



# Spatial and temporal associations of road traffic noise and air pollution in London: Implications for epidemiological studies



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## ABSTRACT

Road traffic gives rise to noise and air pollution exposures, both of which are associated with adverse health effects especially for cardiovascular disease, but mechanisms may differ. Understanding the variability in correlations between these pollutants is essential to understand better their separate and joint effects on human health. We explored associations between modelled noise and air pollutants using different spatial units and area characteristics in London in 2003–2010.

We modelled annual average exposures to road traffic noise ( $L_{Aeq,24\text{ h}}$ ,  $L_{den}$ ,  $L_{Aeq,16\text{ h}}$ ,  $L_{night}$ ) for ~190,000 postcode centroids in London using the UK Calculation of Road Traffic Noise (CRTN) method. We used a dispersion model (KCLurban) to model nitrogen dioxide, nitrogen oxide, ozone, total and the traffic-only component of particulate matter  $\leq 2.5\ \mu\text{m}$  and  $\leq 10\ \mu\text{m}$ . We analysed noise and air pollution correlations at the postcode level (~50 people), postcodes stratified by London Boroughs (~240,000 people), neighbourhoods (Lower layer Super Output Areas) (~1600 people), 1 km grid squares, air pollution tertiles, 50 m, 100 m and 200 m in distance from major roads and by deprivation tertiles.

Across all London postcodes, we observed overall moderate correlations between modelled noise and air pollution that were stable over time (Spearman's rho range: |0.34–0.55|). Correlations, however, varied considerably depending on the spatial unit: largest ranges were seen in neighbourhoods and 1 km grid squares (both Spearman's rho range: |0.01–0.87|) and was less for Boroughs (Spearman's rho range: |0.21–0.78|). There was little difference in correlations between exposure tertiles, distance from road or deprivation tertiles.

Associations between noise and air pollution at the relevant geographical unit of analysis need to be carefully considered in any epidemiological analysis, in particular in complex urban areas. Low correlations near roads, however, suggest that independent effects of road noise and traffic-related air pollution can be reliably determined within London.

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## 1. Introduction

Road traffic is a source of both noise and air pollution, particularly in urban areas. Air pollution is a long studied environmental exposure with well-known health effects, including effects on cardiovascular morbidity and mortality (WHO, 2013). There is a smaller evidence base for road traffic noise, but studies both at the individual level and

the community level have linked long-term exposure to annoyance (WHO, 2011), increased blood pressure (Babisch et al., 2012), cardiovascular disease (Vienneau et al., 2015) and mortality (Halonen et al., 2015b). Suggested mechanisms for effects of noise and air pollution differ — noise may result in release of stress hormones, activation of the autonomic nervous system and (at night) interference with sleep (Babisch, 2002), whilst suggested mechanisms for air pollution are through oxidative stress and inflammation (Kelly and Fussell, 2015). These different mechanisms may lead to differences and/or interactions in respective health impacts.

Studies have previously explored spatial associations between traffic-related noise and air pollution for specific measurement locations (Allen et al., 2009; Kheirbek et al., 2014; Shu et al., 2014; Weber and Litschke, 2008). To investigate population level health effects,

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however, epidemiological studies have to rely on residential exposure estimates from ambient exposure models because personal exposure or fixed-site measurements are not feasible (Beelen et al., 2009; Bilenko et al., 2015; De Roos et al., 2014; Gan et al., 2012).

75% of the population in Europe live in urban areas but few studies have assessed correlations of modelled data across large geographical areas such as cities (Gan et al., 2012). Thus, the extent to which the spatial unit chosen and spatial characteristics of the study area influence correlations is not clear. Correlations might vary substantially depending on the local geography and presence of major traffic sources. If noise and air pollution are, for example, highly correlated near roads where levels are highest, the choice of spatial unit of analysis will have important implications for results of epidemiological studies. Understanding the variability and differences in correlations between noise and air pollution levels over space and time is therefore essential to investigate potential for confounding or interactions, that will affect exposure-response estimations used to inform policy interventions.

We provide a detailed exploration of associations between modelled exposures to traffic-related noise and air pollution for residential postcodes in London; overall and within different spatial units and area characteristics including air pollution exposure bands, specific distance bands from heavily trafficked roads and deprivation bands.

## 2. Methods

### 2.1. Setting

We investigated the associations between annual average road traffic noise and air pollution levels between 2003 and 2010 for ~9 million residents in London.

Our study region was the area within the M25 ring motorway surrounding Greater London (see Fig. 1) and covered approximately 2000 km<sup>2</sup>. Traffic is the main source of noise and air pollution variability in London.

During the study period 2003–2010 major road traffic schemes were implemented by the Greater London Authority and Transport for London that aimed to reduce congestion and air pollution emissions and improve road safety across London. These included the introduction of the Congestion Charging Zone in February 2003 in central London with a Western Extension introduced in February 2007 (in operation until January 2011) (<http://www.tfl.gov.uk/modes/driving/congestion-charge/congestion-charge-zone>); the creation of the Low Emission Zone in 2008, which approximately follows the Greater London boundary (<https://www.tfl.gov.uk/modes/driving/low-emission-zone>); as well as the introduction of various 20 miles per hour speed limit zones across the city (Grundy et al., 2008). All these schemes had varying impacts on traffic speed, flow and composition (Transport for London, 2008a; Transport for London, 2008b), emissions of air pollution (Tonne et al., 2008) and potentially noise levels within London.

### 2.2. Unit of analysis

Postcodes were the highest level of resolution in this study. There are 190,122 residential postcodes within the study area each of which represents ~56 residents or ~22 households (Office for National Statistics (ONS), 2011). The postcode is a point location representing the geometric centroid of a postcode area (i.e. mid-point of all addresses associated with a postcode). We explored associations across all postcodes in London and within each London Borough, each neighbourhood, and within each 1 km × 1 km grid cell to analyse the effect of spatial

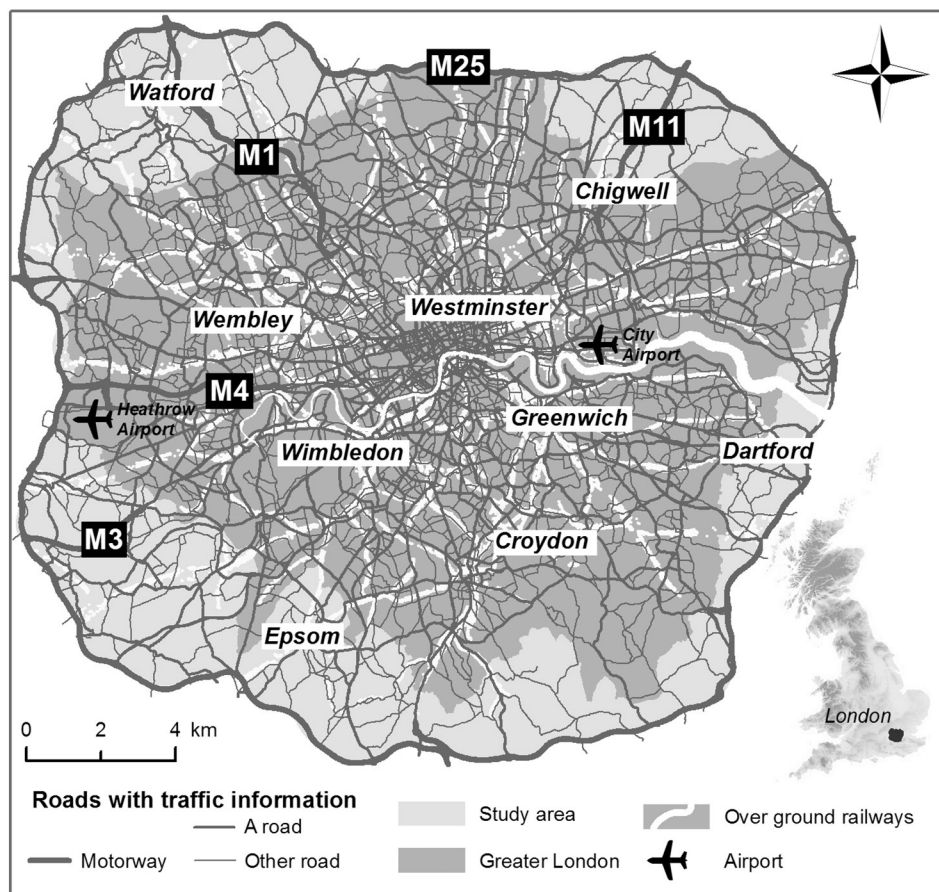


Fig. 1. London study area showing potential sources of traffic-related noise and air pollution.

aggregation and zoning on associations between noise and air pollution levels. London Boroughs are the “district level” geographical unit within Greater London ( $n = 32$ ) and have on average 240,000 residents (range 150,000–350,000), based on the census 2011 population (Office for National Statistics (ONS), 2011). We used Lower layer Super Output Areas (LSOAs) to represent neighbourhoods. LSOAs are a mid-level census dissemination unit which were constructed to represent homogeneous neighbourhoods in terms of key demographic and socio-economic characteristics. In London they have on average 1600 residents (range 1000–5000). We included all LSOAs whose area had at least a 90% overlap with our study area ( $n = 5359$ ). To explore the influence of zone design on associations we included all 1 km  $\times$  1 km grid cells that have their geometric centroid within our study area ( $n = 2,171$ ). The different units of analysis are illustrated in Supplemental Fig. S1.

### 2.3. Road-traffic noise

We modelled noise levels from road traffic sources for all residential postcodes in London following the UK Calculation of Road Traffic Noise (CRTN) methodology (Department of Transport, 1988). Noise estimates were made 1 m from the façade of the building associated with each postcode centroid. Our version of CRTN, TRANEX (Gulliver et al., 2015), created “ray paths” between each receptor (the postcode centroid) and each traffic source within 500 m. Traffic sources were represented by 10 m points along the main roads for which we had annual averaged daily traffic flow speed and characteristics available (see Fig. 1). Detailed information on the traffic variables can be found in Supplement Material ‘Description of road traffic data used to model noise levels in London’ and in Gulliver et al. (2015). Other model inputs were diurnal varying traffic flow, composition and speed from the London Atmospheric Emissions Inventory (LAEI) (London Atmospheric Emissions Inventory (LAEI), 2010), and road geography from the Integrated Transport Network (ITN), part of the Ordnance Survey’s 2009 version of Master Map™. For each receptor, we aggregated noise levels ( $L_{Aeq,1h}$ ) from all traffic sources within 500 m correcting for local traffic characteristics, shielding effects of buildings, ground cover attenuation and angle of view from the road. A TRANEX model evaluation exercise, conducted in the cities of Norwich and Leicester, showed very high agreement of  $R^2 = 0.80$  ( $p = 0.000$ ) between measured and modelled  $L_{Aeq,1h}$  across 73 sites (Gulliver et al., 2015).

We modelled annual average hourly sound levels in decibels,  $L_{Aeq,1h}$  where A is the A-weighting used to represent the relative loudness of sound as perceived by the human ear. We averaged  $L_{Aeq,1h}$  for the hours 00:00–23:00, 07:00–23:00 and 23:00–06:00 to produce noise metrics  $L_{Aeq,24h}$ ,  $L_{Aeq,16h}$  and  $L_{night}$  respectively which are commonly used in epidemiological studies (WHO, 2011).  $L_{Aeq,24h}$  corresponds to the time period of the air pollution estimates. We also calculated annual day-evening-night A-weighted equivalent continuous noise levels ( $L_{den}$ ), where an arbitrary weighting of 5 dB for noise in the evening (19:00–23:00) and 10 dB for noise at night (23:00–07:00) is added, a metric commonly used to predict noise annoyance and also used in some epidemiological studies (Beelen et al., 2009; de Kloizenaar et al., 2007; Gan et al., 2012; Selander et al., 2009; Sorensen et al., 2011).

### 2.4. Air pollution

We used high-resolution air pollution data provided by King’s College London as part of the TRAFFIC study to estimate air pollution exposure for each postcode in London. The air pollution model, KCLurban, uses a kernel modelling technique based on a dispersion model (ADMS) taking account of road transport, regulated industrial processes and other diffuse sources (Beevers et al., 2013). The model produced continuous surfaces (20 m  $\times$  20 m resolution) of annual mean concentrations for the period 2003–2010 for 12 pollutants (Beevers and Dajnak, 2015a). Model validation across fixed site monitors in London

(varying from a minimum of 62 sites in 2003 to a maximum of 100 sites in 2008) showed high agreement between measured and modelled values of Spearman’s rho between 0.7 and 0.8 depending on the pollutant and year (Beevers and Dajnak, 2015b). We aggregated air pollution estimates to postcode centroids by bi-linear interpolation of the 20 m  $\times$  20 m grid cells surrounding the postcode centroid. For the purpose of this study we focused on the gases nitrogen dioxide ( $NO_2$ ), nitrogen oxides ( $NO_x$ ) and ozone ( $O_3$ ) as well as on particles with diameters less than 2.5  $\mu m$  ( $PM_{2.5}$ ) and 10  $\mu m$  ( $PM_{10}$ ) and the respective local (i.e. generated within the study area) traffic component only ( $PM_{2.5traffic}$ ,  $PM_{10traffic}$ ), which included exhaust and non-exhaust PM from brake and tyre wear and re-suspension.  $NO_2$ ,  $NO_x$ ,  $PM_{2.5traffic}$  and  $PM_{10traffic}$  are local, primary traffic pollutants and biological relevant indicators of exposure to air pollution with known health effects.  $PM_{2.5}$  and  $PM_{10}$  are urban or regional background pollutants which often travel over wide distances while  $O_3$  is a regional, secondary pollutant.

### 2.5. Traffic indicators

We also studied associations of noise and air pollution correlations with traffic indicators, which have been applied in epidemiological studies in the past (Hoek et al., 2002). We derived road density by road class for each LSOA as follows:

$$D_{ri} = \frac{l_{ri}}{A_i},$$

where  $D_{ri}$  is the road density of road class  $r$  in LSOA  $i$ ,  $l_{ri}$  the length of road class  $r$  in LSOA  $i$  (m) and  $A_i$  the area of LSOA  $i$  ( $m^2$ ). We used the ITN to define the road geometry and, following visual inspection of the spatial distribution, reclassified road types into major roads (motorways and A-roads) and minor roads (B-roads, minor roads and all other roads). A-roads connect areas of regional importance, B-roads connect places of local significance and minor roads are local roads intended for local traffic (Ordnance Survey (OS), 2010). To account for traffic composition on roads we calculated vehicle kilometres for total and heavy vehicles only within each LSOA as follows:

$$VKT_{ti} = \sum_i V_i l_i,$$

where  $VKT_{ti}$  is the vehicle kilometres travelled for traffic type  $t$  in LSOA  $i$ ,  $V_i$  the number of vehicles of type  $t$  and  $l_i$  the length of the road segment. We defined traffic type ‘heavy vehicles’ to include coaches, light and heavy goods vehicles; counts come from the 2008 LAEI.

### 2.6. Deprivation

To analyse differences in associations between noise and air pollution by deprivation, we assigned each LSOA a Carstairs score, an area level composite measure of multiple deprivation. Carstairs scores were originally developed in 1991 (Carstairs and Morris, 1991), here it is based on 2011 Census variables. This was compiled for LSOAs in the study area and categorised into tertiles.

### 2.7. Statistical analysis

Noise and air pollution levels were modelled for all residential postcode centroids in London. In order to analyse potential changes over time in the associations between noise and air pollution we looked at yearly associations for the period 2003 to 2010 and the median across these years.

We used Spearman’s rho to assess correlations between noise and air pollution metrics, which is a non-parametric measure of dependence between two variables based on case rankings and suitable to assess correlations for variables with skewed distributions. We also calculated Pearson’s correlations of log transformed noise and air pollution levels



for comparison reasons. We excluded postcodes with either missing noise estimates (0.3%), as a result of the receptor placement issues in the noise model (for more information see Gulliver et al., 2015); or air pollution estimates (0.004%), locations outside the air pollution modelling domain.

We explored Spearman's rho correlations between noise and air pollution metrics (mean for 2003–2010) across postcodes within London, across postcodes within London Boroughs, across postcodes within LSOAs and across postcodes within 1 km × 1 km grid cells. We used Moran's I to assess whether there were geographical patterns in correlations across LSOAs and 1 km × 1 km grid cells. For selected air pollutants (PM<sub>2.5traffic</sub>, PM<sub>10</sub>, NO<sub>x</sub>, O<sub>3</sub>), we explored correlations with noise metrics for postcodes within different exposure categories (tertiles); different distances to major roads ( $\leq 50$  m,  $\leq 100$  m,  $\leq 200$  m from major roads with more than 10,000 vehicles per day); different deprivation bands (tertiles); and within Central London only (defined by the Inner London Boroughs, see Supplemental Fig. S1).

### 3. Results

#### 3.1. Postcode-level associations across all postcodes in London

Spatial patterns of noise ( $L_{Aeq,24\text{ h}}$ ) and air pollution (PM<sub>2.5traffic</sub>) exposure estimates for postcode locations in the centre of London are shown in Fig. 2, stratified by equal sized categories. In general,  $L_{Aeq,24\text{ h}}$  and PM<sub>2.5traffic</sub> exhibited different spatial distributions with higher noise levels strongly determined by proximity to main roads, while PM<sub>2.5traffic</sub> estimates, although higher near main roads, were generally more smoothly varying and higher in central London. We observed this spatial pattern for other traffic-related and background air pollutants.

We saw only small changes for each noise and air pollution metric over the study period (Supplemental Table S1). Noise level estimates changed very little between 2003 and 2010 (change in mean

$L_{Aeq,24\text{ h}} < 0.3$  dB). Particulates decreased over the study period (6.3  $\mu\text{g}/\text{m}^3$  and 5.4  $\mu\text{g}/\text{m}^3$  decrease in median concentration for PM<sub>10</sub> and PM<sub>2.5</sub>, respectively) while the traffic components as well as NO<sub>2</sub> and NO<sub>x</sub> fluctuated but decreased little. Differences are visualised in Supplemental Fig. S2. O<sub>3</sub> levels, which are not just due to local sources, increased by 5.5  $\mu\text{g}/\text{m}^3$ .

Table 1 shows the associations (Spearman's rho) between noise and air pollution exposure across all postcodes in London. Due to the small observed temporal variation, we report correlations between the median of metrics across all years. Correlations were very strong between the different noise metrics ( $r > 0.99$ ) and between the different air pollutants ( $r > |0.94|$ ); correlations with ozone were negative, as expected, due to it being generally elevated in background locations whereas other pollutants are mostly elevated close to source. We observed very similar patterns in correlations between the different noise and air pollutants, the magnitude of these correlations changes marginally between most traffic-related and background pollutants. PM<sub>10traffic</sub> and PM<sub>2.5traffic</sub> had the strongest correlations with any of the noise metrics ( $r = 0.51$ – $0.55$  and  $r = 0.49$ – $0.53$ , respectively); PM<sub>2.5</sub> the weakest ( $r = 0.39$ – $0.42$ ), apart from ozone. Pearson's correlations were marginally higher than Spearman's correlations (see Supplemental Table S2 and Supplemental Fig. S3).

#### 3.2. Postcode-level associations across London by exposure characteristics

As Supplemental Table S3 shows for selected air pollutants (PM<sub>2.5traffic</sub>, PM<sub>10</sub>, NO<sub>x</sub>, O<sub>3</sub>), these moderate correlations with noise metrics were mainly driven by postcodes in the highest third of air pollution estimates (Tertile 3) and lowest third (Tertile 1) for ozone; correlations in middle and lower exposure bands were very weak – for example correlations between  $L_{Aeq,24\text{ h}}$  and PM<sub>2.5traffic</sub> Tertile 1:  $r = 0.22$ ; Tertile 2:  $r = 0.14$ ; Tertile 3:  $r = 0.36$ . Including only postcodes close to major roads ( $< 50$  m) where both air pollution and noise levels are expected to be higher showed that correlations between noise and

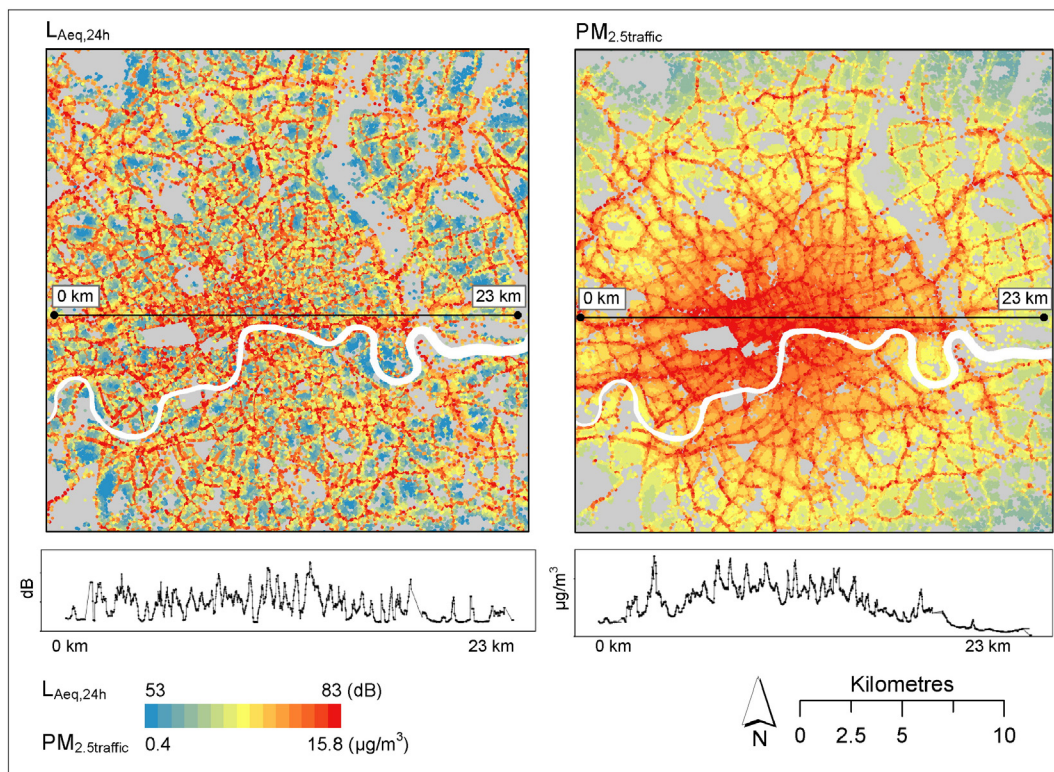


Fig. 2. Spatial distribution of noise ( $L_{Aeq,24\text{ h}}$ ) and air pollution (PM<sub>2.5traffic</sub>) levels in central London. Estimates are for postcode centroids, median across the study period 2003–2010. Graphs show  $L_{Aeq,24\text{ h}}$  (left) and PM<sub>2.5traffic</sub> (right) levels across a transect in the centre of London.

**Table 1**  
Spearman's correlations of all noise and air pollution metrics based on all postcodes in London (n = 189,583), median across years 2003–2010.

	Noise <sup>a</sup>				Air pollutants <sup>a</sup>						
	L <sub>Aeq,24 h</sub>	L <sub>den</sub>	L <sub>Aeq,16 h</sub>	L <sub>night</sub>	PM <sub>2.5traffic</sub>	PM <sub>2.5</sub>	PM <sub>10traffic</sub>	PM <sub>10</sub>	NO <sub>2</sub>	NO <sub>x</sub>	O <sub>3</sub>
L <sub>Aeq,24 h</sub>	–	1.00	1.00	0.999	0.506	0.399	0.524	0.452	0.402	0.417	–0.351
L <sub>den</sub>		–	0.999	0.999	0.513	0.407	0.531	0.460	0.409	0.425	–0.358
L <sub>Aeq,16 h</sub>			–	0.997	0.493	0.385	0.511	0.438	0.388	0.404	–0.336
L <sub>night</sub>				–	0.528	0.424	0.546	0.476	0.425	0.440	–0.375
PM <sub>2.5traffic</sub>					–	0.975	0.998	0.990	0.969	0.972	–0.947
PM <sub>2.5</sub>						–	0.969	0.995	0.984	0.981	–0.981
PM <sub>10traffic</sub>							–	0.987	0.967	0.971	–0.942
PM <sub>10</sub>								–	0.982	0.982	–0.971
NO <sub>2</sub>									–	0.998	–0.993
NO <sub>x</sub>										–	–0.989

<sup>a</sup> All correlations are statistically significant (p < 0.001).

selected air pollution metrics were of similar magnitude to those >200 m away from major roads – for example, correlations between L<sub>Aeq,24 h</sub> and PM<sub>2.5traffic</sub> ≤ 50 m: r = 0.31; >200 m: r = 0.27 (see Supplemental Table S4). In analysis stratified by Inner London and Outer London the magnitude of correlations was similar to that across all postcodes in London – for example, correlations between L<sub>Aeq,24 h</sub> and PM<sub>2.5traffic</sub> Inner London: r = 0.54; Outer London: r = 0.53 (see Supplemental Table S5).

3.3. Postcode-level associations by neighbourhood deprivation

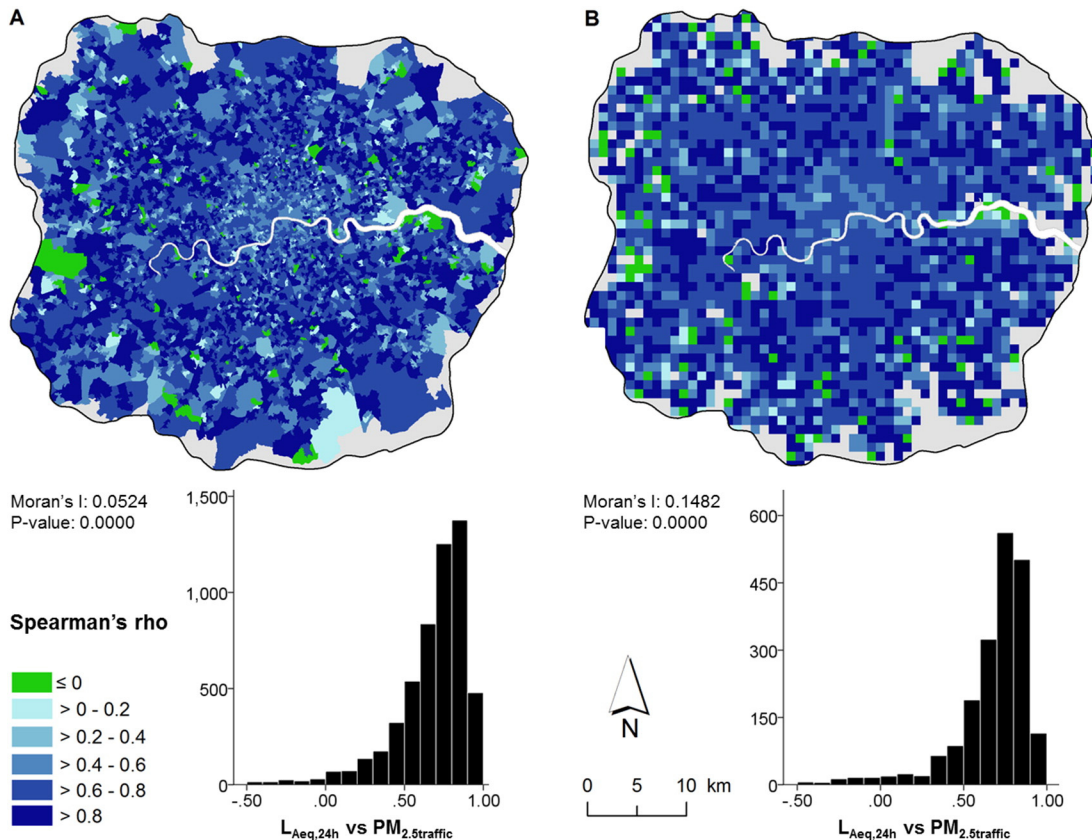
Deprivation status of LSOAs did not affect the correlations. If postcodes in London were stratified by deprivation tertiles, correlations between noise and air pollution metrics were of similar magnitude than across all of London. As Supplemental Table S6 shows, correlations between noise and air pollution metrics were marginally higher in the

2nd Tertile and lowest in the 1st Tertile – for example, correlations between L<sub>Aeq,24 h</sub> and PM<sub>2.5traffic</sub> Tertile 1: r = 0.49; Tertile 2: r = 0.57, Tertile 3: r = 0.55.

3.4. Postcode-level associations by different spatial units

Postcode level associations between noise and air pollutants were stronger within London Boroughs than across all of London, with strongest associations between L<sub>night</sub> and PM<sub>10traffic</sub> which varied between r = 0.55 in Bromley and r = 0.77 in Haringey. Correlations between L<sub>Aeq,16 h</sub> and NO<sub>2</sub> showed the largest variation between r = 0.31 in Hillingdon and r = 0.71 in Hammersmith and Fulham (see Supplemental Fig. S4 for a map of correlations by London Boroughs).

Correlations between noise and air pollution estimates across postcodes within LOSAs and 1 km × 1 km grid cells varied even more than across London Boroughs and were mostly strong to very strong (see



**Fig. 3.** Spatial distribution of correlations of postcode level exposure estimates for L<sub>Aeq,24 h</sub> and PM<sub>2.5traffic</sub> for (A) LSOAs (n = 5,359) and (B) 1 km × 1 km grid cells (n = 2,171). Histograms show distribution of correlations values between L<sub>Aeq,24 h</sub> and PM<sub>10traffic</sub> (median across 2003–2010).

Fig. 3); 74% of LSOAs, for example, had a correlation between  $L_{Aeq,24\text{ h}}$  and  $PM_{2.5\text{ traffic}}$  of  $r \geq 0.6$ , 35% of  $r > 0.8$ ; 2% of LSOAs, however, had negative correlations. These correlation patterns are randomly distributed across LSOAs and  $1\text{ km} \times 1\text{ km}$  grid cells in London and not clustered as indicated by a statistical significant ( $p < 0.0001$ ) Moran's I close to zero (see Fig. 3).

Very strong correlations between postcode level estimates within LSOAs cannot be explained by the presence or absence of major roads because we did not find any associations between correlations and either road density or vehicle kilometres ( $r = |0.00\text{--}0.17|$ ). We also did not find any association between the direction and magnitude of postcode level correlations between noise and air pollution metrics within LSOAs and the socio-economic status of the LSOA ( $r = |0.01\text{--}0.15|$ ).

## 4. Discussion

### 4.1. Main findings

We studied the spatial and temporal associations between modelled traffic-related noise and air pollution levels. Both noise and air pollution showed different spatial patterns, and correlations across postcodes varied substantially depending on the unit of analysis: London, London Boroughs, neighbourhoods and  $1\text{ km} \times 1\text{ km}$  grid cells. Correlations across all postcodes in London were moderate and varied depending on the exposure metrics between Spearman's rho of 0.34 and 0.55. Correlations were mostly stable across exposure tertiles, distance to major road and deprivation tertiles.

We saw little temporal changes between 2003 and 2010 in noise levels and air pollution concentrations even though some major traffic policy schemes had been implemented across this period such as the Low Emission Zone in 2008 and Congestion Charging Zones in 2003 and 2007. Little temporal change across London in traffic volume, one of the most important parameters together with distance in our noise model, means that noise levels change little over time. The logarithmic scale of the noise values further suppresses any temporal variability. Air pollution levels also vary by relatively small amounts over time, even those which relate specifically to local air pollution ( $NO_x$ ) and traffic components of PM ( $PM_{2.5\text{ traffic}}$ ,  $PM_{10\text{ traffic}}$ ). For PM metrics, reductions in tail-pipe emissions have been offset by increases in the number and weight of vehicles which has resulted in a proportional increase in particulate matter from brake and tyre wear and road abrasion (Boulter, 2005). Despite little temporal change in either noise or air pollution levels potential qualitative changes in particle composition and noise characteristics due to changes in the fleet might lead to changes in associations with health.

The common and dominant source of air pollution concentrations for the various metrics in London is road traffic, hence the high level of correlation between most of the different pollutant metrics. This is sometimes but not always the case elsewhere and especially where there are a number of very different sources which contribute to air pollution. The other main method for air pollution exposure modelling is land use regression (LUR), where high correlation ( $r > 0.8$ ) has also been shown between different pollutants (e.g.  $PM_{2.5}$  and  $NO_2$ ), but also much lower correlations in others ( $r < 0.5$ ) (Cesaroni et al., 2014). Unlike dispersion modelling, used here, where the same fundamental dispersion processes are used for many of the pollutant metrics, correlation between pollutants tends to decrease in LUR where different variables and zones of influence (e.g. size of circular buffers) are selected.

There were only marginal differences in correlations between different noise and air pollution metrics. Correlations were strongest between noise metric and the traffic component of particulates ( $PM_{2.5\text{ traffic}}$  and  $PM_{10\text{ traffic}}$ ) because modelled noise in our analysis comes entirely from road traffic sources. We observed weak negative correlations, as expected, between ozone and noise metrics ( $r = -0.34$  to  $-0.38$ ). A reason for these weaker correlations might be that noise is geographically smoother in background areas where

we did not have traffic information for minor roads, and consequently no variability in locations where ozone variability is expected to be high. This is confirmed by the correlations between ozone and noise metrics stratified by exposure bands where negative correlations are strongest in the lowest ozone exposure tertile (i.e. close to main roads), and weakest in the highest exposure tertile (i.e. in background locations).

Some noise metrics ( $L_{Aeq,16\text{ h}}$ ,  $L_{\text{night}}$ ) are not representing the same overall time periods as air pollutants which are summarised as annual average daily values. Although the time periods are not the same as for air pollutants we included  $L_{Aeq,16\text{ h}}$  and  $L_{\text{night}}$  in our analysis as they are commonly used in epidemiological studies.

Correlations between noise and air pollution metrics do not seem to be driven by higher exposures. We only saw marginally higher correlations between noise and air pollution in the highest air pollution tertiles. Distance from road had also little effect on correlations. Stratified analysis by Inner London and Outer London indicated that the observed patterns in correlation across London overall are not a central London effect where both noise and air pollution are higher. Slightly stronger correlations tend to be seen where there is more variability in exposures, hence the stronger correlations seen in the highest exposure band and more noticeably across all postcodes in London.

### 4.2. Differences in correlation by spatial units

The range of correlations between noise and air pollution metric increased by decreasing size of spatial unit from London Boroughs to  $1\text{ km} \times 1\text{ km}$  grid cells and neighbourhoods. The values of Moran's I confirmed that there were no spatial patterns in correlation values associated with neighbourhoods. Furthermore, we could not detect any associations between noise or air pollutants and traffic indicators, such as road density and vehicle kilometres that might explain the large differences in postcode level correlations between neighbourhoods. This is in line with previous research looking at associations between community noise and traffic-related air pollution. Gan et al. (2012), for example, did not detect stronger correlations between noise and traffic related air pollutants ( $PM_{2.5}$  and  $NO_2$ ) in areas close to major roads. Allen et al. (2009) did not find an effect of proximity to roads on correlations between noise and air pollution, comparable to our findings that correlations across postcodes close to road are of similar magnitude than across all of London.

The choice of geographical unit affects both the average and range of correlations in the comparison of all noise and air pollution metrics. The degree of correlation is a function of zone design which is commonly referred to as the Modifiable Areal Unit Problem (MAUP) (Openshaw, 1984). We observed the same patterns of MAUP as previously described in other studies (Cockings and Martin, 2005) in that the average and range of correlations increase when the size of spatial units decrease. In decreasing order of size, we observed for Greater London, London Boroughs,  $1\text{ km} \times 1\text{ km}$  grid cells and LSOAs (average size of  $0.5\text{ km}^2$ ) median correlations (min, max) between  $L_{Aeq,24\text{ h}}$  and  $PM_{2.5\text{ traffic}}$  of  $r = 0.51$  (0.51, 0.51), 0.67 (0.53, 0.90), |0.69 (0.04–0.98)|, |0.69 (0.01–0.98)|.

### 4.3. Strength and limitations

The noise model (TRANEX) does account for shielding by buildings and diffraction of noise around buildings, but not reflections (Gulliver et al., 2015). The CRTN method on which TRANEX is based includes reflection terms for buildings adjacent to major roads but not elsewhere. The maximum increment in noise levels in CRTN due to reflections is 1.5 dB(A) which applies in full street canyons. In partial street canyons or where buildings along major roads are more dispersed the increment due to reflections will be much lower, in many cases  $< 1\text{ dB(A)}$ . Although we do not account for reflections we, therefore, expect any



misclassification to be small, especially if used in an epidemiological context where relative ranking is important, rather than absolute levels of noise.

Noise and air pollution gradients differently vary with distance from sources due to the differential effects of buildings, land cover and meteorology on their dispersion/propagation. Buildings in particular are an important determinant of noise propagation but they may also influence the trapping of air pollutants (e.g. street canyons). Meteorology is an important determinant of air pollution dispersion and also influences the propagation of noise. Detailed information on building footprints and heights were included in the noise model, type of street canyon with a proxy for building height in the air pollution model. Conversely meteorology (i.e. wind direction) was included in the air pollution modelling but is not included in our noise model (but is included in other noise models such as NMPB 2008 (Setra, 2009) or Nord2000 (Kragh et al., 2006)). Thus, the associations presented here between noise and air pollution metrics may generally be weaker than in reality, to an unknown degree, due to these differences in model parameterisation.

Despite the differences in model parameterisation and distance-decay functions, however, measurement studies have also shown that the correlation between noise and air pollutants are moderate overall, which supports the findings of our study. Kheirbek et al. (2014) found moderate correlations for NO<sub>2</sub> (Pearson's  $r = 0.6–0.7$ ) and PM<sub>2.5</sub> ( $r = 0.45–0.51$ ) and selected noise metrics across 56 sampling sites in New York City. Foraster et al. (2011) also found moderate correlations (Pearson's  $r = 0.62$ ) between measured NO<sub>2</sub> concentrations and modelled noise levels (A-weighted long-term average sound level for 24 h) across 83 sites in the Spanish city of Gerona. These are of similar magnitude to correlations found between NO<sub>2</sub> and 5-min A-weighted equivalent continuous sound pressure levels (L<sub>eq,5 min</sub>) measurements in Vancouver (Davies et al., 2009). A study conducted in two US cities (Chicago and Riverside) observed slightly lower correlations between 151 measurements of L<sub>eq,5 min</sub> and NO<sub>2</sub> ( $r = 0.38$  and  $0.46$ , respectively) (Allen et al., 2009).

We did not have any co-located noise and air pollutant measurements; therefore, it was not possible to assess the level of correlation in model errors between various combinations of noise and air pollution metrics, and thus assess the implications for epidemiologic studies in terms of correlated measurement error (Dionisio et al., 2014; Zeger et al., 2000).

#### 4.4. Implications for epidemiological studies

Based on our work it is likely that some of the differences between previous studies investigating correlations may relate to differences in size and shape of spatial units at which assessments were made, and, therefore, the spatial units used in population health studies where exposures and/or health data are at aggregated level should be carefully considered. As discussed previously in Section 4.2, this is a recognised geographical issue (MAUP) (Openshaw, 1984), but it has seldom been explicitly considered in epidemiological studies. It has been suggested in previous work that the spatial unit of analysis should reflect the expected geographical scale of interaction between the pollutant and the health outcome (Parenteau and Sawada, 2011). We recommend investigating the impact of using different spatial scales to help to test the stability of results. We also suggest that epidemiological studies should consider incorporating a measure of the distribution within the analyses. It is reassuring that even for postcodes closest to roads or in the highest air pollution tertiles within London, where road traffic noise and air pollution might be expected to be very highly correlated, postcode level correlations were moderate. This suggests that studies estimating independent health effects of noise and air pollution in London where both pollutants are considered in the same model (Halonen et al., 2015a; Halonen et al., 2015b) are less likely to be affected by

colinearity, although the extent to which this may occur even with moderate correlations is debated (Foraster et al., 2014).

We conclude that exploring the differences in spatial correlation between traffic-related noise and air pollution exposure is important to evaluate the potential joint effects of noise and air pollution and this cannot be readily predicted in advance. Careful consideration of the spatial unit of analysis is important and inclusion of within unit distribution of correlations within statistical models should be considered where this information is available.

#### Conflict of interest

The authors declare no conflict of interest.

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#### Appendix A. Supplementary data

Supplementary data to this article can be found online at <http://dx.doi.org/10.1016/j.envint.2015.12.001>.

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