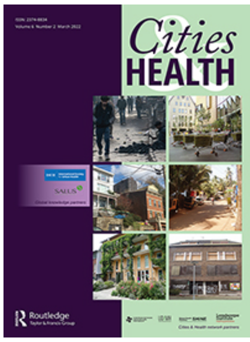

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Road traffic noise, quality of life, and mental distress among older adults: evidence from Ireland

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ABSTRACT
This research contributes to an emerging evidence base that considers a possible relationship between exposure to road traffic noise and mental distress. This study aimed to determine whether chronic exposure to road traffic noise was associated with quality of life or various measures of mental distress. We spatially linked high-quality modelled noise exposure data for the cities of Dublin and Cork in Ireland to The Irish Longitudinal Study on Ageing, allowing an examination of these health outcomes among older adults while adjusting for socio-demographic and behavioural characteristics. Exposure to air pollution was also considered in the analysis, allowing any associations between noise and either quality of life or mental distress that were independent of this other stressor to be isolated. While the study did not detect evidence of an association between noise exposure and depression, anxiety, stress, or worry, it identified a negative association between exposure to road traffic noise and quality of life that was independent of a range of socio-demographic and behavioural factors. Moving from the highest quintile of noise exposure to the lowest was associated with an increase on the CASP-12 quality of life scale of 1.08 of a standard deviation.

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Road traffic noise; Common Noise Assessment Methods in the European Union (CNOSSOS-EU); mental distress; quality of life; older adults; Ireland

Introduction

Environmental noise is defined by the World Health Organization (WHO) as 'noise emitted from all sources, except noise at the industrial workplace,' and includes 'road, rail, air traffic, industries, construction and public work, and the neighbourhood' (WHO 2011). It is an omnipresent feature of daily life, particularly in cities and urban areas. Urban transportation is a key source of environmental noise. For example, one study found high average levels of noise exposure on all transport modes around Paris (Kreuzberger *et al.* 2019), while unhealthy levels of noise pollution were recorded at bus stops in Chennai (Mahesh 2021). Unlike some other environmental stressors in cities, such as second-hand smoke, dioxins and benzenes, population exposure to environmental noise from road traffic is increasing in Europe (WHO 2011, European Environment Agency 2020, Murphy and King 2022). Increasing levels of mobility and road traffic linked with unbalanced urban development have been identified as particular factors in this increase (Silva and Mendes 2012). Overall, environmental noise has been associated with significant harmful effects on health (Faulkner and Murphy 2022) and it is now considered a key environmental and societal concern in cities, in both developed and developing countries (WHO 2018, Xu *et al.* 2020).

A recent literature review on links between transportation and health noted that noise had been identified as a pathway through which urban transportation can have a negative effect on health (Glazener *et al.* 2021). Noise is considered a psychological stimulus that induces a central nervous system response, which can in turn disrupt homeostasis, the optimal physical and chemical balance in the human organism (Recio *et al.* 2016). In terms of its pathology, the physiological reaction to noise is understood to include the hyperarousal of the sympathetic nervous and endocrine systems, and the secretion of stress hormones including cortisol and catecholamines through the hypothalamic-pituitary-adrenal and sympathetic-adrenal-medullary axes (Ader 2000). The biological defensive response is thus exacerbated and extended by exposure to noise (Aich and Potter 2009). Chronic exposure to environmental noise is hypothesised to impact negatively on quality of life, wellbeing and mental health through a prolonged activation of these physiological responses (Clark and Paunovic 2018). Night-time noise interfering with sleep has also been proposed as a mechanism through which noise could negatively affect quality of life or mental wellbeing (Clark and Paunovic 2018).

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Despite the high prevalence of environmental noise such as road traffic noise, previous research has tended to focus on the auditory effects of once-off extreme impulse noise or chronic exposure to noise in an occupational setting on human health (Basner *et al.* 2015). Two systematic reviews of environmental noise studies concluded that there is a relative lack of evidence linking noise exposure to quality of life, wellbeing and mental health (Clark and Paunovic 2018, Clark *et al.* 2020). The earlier review informed the WHO Environmental Noise Guidelines for the European Region (WHO 2018). While in general no effects were found, the quality of the evidence was considered low, so the authors could not definitively rule out associations between road traffic noise and these outcomes (Clark and Paunovic 2018). The more recent review found evidence of a harmful effect, but also deemed this evidence to be of low quality (Clark *et al.* 2020). Both reviews found similar evidence using antidepressant or anxiolytic medication as a proxy for depression or anxiety. In terms of quality of life, the Clark and Paunovic (2018) review noted several studies that utilised self-rated measures of quality of life and general health, but again with only low-quality evidence of no substantial effect.

Two groups highlighted as potentially vulnerable in relation to noise exposure and other health outcomes, such as cognitive performance or cardiovascular health, are children and older adults (Hygge 2003, van Kamp and Davies 2013). However, while several studies of noise and mental wellbeing or mental distress have focused on children, there appears to be a relative dearth of evidence that specifically considers older adults (Clark and Paunovic 2018, Clark *et al.* 2020, Douglas and Murphy 2020). One recent study empirically examined possible associations between road traffic noise exposure and cognitive health among older adults and found some evidence of an association between noise and executive function (Mac Domhnaill *et al.* 2021).

The current paper focuses on quality of life, mental distress and exposure to road traffic noise among older adults in Ireland. It contributes to an emerging evidence base on the impact of road traffic noise on mental health with a representative, cross-sectional study of older adults using regression models on microdata on health and wellbeing from The Irish Longitudinal Study on Ageing (TILDA). The study aimed to determine whether chronic exposure to road traffic noise was associated with quality of life and various measures of mental distress, including depression, anxiety, stressand worry.

Methods and data

The Irish Longitudinal Study on Ageing (TILDA)

TILDA is a nationally representative longitudinal study of over 8,000 persons aged 50 and over in Ireland (Kearney *et al.* 2011, Donoghue *et al.* 2018). At each wave of data collection (every two years), all members of sample households aged 50 years and over provide information on multiple aspects of their lives, including health, economic and social circumstances. To date, five waves of data collection have been completed. TILDA is part of an international network of longitudinal ageing studies, and therefore the survey content is harmonised with other studies, such as the Survey of Health, Ageing and Retirement in Europe, the English Longitudinal Study of Ageing and the Health and Retirement Survey in the US. At the first wave of data collection, the geo-code of each TILDA respondent's home address was recorded, allowing the dataset to be linked with other geo-coded spatial data. Three different methods are utilised to collect data for TILDA. In the first instance, Computer Assisted Personal Interviewing is used by trained interviewers who call to respondents' homes. Second, at the home interview, participants are given a self-completion questionnaire (to gather information on more sensitive issues, such as alcohol consumption, quality of relationships, etc.) which they then return by post to TILDA. Finally, at the first and third waves of data collection, respondents attended a health assessment at specialised Health Assessment Centres, or a modified partial assessment in their homes if travel to a centre was not practicable. These assessments were carried out by trained nurses. Written informed consent was obtained for all TILDA participants at each wave of data collection. Ethical approval for each wave of data collection was granted by the Faculty of Health Sciences Research Ethics Committee at Trinity College Dublin. The study was carried out according to the guidelines of the Declaration of Helsinki.

The current study employed data from the third wave of TILDA, which was collected between March 2014 and October 2015 from 6,396 individuals aged 54 and over. Data gathered from the Computer Assisted Personal Interviewing and self-completion questionnaire was used. Data on estimated noise exposure was matched to respondents living in the Dublin and Cork agglomerations, which are subject to noise mapping under the EU Environmental Noise Directive. Twelve respondents whose residences were exposed to railway (Environmental Protection Agency 2018a) or aircraft noise (Environmental Protection Agency 2018b) above 45 dB at night were removed from the analysis to isolate any associations between road traffic noise specifically and the outcome variables, giving a sample size of 1,706 for analysis. A flow chart detailing the sample size is provided in Appendix A. This sample size varied by outcome variable due to varying response rates in TILDA.

Noise exposure: modelling exposure to road traffic noise

Modelled levels of exposure to road traffic noise were spatially linked to TILDA using a Geographic Information System (GIS) platform, QGIS 3.10. This assigned a modelled level of noise exposure to each respondent at their residence. Noise exposure levels were estimated in decibels, dB(A), at the façades of residential buildings in Dublin and Cork, using input data collated between 2012 and 2016 including traffic flow and composition, building height and geometry, ground cover and topology as well as meteorological information. The Predictor-LimA Advanced V2019.02 software package was employed to model road traffic noise using the 'Common Noise Assessment Methods in the European Union' (CNOSSOS-EU) methodology, a common methodological framework recently developed for EU Member States (see Murphy *et al.* 2020). This study used L_{night}, the EU indicator of annual average (A-weighted) noise level for night-time periods, taken at the most exposed receiver point on the most exposed façade as the noise exposure variable. 'Night-time' is defined for L_{night} as an eight-hour period between 23:00 and 07:00. Figure 1 illustrates modelled values of L_{night} for Dublin and Cork based on this methodology, and descriptive statistics for the L_{night} variable are included in Table 1.

Previous literature was followed in spatially linking a single value of noise exposure estimated at the most exposed façade of the residence for each TILDA respondent (Tzivian *et al.* 2017). While the study had no data on the times at which TILDA respondents are usually in their residence, it is reasonable to assume that most respondents spend this overnight period at home, and that L_{night} is thus a good measure of a respondent's actual exposure to road traffic noise. In addition, sleep interference has been identified as a possible mechanism for a negative effect of noise exposure on quality of life or mental health, indicating the importance of studying L_{night} as an exposure variable (Clark and Paunovic 2018). Noise exposure was categorised using quintiles to protect anonymity (as a condition of linking noise exposure data with TILDA) and to allow for possible non-linear effects in any relationships between noise and the outcome variables. Noise exposure is measured on a logarithmic scale, and quintiles allow for the logarithmic form of the exposure variable by imposing no functional form restriction on the modelled relationship.

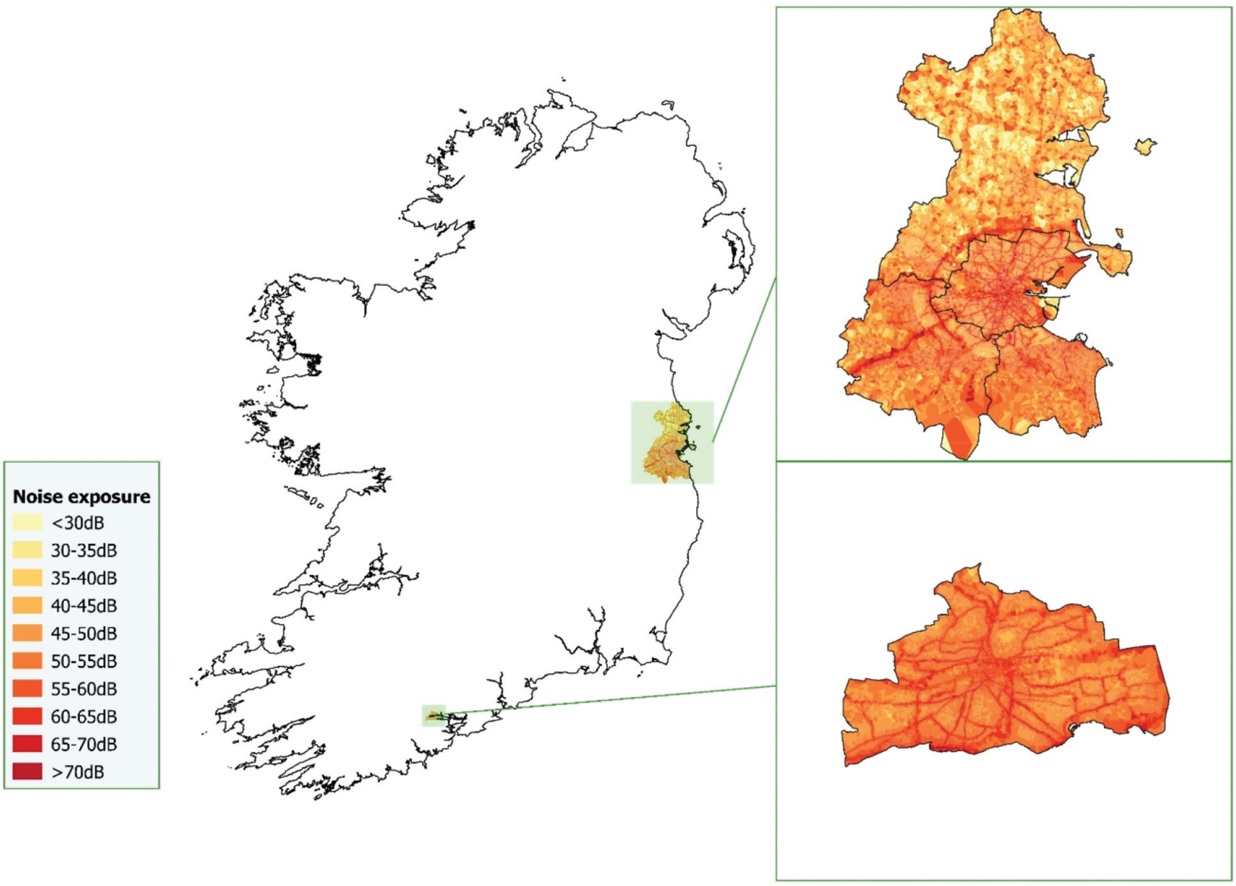

Figure 1. Night-time road traffic noise exposure (L_{night}) at the most exposed façade or residential buildings, Dublin and Cork.

Table 1. Descriptive statistics, pollution exposure variables.

Exposure variable	Units	n	Mean	SD	Min.	Max.
Night-time noise (L_{night})	dB (A)	1706	51.4	5.1	35.7	69.3
NO_2[a]	μg m^{-3}	568	15.6	3.9	8.0	38.6

SD denotes standard deviation.
[a]Source: Environmental Protection Agency, Ireland.

Outcome variables: quality of life and mental distress

A comprehensive picture of the mental health of respondents is established in TILDA by a rich set of questionnaire-based variables measuring quality of life and various forms of mental distress, including depression, anxiety, stress and worry. Descriptive statistics for the variables examined are presented in Table 2.

Quality of life was determined in wave 3 of TILDA across four domains using the twelve-item Control, Autonomy, Self-realisation and Pleasure (CASP-12) scale, with a higher score indicating a better quality of life (Hyde *et al.* 2003). To test for depression, TILDA employed the eight-item Center for Epidemiologic Studies Depression (CES-D) scale, which scored respondents on a scale from 0 to 24 with a score of 9 or above indicative of clinical depression (Radloff 1977). The anxiety subscale of the Hospital Depression and Anxiety scale (HADs-A) was used to test for anxiety, ranging from 0 to 21 with a score of 11 or above considered indicative of a case of anxiety (Zigmond and Snaith 1983). Stress and worry were tested using the Perceived Stress Score (PSS; Cohen *et al.* 1983), and the Penn State Worry Questionnaire (PSWQ; Meyer *et al.* 1990), respectively, with higher scores indicating greater levels of stress or worry.

In addition, details of any medication regularly taken by respondents were recorded, including antidepressant and anxiolytic medication, offering an alternative medication-based method of detecting the presence of a depression or anxiety disorder.

Covariates: socio-demographics and behaviour

Adjusting regression models for potentially confounding socio-demographic or behavioural characteristics suggested by the literature and collected in TILDA allowed the study to identify any associations between environmental noise and quality of life or mental distress that were independent of these individual-level characteristics. Table 3 presents descriptive statistics of these covariates.

Models were adjusted for age, gender, socioeconomic status, employment status, residential density within a 1-km radius, alcohol consumption, physical activity, general health status and social connectedness. Net household income and level of education were both used to measure socioeconomic status. Alcohol consumption was included using the outcome of the CAGE problematic alcohol scale (Ewing 1984), and physical activity was captured using the International Physical Activity Questionnaire (Craig *et al.* 2003). General health status was measured as having a long-term health limitation and by the use of five or more medications ('polypharmacy') on a regular basis. Finally, the Berkman-Syme Social Network Index (Berkman and Syme 1979) was used to measure social connectedness.

Table 2. Descriptive statistics and outcome variables.

Variable	Scale	n	Mean	SD	Min.	Max.
Quality of life	CASP-12[b]	1346	27.3	5.2	7.0	36.0
Depression	CES-D[b]	1675	3.5	3.9	0.0	24.0
Anxiety	HADs-A[a]	1679	3.7	3.5	0.0	19.0
Stress	PSS[b]	1426	4.0	3.1	0.0	16.0
Worry	PSWQ[b]	1356	16.0	7.9	8.0	40.0

n denotes number of observations. SD denotes standard deviation.
[a]Source: TILDA Computer Assisted Personal Interview.
[b]Source: TILDA self-completed questionnaire.
Scale ranges: CASP-12 0-36 (higher score indicates higher quality of life); CES-D 0-24 (9+ indicative of depression); HADs-A 0-21 (11+ indicative of anxiety case); PSS 0-16 (higher score indicates greater stress); PSWQ 8-40 (higher score inidcates greater worry).

Table 3. Descriptive statistics and potentially confounding characteristics.

Variable	Category	n	%
Age class	50-64	707	41.4
	65-74	594	34.8
	75-84	323	18.9
	85+	82	4.8
Female		943	55.3
Married		1129	66.2
Employment	Employed	485	28.4
	Retired	921	54.0
	Other	300	17.6
Higher education[a]	Primary or none	404	23.7
	Secondary	582	34.1
	Third level or above	720	42.2
Household income	€0 to €10 000	55	3.2
	€10 000 to €20 000	256	15.0
	€20 000 to €40 000	490	28.7
	€40 000 to €70 000	373	21.9
	€70 000 to €120 000	180	10.6
	€120 000 or above	34	2.0
	Missing data	318	18.6
Physical activity [b]	None	182	10.7
	Low	478	28.0
	Moderate	695	40.7
	High	351	20.6
Social connectedness [c]	Most isolated	141	8.3
	Moderately isolated	501	29.4
	Moderately integrated	681	39.9
	Most integrated	365	21.4
	Missing data	18	1.1
Health limitation		431	25.3
Alcohol problem		217	12.7
Polypharmacy [d]		452	26.5
Total observations		1706	100.0

Source: TILDA.
[a]In Ireland, primary education generally caters for students between 4 and 11 years, and secondary education between 12 and 18 years.
[b]Physical activity is categorised on the International Physical Activity Questionnaire scale (Craig *et al.* 2003).
[c]Social connectedness is categorised on the Berkman-Syme Social Network Index (Berkman and Syme 1979).
[d]In TILDA, polypharmacy indicates the regular use of at least 5 different medications.

Covariate: ambient air quality

Models were further adjusted for exposure to ambient air pollution in order to isolate any independent associations between environmental noise specifically and the outcome variables. Ground-level nitrogen dioxide (NO_2) exposure was modelled in micrograms per cubic metre air (μg m^{-3}) for 2015 by the Irish Environmental Protection Agency (Aves and Williams 2019), and this measure of NO_2 was employed in this study as a proxy for exposure to air pollution. Descriptive statistics for this variable are presented in Table 1. As with the noise exposure variable, exposure to air pollution was categorised using quintiles to protect anonymity and to allow for potential non-linear effects. Data on air quality was only available for a sub-sample of observations in central Dublin, and this investigation was thus limited to adjusting models for air pollution in a sub-sample analysis of 568 respondents. This sub-sample was not representative of the wider sample, however, as these observations reside in the more densely populated centre of Dublin, while the wider study area also encompasses less densely populated areas of both the Dublin and Cork agglomerations.

In this sub-sample analysis, alternative models that included each of $PM_{2.5}$ and PM_{10} as measures of air pollution instead of NO_2 were also run. These measures were also modelled for 2015 by the Environmental Protection Agency (Aves and Williams 2019).

Model specification

With this cross-sectional data, the study used an ordinary least squares linear regression model for the CASP-12 scale, and negative binomial regression models for the CES-D, HADs-A, PSS, and PSWQ scales, with standard errors clustered at the household level in each model. The choices of linear and negative binomial specifications were based on the forms of the respective outcome variables and empirically on a comparison of values for the Akaike Information Criterion and Bayesian Information Criterion for Gaussian, Poisson and negative binomial identities for each outcome variable, with Gaussian preferred for the CASP-12 scale and negative binomial for all other outcomes.

As an alternative measure to the CES-D and HADs-A scales for depression and anxiety, respectively, logistic regression models were employed to test whether noise exposure was independently associated with the use of antidepressant or anxiolytic medication.

As a sensitivity check of any results that indicated a possible association with noise exposure, a separate regression model was run that included L_{den}, an EU indicator of annual average (A-weighted) noise exposure over a 24-h period, as the noise exposure variable in place of L_{night}. As with L_{night}, this L_{den} indicator was generated as part of the study's noise modelling process.

For each outcome variable, these quantitative techniques tested the hypothesis that environmental noise was associated with the outcome. For each model, a backward selection process using F-tests was employed to remove covariates that did not impact the model. Statistical analysis was conducted using Stata 14.

Results are presented in the form of average marginal effects for quintiles of noise exposure relative to the reference category, the highest quintile of exposure. The average marginal effect associated with a quintile of noise exposure is the average of predicted changes in the outcome variable relative to the highest quintile of noise exposure, holding all other covariates constant. *P*-values, indicating the probability of obtaining results as extreme as the observed results under the null hypothesis, are also reported. Outcome variables were standardised using z-scores and reported average marginal effects can thus be interpreted as proportions of a standard deviation.

Results

Results of this study's main regression models, fully adjusted for socio-demographic and behavioural covariates, are summarised for the noise exposure variable in Table 4. A complete set of results for the CASP-12 model is provided as Model I of Table B1 in Appendix B.

Table 4 indicates a negative association between exposure to road traffic noise and quality of life, as measured by the CASP-12 scale. Moving from the highest quintile of noise exposure to the second lowest is associated with an increase on the CASP-12 scale of 1.07 of a standard deviation, and an increase of 1.08 of a standard deviation moving from the highest quintile of exposure to the lowest. By way of comparison, these average marginal effects are larger than the average marginal effect of moving from having completed only primary or no education to having completed secondary education. The *p*-values that correspond to these average marginal effects at lower quintiles of noise exposure are low at 0.01. This association is independent of the socio-demographic and behavioural characteristics considered in this investigation.

A separate linear regression model of the CASP-12 scale was run including L_{den} (24-h noise exposure) as the noise exposure variable to test the sensitivity of this finding to the choice of noise exposure indicator. Results for this regression, shown in Table B1 (Model II) in Appendix B, are broadly reflective of the main CASP-12 results.

However, the main results do not indicate any such association between noise exposure and depression, anxiety, stress or worry as measured by the respective questionnaire-based scales, the CES-D, HADs-A, PSS and PSWQ scales. The results also show no evidence of an

Table 4. Average marginal effects of noise exposure.

		dy/dx	(95% C.I.)	p-value
Quality of life (CASP-12)				
Noise	35.7-45.8 dB(A)	1.08	(0.27, 1.88)	0.01
	45.9-48.9 dB(A)	1.07	(0.22, 1.92)	0.01
	49.0-51.2 dB(A)	0.04	(−0.80, 0.88)	0.93
	51.3-53.7 dB(A)	0.67	(−0.16, 1.51)	0.11
	53.8-69.3 dB(A)	[ref.]		
N		1346		
Depression (CES-D)				
Noise	35.7-45.8 dB(A)	−0.06	(−0.62, 0.50)	0.84
	45.9-48.9 dB(A)	0.22	(−0.35, 0.78)	0.45
	49.0-51.2 dB(A)	0.53	(−0.07, 1.12)	0.08
	51.3-53.7 dB(A)	−0.03	(−0.56, 0.51)	0.92
	53.8-69.3 dB(A)	[ref.]		
N		1675		
Anxiety (HADs-A)				
Noise	35.7-45.8 dB(A)	0.10	(−0.41, 0.61)	0.70
	45.9-48.9 dB(A)	0.09	(−0.41, 0.60)	0.72
	49.0-51.2 dB(A)	0.15	(−0.38, 0.68)	0.58
	51.3-53.7 dB(A)	0.38	(−0.46, 0.53)	0.88
	53.8-69.3 dB(A)	[ref.]		
N		1679		
Stress (PSS)				
Noise	35.7-45.8 dB(A)	−0.10	(−0.64, 0.44)	0.71
	45.9-48.9 dB(A)	−0.08	(−0.61, 0.46)	0.78
	49.0-51.2 dB(A)	0.36	(−0.15, 0.88)	0.17
	51.3-53.7 dB(A)	0.22	(−0.28, 0.72)	0.39
	53.8-69.3 dB(A)	[ref.]		
N		1426		
Worry (PSWQ)				
Noise	35.7-45.8 dB(A)	−0.70	(−1.99, 0.60)	0.29
	45.9-48.9 dB(A)	−0.88	(−2.08, 0.33)	0.15
	49.0-51.2 dB(A)	−0.21	(−1.51, 1.09)	0.75
	51.3-53.7 dB(A)	−1.23	(−2.45, −0.01)	0.05
	53.8-69.3 dB(A)	[ref.]		
N		1356		

Noise exposure categorised using quintiles. Outcome variables standardised using z-scores.
Results correspond to models that adjust for socio-demographic, behavioral and health characteristics. *dy/dx* denotes average marginal effect. C.I. denotes confidence interval. N denotes sample size.

association with depression or anxiety when employing the use of antidepressant or anxiolytic medications as outcome variables.

When focusing on the sub-sample of 568 respondents who live in central Dublin and for whom data on ambient air quality was available to the study, however, there is no evidence of a relationship between noise exposure and respondents' score on the CASP-12 scale. Summary results of this sub-sample analysis are presented in Table B2 in Appendix B. There is also no evidence of a relationship between exposure to air pollution and the CASP-12 variable among this sub-sample.

Discussion

Employing a questionnaire-based measure of quality of life that scores respondents across four different domains, the CASP-12 scale, this study detects a negative association between exposure to road traffic noise and quality of life that is independent of a range of socio-demographic and behavioural factors. The association is also robust to the choice of L_{night} over L_{den} as the noise exposure variable. This indicates that respondents with lower exposure to road traffic noise achieve a higher quality of life score on the CASP-12 scale relative to the respondents most exposed to noise.

When focusing on a sub-sample in central Dublin to further adjust the model for exposure to ambient air pollution, no evidence of an association between noise and quality of life was found. Interpreting this result is difficult, as the sub-sample for which air pollution data was available was not representative of the wider sample, which is a limitation of the study. It may also be the case that the study did not have the statistical power to identify and disentangle any true associations with noise and air pollution in this sub-sample analysis. There is a moderate correlation of 0.69 between the study's measures of noise and NO_2 pollution, suggesting the possibility that multicollinearity may be an issue, which could mask a true association from detection. Alternatively, it may be an indication that the result for quality of life in the main model is spurious. Overall, however, due to the reduced sample size, results from the sub-sample analysis are not directly comparable with the main analysis. The 2018 review of previous research (Clark and Paunovic 2018) suggested that there is no substantial effect of noise on quality of life, although the measures of quality of life considered in these studies, such as the Short Form Health Survey or the General Health Questionnaire, were not as comprehensive as the CASP-12 scale in relation to quality of life.

No evidence was found of an association between noise exposure and depression or anxiety, measured using questionnaire-based scales, the CES-D and HADs-A, or measured through the use of antidepressant and anxiolytic medication. As discussed in the Introduction, the physiological reaction to noise exposure includes the secretion of stress hormones, and stress can thus be considered a potential mechanism for any effect on quality of life and mental distress. However, the study finds no evidence of an association between noise and either stress or worry, also measured utilising questionnaire-based scales, the PSS and PSWQ. Therefore, the null hypothesis of there being no relationship between road traffic noise exposure and either depression, anxiety, stress or worry cannot be rejected in this study. It is possible that testing for relationships between noise exposure at the most exposed façade of the respondent's residence and their quality of life or mental health may impede the clear detection of true effects of noise on mental distress. The earlier review of research in this area had suggested that there was evidence of road traffic noise having no effect on depression or anxiety (Clark and Paunovic 2018), although the more recent review noted some emerging longitudinal evidence of a harmful effect on various measures of depression and anxiety (Clark *et al.* 2020).

Conclusions

This paper contributes to an emerging evidence base that considers a possible relationship between exposure to road traffic noise and quality of life or mental distress. As environmental noise is an important feature of daily living in cities and urban areas, robust evidence on this issue is necessary in a global context of increasing urbanisation. While the study does not find any evidence of an association between road traffic noise exposure and depression, anxiety, stress or worry, it finds some evidence of a negative association between exposure to road traffic noise and quality of life as measured by the CASP-12 scale that is independent of a range of socio-demographic and behavioural factors.

This study's findings should be interpreted in the context of certain limitations. First, while this paper broadens the evidence base in relation to older adults, its findings cannot be generalised to the whole population. Subject to data availability, further research into different aspects of mental health that considers environmental noise as well as a wide range of socio-demographic and behavioural covariates for different population groups would be useful. Second, as with many studies in this literature, this investigation is cross-sectional in nature, and therefore causality cannot be inferred. Subject to data availability, future research could exploit longitudinal data to improve on this, although causality would still remain elusive given the possibility of confounding by unobservable time-varying factors.

Due to limitations in data availability, data on road traffic noise exposure was modelled based on input data collated between 2012 and 2016, while data on cognitive health, socio-demographic and behavioural variables were collected by TILDA during 2014 and 2015. It is considered unlikely, however, that the noise modelling input data changed significantly during this period, and the quantitative effect of this limitation is therefore likely to be small. Data limitations also dictated that models of quality of life or mental distress could only be adjusted for exposure to ambient air pollution for a smaller sub-sample in central Dublin. Applying this model of quality of life including both noise and air pollution to a larger sample could be useful.

Subject to the availability of a larger sample size, another potential avenue for further research is to explore whether the results of this study differ by other demographic characteristics, such as gender or socio-economic status among older adults. We have adjusted for these characteristics in this study such that results are independent of them, but it may be the case that there is a stronger association between noise exposure and quality of life among further sub-groups. Further research into possible mechanisms underlying a relationship between noise exposure and quality of life would also be beneficial.

This study makes several valuable contributions to the literature. First, high-quality modelled noise exposure data from two cities, based on the new CNOSSOS-EU standard, was spatially linked to TILDA. This allowed the study to examine quality of life and various forms of mental distress through well-established questionnaire-based measures, while adjusting for a wide range of socio-demographic and behavioural characteristics. In addition, the study focused on older adults, a group potentially vulnerable to noise exposure that has thus far received little attention in the literature. Furthermore, exposure to air pollution was considered in a sub-sample analysis, allowing the study to test for any associations between noise and the outcome variables that are independent of this other environmental stressor.

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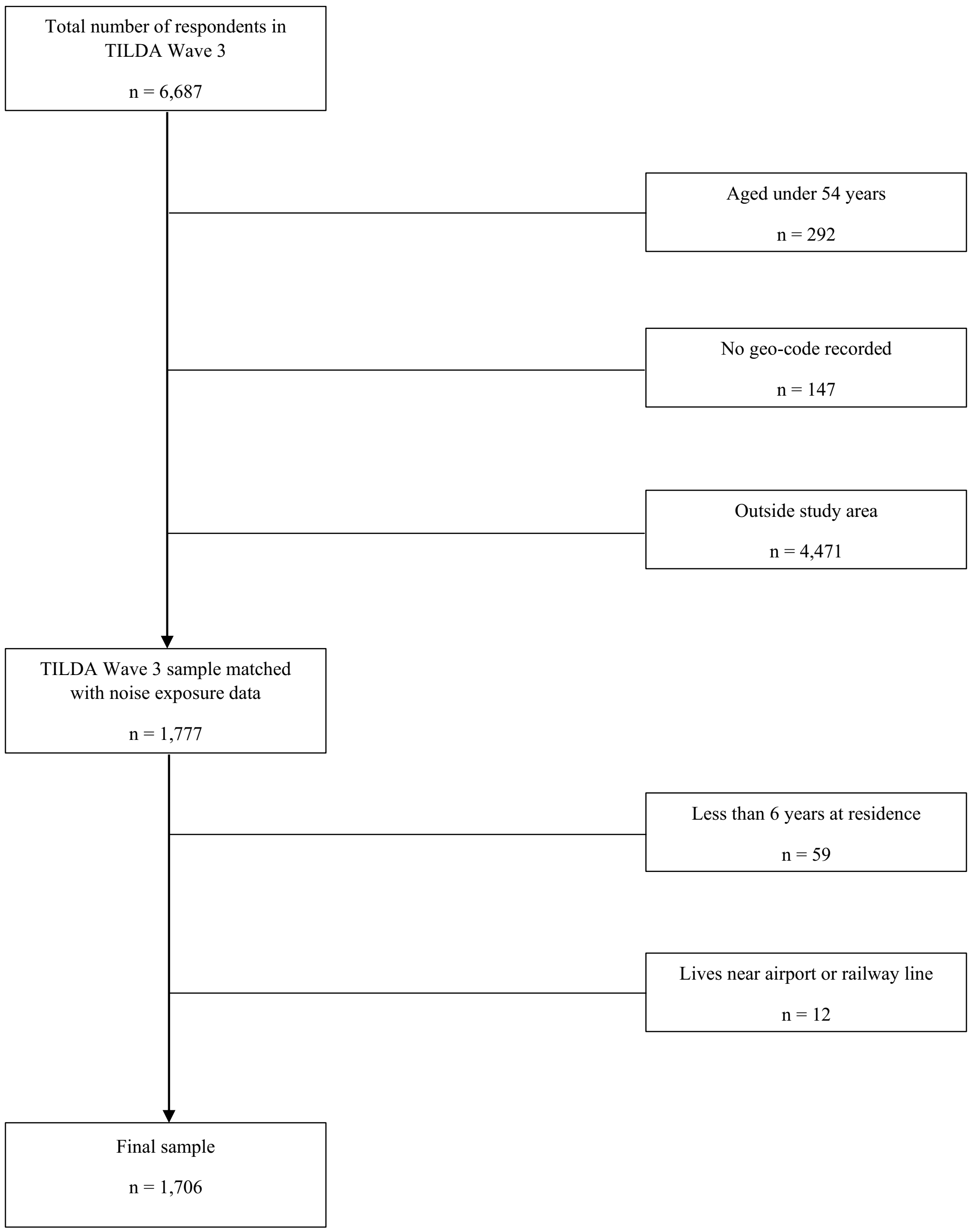
Appendix A. Sample size



Appendix B. Additional tables

Table B1: Average marginal effects, CASP-12 scale

		Model (I)			Model (II)		
Quality of life (CASP-12)		*dy/dx*	(95% C.I.)	p-value	*dy/dx*	(95% C.I.)	p-value
Noise (L_{night})	Lowest quintile	1.08	(0.27, 1.88)	0.01			
	Second quintile	1.07	(0.22, 1.92)	0.01			
	Third quintile	0.04	(-0.80, 0.88)	0.93			
	Fourth quintile	0.67	(-0.16, 1.51)	0.11			
	Highest quintile	[ref.]					
Noise (L_{den})	Lowest quintile				0.85	(0.04, 1.66)	0.04
	Second quintile				0.99	(0.13, 1.85)	0.02
	Third quintile				0.37	(-0.48, 1.23)	0.38
	Fourth quintile				0.97	(0.13, 1.81)	0.02
	Highest quintile				[ref.]		
Age		1.18	(0.72, 1.64)	0.00	1.18	(0.71, 1.64)	0.00
Age^2		-0.01	(-0.01, -0.01)	0.00	-0.01	(-0.01, -0.01)	0.00
Female		0.51	(0.02, 1.00)	0.04	0.61	(0.10, 1.11)	0.02
Employment	Employed	[ref.]			[ref.]		
	Retired	-0.70	(-1.47, 0.06)	-0.07	-0.65	(-1.42, 0.12)	0.10
	Other	-1.69	(-2.59, -0.79)	0.00	-1.52	(-2,43, -0.61)	0.01
Education	Primary/none	[ref.]			[ref.]		
	Secondary	0.93	(0.18, 1.68)	0.02	0.79	(0.04, 1.54)	0.04
	Tertiary/higher	1.35	(0.62, 2.08)	0.00	1.15	(0.42, 1.88)	0.00
Health limitation		-3.22	(-3.86, -2.58)	0.00	-3.21	(-3.85, -0.45)	0.00
Alcohol problem		-0.97	(-1.68, -0.25)	0.01	-1.08	(-1.79, -0.36)	0.00
Polypharmacy		-1.40	(-2.07, -0.73)	0.00	-1.39	(-2.06, -0.72)	0.00
Physical activity	None	[ref.]			[ref.]		
	Low	1.69	(0.58, 2.80)	0.00	1.66	(0.53, 2.78)	0.00
	Moderate	2.20	(1.09, 3.31)	0.00	2.09	(0.97, 3.22)	0.00
	High	2.82	(1.64, 4.00)	0.00	2.74	(1.54, 3.93)	0.00
Social network	Most isolated	[ref.]			[ref.]		
	Moderately isolated	1.38	(0.19, 2.57)	0.02	1.23	(0.00, 2.45)	0.05
	Moderately integrated	2.63	(1.46, 3.80)	0.00	2.42	(1.20, 3.63)	0.00
	Most integrated	2.08	(0.82, 3.33)	0.00	1.85	(0.54, 3.16)	0.01
N		1346			1346		

Model (I) includes L_{night} as the noise exposure variable. Model (II) includes L_{den} as the noise exposure variable.
Noise exposure categorised using quintiles. Outcome variable standardised using z-score. *dy/dx* denotes average marginal effect. C.I. denotes confidence interval. N denotes sample size.

Table B2: Sub-sample analysis average marginal effects, CASP-12 scale

		Model (I)			Model (II)			Model (III)		
Quality of life (CASP-12)		*dy/dx*	(95% C.I.)	p-value	*dy/dx*	(95% C.I.)	p-value	*dy/dx*	(95% C.I.)	p-value
Noise	Lowest quintile	1.17	(-0.40, 2.75)	0.14	0.75	(-0.67, 2.16)	0.30	0.73	(-0.75, 2.21)	0.33
	Second quintile	0.93	(-0.69, 2.55)	0.26	0.51	(-0.98, 2.00)	0.50	0.45	(-1.14, 2.03)	0.58
	Third quintile	0.77	(-1.36, 2.91)	0.48	0.53	(-1.42, 2.48)	0.59	0.56	(-1.43, 2.55)	0.58
	Fourth quintile	0.80	(-0.72, 2.33)	0.30	0.81	(-0.64, 2.27)	0.27	0.78	(-0.66, 2.23)	0.29
	Highest quintile	[ref.]			[ref.]			[ref.]		
NO_2	Lowest quintile	0.07	(-1.78, 1.92)	0.94						
	Second quintile	-0.14	(-1.98, 1.69)	0.88						
	Third quintile	0.08	(-1.67, 1.82)	0.93						
	Fourth quintile	-0.02	(-1.54, 1.51)	0.98						
	Highest quintile	[ref.]								
$PM_{2.5}$	Lowest quintile				0.77	(-1.03, 2.56)	0.40			
	Second quintile				1.01	(-0.63, 2.65)	0.23			
	Third quintile				0.90	(-0.73, 2.53)	0.28			
	Fourth quintile				0.17	(-1.37, 1.70)	0.83			
	Highest quintile				[ref.]					
PM_{10}	Lowest quintile							0.70	(-1.13, 2.53)	0.45
	Second quintile							0.69	(-1.02, 2.40)	0.43
	Third quintile							0.97	(-0.69, 2.63)	0.25
	Fourth quintile							-0.05	(-1.57, 1.47)	0.95
	Highest quintile							[ref.]		
N		568			568			568		

Models (I)-(III) all include L_{night} as the noise exposure variable. In Model (I), exposure to NO_2 pollution is included. In Model (II), exposure to $PM_{2.5}$ is included. In Model (III), exposure to PM_{10} is included.
Noise exposure categorised using quintiles. Outcome variable standardised using z-score. Results correspond to models that adjust for socio-demographic, behavioural and health characteristics. *dy/dx* denotes average marginal effect. C.I. denotes confidence interval. N denotes sample size.