplaces. Indoor noise from outdoor is influenced by the façade insulation of buildings and windows-opening behaviour. Although a range of covariates had been considered in our modelling approach, it is possible that residual confounding may exist. We dichotomised three individual-level SES variables (education level, household income and paid employment), which capture different aspects of socioeconomic status and used each in the model. Dichotomisation may result in loss of information and potentially mask non-linear relationships between these variables and the outcome, which may result in some residual confounding. We have conducted multiple testing correction for all the studied associations; however, we still cannot completely rule out any chance finding.

With regards to the quality of the cognitive function test, they were generally brief and were subject to measurement error; with validation studies indicating low stability on some of the tests over time (e.g.: pairs matching test–retest in the follow up sample) (Lyall et al., 2016). Regardless, the pairs matching test for visuospatial memory was completed by nearly all the study participants and any large inter-subject variation is minimised by the large sample size of this study. Furthermore, it was not possible to tell the premorbid cognitive status that could be indicative of mild cognitive impairments and therefore may bias the results of this study towards the null. It is also worth noting that an individual's cognitive ability cannot be based on a few cognitive tests. Lastly, there exists a potential selection bias in the UK Biobank (Fry et al., 2017), where those who are more physically and cognitively healthy may be more likely or able to participate and travel to the assessment centre. Findings from UK Biobank may not be fully generalisable to the UK general population, given that most UK Biobank participants are of White ethnicity and have over-representation of those of more advantaged socioeconomic status.

Table 3
Linear and categorical analyses of associations between night-time (L_{night}) and 24-hour (L_{den}) transport noise (road, rail and aircraft) with visuospatial memory test performance in UK Biobank.

Exposure	Noise categ.	Cases (n)	Unadjusted PRR 95 % CI	Unadjusted pFDR	Model 1[a] PRR 95 % CI	Model 1[a] pFDR	Main model[b] PRR 95 % CI	Main model[b] pFDR
Road noise Lnight	per 10 dB	496,027	**1.02 (1.017–1.029)**	0.000	1.042 (1.010–1.070)	0.064	1.030 (0.997–1.061)	0.395
Noise category								
	<45 dB	228,045	**Ref**		Ref		Ref	
	≥45–50 dB	170,378	**1.031 (1.022–1.040)**	0.000	1.030 (0.988–1.073)	0.509	1.009 (0.966–1.054)	0.878
	≥50–55 dB	53,481	**1.033 (1.020–1.046)**	0.000	1.057 (0.996–1.123)	0.260	1.054 (0.989–1.123)	0.439
	≥55 dB	44,123	**1.050 (1.036–1.065)**	0.000	**1.086 (1.017–1.161)**	0.075	1.062 (0.989–1.140)	0.439
Rail noise Lnight	per 10 dB	220,326	0.992 (0.987–0.997)	0.008	0.998 (0.974–1.024)	0.989	1.003 (0.978–1.029)	
Noise category								
	<45 dB	212,371	Ref		Ref		Ref	
	≥45–50 dB	4,209	0.980 (0.940–1.021)	0.489	0.927 (0.760–1.130)	0.795	0.898 (0.732–1.102)	0.873
	≥50–55 dB	2,217	0.990 (0.935–1.047)	0.815	1.371 (1.065–1.765)	0.082	1.399 (1.086–1.801)	0.088
	≥55 dB	1,529	1.032 (0.965–1.103)	0.515	**1.623 (1.252–2.120)**	0.000	**1.677 (1.288–2.183)**	0.000
Aircraft noise† Lnight	<45 dB	90,243	Ref		Ref		Ref	
	≥45–50 dB	6,805	1.014 (0.982–1.048)	0.559	1.013 (0.841–1.218)	0.989	1.044 (0.868–1.257)	0.949
	≥50–55 dB	3,211	0.983 (0.939–1.029)	0.600	1.158 (0.910–1.473)	0.795	1.181 (0.923–1.511)	0.711
	≥55 dB	1,327	**1.365 (1.287–1.447)**	0.000	**2.300 (1.635–3.235)**	0.000	**2.334 (1.657–3.288)**	0.000
Road noise L_{den}	per 10 dB	496,027	**1.022 (1.017–1.029)**	0.000	**1.041 (1.011–1.070)**	0.068	1.030 (0.998–1.061)	0.370
Noise category								
	<50 dB	137,426	Ref		Ref		Ref	
	≥50– 55 dB	229,353	1.025 (1.015–1.034)	0.000	1.080 (1.033–1.129)	0.029	1.073 (1.024–1.124)	0.066
	≥55– 60 dB	71,689	1.039 (1.026–1.051)	0.000	1.085 (1.023–1.151)	0.068	1.077 (1.012–1.145)	0.152
	≥60 dB	57,559	**1.057 (1.043–1.071)**	0.000	**1.085 (1.018–1.157)**	0.082	1.070 (0.999–1.146)	0.352
Rail noise L_{den}	per 10 dB	220,326	0.997 (0.991–1.002)_	0.350	1.002 (0.978–1.026)	0.989	1.005 (0.981–1.030)	0.950
Noise category								
	<50 dB	216,094	Ref		Ref		Ref	
	≥50–55 dB	5,695	1.004 (0.969–1.041)	0.858	0.845 (0.706–1.012)	0.320	0.837 (0.697–1.005)	0.352
	≥55–60 dB	3,209	0.963 (0.918–1.010)	0.220	1.318 (1.076–1.615)	0.070	**1.303 (1.059–1.604)**	0.106
	≥60 dB	2,483	1.010 (0.957–1.065)	0.816	1.354 (1.075–1.707)	0.073	**1.387 (1.101–1.748)**	0.088
Aircraft noise†								
	<50 dB	74,117	Ref		Ref		Ref	
	≥50–55 dB	16,036	0.996 (0.975–1.019)	0.823	0.898 (0.799–1.007)	0.320	0.890 (0.792–0.999)	0.352

(*continued on next page*)

Table 3 (*continued*)

Exposure	Noise categ.	Cases (n)	Unadjusted PRR 95 % CI	Unadjusted pFDR	Model 1[a] PRR 95 % CI	Model 1[a] pFDR	Main model[b] PRR 95 % CI	Main model[b] pFDR
	≥55–60 dB	7,560	0.943 (0.914–0.974)	0.000	1.112 (0.939–1.316)	0.795	1.141 (0.964–1.351)	0.524
	≥60 dB	3,873	**1.156 (1.114–1.201)**	0.000	**1.424 (1.136–1.785)**	0.035	**1.480 (1.175–1.863)**	0.029

a – adjusted for age, sex, individual-level SES (education, household income, current employment status) and area-level SES (Townsend), cardiovascular risk score (CRS), b – Model 1 + PM2.5 and proximity to greenness † – aircraft noise exposure estimates were provided as noise contours (categorical data) with those exposed to less than 50 dB for L_{den} and 45 dB for Lnight being labelled <50 dB and 45 dB, respectively.; PRR-prevalence rate ratio.

5. Conclusion

We found significant associations between aircraft noise, and rail noise, with poorer performance on the visuospatial memory test, one of the major indicators in cognitive health. Future studies and longitudinal follow-up are needed to replicate and strengthen these findings.

CRediT authorship contribution statement

Enock Havyarimana: Writing – review & editing, Writing – original draft, Visualization, Supervision, Project administration, Methodology, Investigation, Formal analysis, Data curation. **Rabiya Gangrekar:** Writing – review & editing, Writing – original draft, Methodology, Formal analysis. **Xiangpu Gong:** Writing – review & editing, Project administration, Methodology, Data curation. **Calvin Jephcote:** Writing – review & editing, Methodology, Data curation. **Sarah Johnson:** Writing – review & editing, Methodology. **Sana Suri:** Writing – review & editing, Methodology, Investigation. **Wuxiang Xie:** Writing – review & editing, Investigation. **Charlotte Clark:** Writing – review & editing, Investigation. **Anna L Hansell:** Writing – review & editing, Resources, Project administration, Methodology, Investigation, Data curation. **Yutong Samuel Cai:** Writing – review & editing, Writing – original draft, Supervision, Project administration, Methodology, Investigation, Data curation.

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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.envint.2025.109941.

Data availability

Data will be made available on request.

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