Commentary

Local determinants of road traffic noise levels versus determinants of air pollution levels in a Mediterranean city

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ABSTRACT

Background: Both traffic-related noise and air pollution have been associated with cardiovascular disease (CVD). Spatial correlations between these environmental stressors may entail mutual confounding in epidemiological studies investigating their long-term effects. Few studies have investigated their correlation – none in Spain – and results differ among cities.

Objectives: We assessed the contribution of urban land-use and traffic variables to the noise–air pollution correlation in Girona town, where an investigation of the chronic effects of air pollution and noise on CVD takes place (REGICOR-AIR).

Methodology: Outdoor annual mean concentrations of nitrogen dioxide (NO_2) derived from monthly passive sampler measurements were obtained at 83 residential locations. Long-term average traffic-related noise levels from a validated model were assigned to each residence. Linear regression models were fitted both for NO_2 and noise.

Results: The correlation between NO_2 and noise ($L_{24 h}$) was 0.62. However, the correlation differed across the urban space, with lower correlations at sites with higher traffic density and in the modern downtown. Traffic density, distance from the location to the sidewalk and building density nearby explained 35.6% and 73.2% of the variability of NO_2 and noise levels, respectively. The correlation between the residuals of the two models suggested the presence of other unmeasured common variables.

Conclusions: The substantial correlation between traffic-related noise and NO_2, endorsed by common determinants, and the dependence of this correlation on complex local characteristics call for careful evaluations of both factors to ultimately assess their cardiovascular effects.

1. Introduction

Noise is a well-known health hazard that disturbs sleep and activities, and affects cognitive and emotional responses (Muzet, 2007). Exposure to noise levels defined as unacceptable – above 65 dBA outdoors – affects a substantial proportion of the European population (20%), being particularly high in Spain (74%) (European Commission, 1996; OECD, 1993). Moreover, lower exposures to noise (45–55 dBA) could be hazardous as well (Ising and Kruppa, 2004). There is increasing evidence that chronic exposure to transportation-related noise contributes to cardiovascular effects (Babisch, 2006) through subcortical stress reactions (Ising and

Kruppa, 2004), and that short- and long-term night-time exposure is particularly relevant for hypertension (Haralabidis et al., 2008; Jarup et al., 2008).

Road traffic is also a major source of air pollution, an environmental stressor that may lead to cardiovascular disease (CVD) through oxidative stress and inflammation (van Eeden et al., 2001). Proximity to roads is a marker of exposure to air pollution (HEI, 2010) that, in turn, has been associated with cardiovascular mortality and atherosclerosis (Hoek et al., 2002; Hoffmann et al., 2007).

While the mechanisms leading to CVD may well differ for traffic-related noise and traffic-related air pollution, some pathways may be interrelated and ultimately contribute to the same ailments, such as atherosclerosis. Thus, if exposure to these traffic-related factors is highly correlated and both are causally associated with the cardiovascular outcome, studies analysing the long-term association between traffic-related air pollution and CVD may be confounded by road traffic noise and vice-versa, resulting in misleading conclusions. Few epidemiological studies have mutually adjusted for these potential confounders (Beelen et al., 2008; de Kluizenaar et al., 2007; Klaeboe et al., 2000; Selander et al., 2009; Tobias et al., 2001) and evaluations were mostly based on the comparison of modelled noise with modelled pollutants. As models use partly the same input variables for both environmental factors, derived correlations from models may not well reflect the true conditions.

Few studies have characterised the spatial correlation between both factors (Allen et al., 2009; Davies et al., 2009; Tang and Wang, 2007; Weber and Litschke, 2008). These studies – none in Mediterranean areas – indicate that the correlation structure between outdoor traffic-related noise and air pollution may depend on local factors, thus differ between cities. Whether and to what extent these correlations may vary among cities has not been investigated.

Therefore, the aim of this study is, first, to evaluate the correlation between the annual average concentration of measured nitrogen dioxide (NO_2) and the long-term average level of modelled traffic-related noise taken at different locations throughout Girona city, Spain; second, to analyse the contribution of traffic and the urban structure to the spatial distribution of each environmental factor and to their correlation. This study is part of the REGICOR-AIR study, a population-based cohort investigation evaluating the association between long-term exposure to air pollution as well as noise and atherosclerosis in the province of Girona (www.regicor.org).

2. Materials and methods

This study analysed the correlation between the long-term average of measured NO_2 and the long-term average of modelled noise at 83 locations distributed around Girona town. In the present manuscript, the terms *noise* and *air pollution* refer to *traffic-related noise* and *traffic-related air pollution*, respectively.

2.1. City of study

The city of Girona, located in Catalonia (north-eastern Spain), has 94 484 inhabitants, a surface of 39.1 km² (2415 inhabitants/km²) (Idescat, 2008) and an urban area of less than 10 km² (UMAT, 2009). It is a typical mid-sized Mediterranean urban area with a densely populated centre where traffic is expected to be the main determinant of the local variation of both air pollution and noise. Industries and the airport are located outside the city, thus, they do not contribute to the local inner-city contrasts of these ambient factors. We collected data for NO_2, noise and urban and traffic characteristics for 83 locations. These sites were selected to represent the full range of traffic density, street canyons and population density across the populated neighbourhoods of Girona city. All locations were homes or workplaces of REGICOR-AIR subjects and collaborators that volunteered for the NO_2 campaigns (see below).

2.2. Annual mean of nitrogen dioxide concentration

We used NO_2 as an indicator of traffic-related air pollution (Beckerman et al., 2008). For each site, the annual mean was estimated using the NO_2 monthly measurements taken with *Palmes* passive samplers supplied by AEA, Energy & Environment (London). In line with the standard protocol used by AEA elsewhere, as well as in a previous Girona-based measurement campaign, tubes were deployed for an one-month period, with duplicates and blanks in 10% and 2% of the locations, respectively. Tubes were mailed to the participants with instructions for deployment and dismantling at the start and end date of the one-month measurement period. Participants placed the tubes in their balconies. A total of 12 measuring campaigns were conducted between August 2007 and June 2008. Based on previous monitoring data it was known that the March–May period was usually very close to the annual mean. Therefore, we organised a large campaign for the April/May 2008 period that involved 77 sites simultaneously. In addition, measurements were repeated in some locations, resulting in two sites participating during three periods and 18 being served twice.

During the entire study period we also conducted parallel monthly measurements with *Palmes* tubes at the city-operated continuous NO_2 monitoring station to capture the seasonal pattern of NO_2 and to calibrate the passive samplers results.

The estimation of the annual mean concentration of NO_2 at each location i (AC_i) was calculated as follows. First, the NO_2 results of our *Palmes* tubes were multiplied by the derived calibration factor of 0.92. Second, every NO_2 measurement taken during time period t (C_{it}), was temporally detrended using the mean concentrations at the reference station during the same period t (C_{0t}) and the fixed station's annual mean (AC_0) according to Eq. (1)

$$AC_i = \frac{C_{it}}{C_{0t}} AC_0 \tag{1}$$

At sites with more than one measurement, the annual mean was derived as the time-weighted average of Eq. (1) for each measurement and its duration.

2.3. Modelled road traffic noise levels

Traffic noise levels were estimated from the traffic noise model of Girona, elaborated in 2005 by the University of Girona as a response to the European Union Directive 2002/49/EC for traffic noise mapping. It was based on the interim European noise model for road noise NMPB routes-96 (CERTU/CSTB/LCPC/SETRA, 1997). The main input variables were: the slopes, the type of asphalt, the streets' geometry (e.g. height of buildings) and traffic density in the city (Deltell, 2005). The model, with a grid of 5 × 5 m, provides estimates of the long-term average level of traffic-related noise during day and night: L_{day} (7am–11pm) and L_{night} (11pm–7am), respectively.

The noise model was validated with 120 noise measurements (118 L_{day} and two $L_{24\,h}$) distributed throughout the city and performed at 1.5 m from the ground and 1–2 m from the façades. Measurements were taken with a CESVA SC20c and a CESVA SC30 sound level metre and a CESVA CB-5 calibrator. No measurements were done during extreme weather conditions, defined as wind above 4 m/s, rain or wet asphalt. Measured and modelled values did not differ more than 3 dB and model's R^2 was very high (0.93).

Noise predictions for the L_{day} and L_{night} indicators were computed by numerical calculations in CADNA/A software (from DataKustic), which implements the NMPB-routes 96 model, among others. L_{day} and L_{night} were estimated for all of our geo-referenced locations at the façade of the building and at the passive samplers' height. Noise was estimated in dBA (A-weighted decibels), i.e. decibels adjusted to the human ear perception (Directive 2002/49/EC). The A-weighted long-term average sound level for 24 h ($L_{24\,h}$) was derived as the time-weighted logarithmic mean of L_{day} and L_{night} (Eq. (2)). Evening values (9pm–11pm) were included in L_{day}, as L_{day} and $L_{evening}$ are very similar in Spain and $L_{evening}$ represents only a 2 h time window (Catalan Government Order, 176/2009).

$$L_{24\,h} = 10\log_{10}\left(\frac{16 \times 10^{L_{day}/10} + 8 \times 10^{L_{night}/10}}{24}\right) \tag{2}$$

2.4. Determinants of NO_2 and noise

Characteristics of the sampling locations potentially influencing noise and air pollution levels were derived manually from the web map of Girona's City Council (UMAT, 2009) and consisted of urban and traffic characteristics as well as degree of urbanisation (a) *in front of the location* (*number of street lanes, traffic direction, street width* and *number of bus lines*); (b) *in radius buffers*—used in other studies (Davies et al., 2009; Hoek et al., 2008); e.g. *presence of the river basin within 50 m*); (c) *in a segment of the street in front of the location* (*e.g. building density*) and (d) *nearest horizontal distance from the location to an urban structure*, namely: *distance to the sidewalk* and *distance to the nearest crossroad*.

The degree of urbanisation was characterised for segments of 50 m or 150 m in both directions along the street in front of each measurement location (Table 1). These variables were *building density* (defined as "isolated house" or "one side of the street built" or "two sides built"), *number of open areas* (defined as an open space-no building of more than 15 m deep × 15 m wide) and *number of gaps* (defined as open space of more than 15 m deep and < 15 m width).

Table 1

Description of the potential determinants and crude linear regressions between all potential determinants and both NO_2 and $L_{24\,h}$ levels ($n=83$).

Potential determinants Continuous variables	Median (Interquartile range)	NO_2 levels			$L_{24\,h}$ levels		
		Regression coefficient[a]	95% confidence interval	R^2	Regression coefficient[a]	95% confidence interval	R^2
Height of the location (m)	5.70 (6.90)	2.16	−0.48; 4.81	0.03	1.24	−0.24; 2.72	0.03
Height of the building (m)	11.2 (11.2)	4.45	1.24; 7.66	0.09	2.50	0.71; 4.30	0.09
Height opposite building (m)	11.2 (11.2)	7.64	4.22; 11.06	0.20	2.48	0.42; 4.55	0.07
Average daily traffic (veh./day)[b]	1000 (6500)	0.33[b]	0.19; 0.47	0.21	0.32[b]	0.26; 0.37	0.60
Average night-time traffic (veh./day)[b]	10 (60)	0.33[b]	0.19; 0.47	0.22	0.32[b]	0.26; 0.37	0.60
Heavy duty day (veh./day)[b]	5.88 (16.68)	0.30[b]	0.17; 0.42	0.21	0.28[b]	0.23; 0.33	0.59
Heavy duty night (veh./day)[b]	0.42 (2.60)	0.23[b]	0.13; 0.33	0.21	0.22[b]	0.18; 0.26	0.59
Crossroads within 50 m (along street[d]) (n)	1.00 (1.00)	1.71	−0.89; 4.31	0.02	1.82	0.41; 3.24	0.08
Crossroads within 150 m (along street[d]) (n)	3.00 (2.00)	4.54	1.49; 7.60	0.10	3.34	1.71; 4.98	0.17
Distance to the sidewalk (m)[b]	4.70 (5.70)	−0.28[b]	−0.45; −0.10	0.05	−0.15[b]	−0.25; −0.05	0.08
Street width (m)	8.00 (3.80)	3.85	1.82−5.88	0.15	1.95	0.80; 3.11	0.12
Distance to nearest crossroad (m)	28.00 (32.4)	−2.26	−4.05; −0.48	0.07	−1.28	−2.28; −0.28	0.08
Street lanes (n)	2.00 (1.00)	1.72	−0.35; 3.78	0.03	1.29	0.15; 2.43	0.06
Bus lines in front of location (n)	0 (1.00)	2.12	0.23; 4	0.06	1.87	0.86; 2.87	0.14
Bus lines within 100 m (along street[d]) (n)	0 (1.00)	1.12	0.05; 2.19	0.05	0.77	0.18; 1.36	0.08
Bus lines within 150 m radius buffer (n)	2.00 (2.00)	2.93	1.18; 4.68	0.12	1.41	0.41; 2.40	0.09
Bus stops within 50 m radius buffer (n)[c]	0 (0)	1.73*,[c]	−1.53; −4.99	0.01	1.92[c]	0.13; 3.70	0.05
Bus stops within 150 m radius buffer (n)	2.00 (2.00)	3.09	0.04; 6.13	0.05	1.77	0.06; 3.47	0.05

Categorical variables	n (%)	Regression coefficient	95% confidence interval	R^2	Regression coefficient	95% confidence interval	R^2
Building density within 150 m (along street[d])							
Isolated house	11 (13.3)	Ref.	Ref.	0.26	Ref.	Ref.	0.30
1 side built	10 (12.1)	10.59	3.05; 18.13		8.65	4.57; 12.73	
2 sides built	62 (74.7)	14.73	9.09; 20.38		9.01	5.96; 12.07	
Traffic in both directions							
Yes (vs. No)	44 (53.0)	−2.72*	−7.04; 1.60	0.02	−0.36*	−2.80; −2.09	< 0.01
River within 50 m radius buffer							
Yes (vs. No)	8 (9.6)	2.26*	−5.11; 9.63	0.01	1.46*	−2.66; 5.58	0.01
River within 150 m radius buffer							
Yes (vs. No)	13 (15.7)	2.01*	−3.98; 7.99	0.01	0.87*	−2.48; 4.22	< 0.01
Open areas within 50 m (along street[d]) (n)							
Few (0–1)	46 (55.4)	Ref.	Ref.	0.26	Ref.	Ref.	0.36
Some (2–4)	25 (30.1)	0.19	−4.07; 4.45		−0.26	−2.49; 1.98	
Many (> 4)	12 (14.5)	−14.34	−19.90; −8.78		−9.44	−12.35; −6.52	
Open areas within 150 m (along street[d]) (n)							
Few (0–2)	34 (41)	Ref.	Ref.	0.26	Ref.	Ref.	0.36
Some (3–9)	37 (44.6)	0.08	−3.99; 4.16		0.21	−1.92; 2.35	
Many (> 9)	12 (14.5)	−14.37	−20.12; −8.60		−9.24	−12.25; −6.22	
Gaps within 50 m (along street[d]) (n)							
No gaps (0)	59 (71.1)	Ref.	Ref.	0.28	Ref.	Ref.	0.31
Some (1–4)	14 (16.9)	−1.88	−6.92; 3.15		0.68	−3.44; 2.08	
Many (> 4)	10 (12.1)	−16.29	−22.09; −10.50		−9.56	−12.74; −6.39	
Gaps within 150 m (along street[d]) (n)							
Few (0)	48 (57.8)	Ref.	Ref.	0.28	Ref.	Ref.	0.31
Some (1–7)	25 (30.1)	−1.28	−5.46; 2.91		0.66	−1.63; 2.94	
Many (> 7)	10 (12.1)	−16.37	−22.26; −10.47		−9.21	−12.43; −5.98	

R^2: coefficients of determination, veh.: vehicles.

* p-value < 0.2 for a crude linear regression.
[a] For continuous determinants, the regression coefficients and confidence intervals are expressed as the increase in NO_2 or $L_{24\,h}$ for an interquartile range increase in the potential determinant, except in coefficients marked with [b] and [c].
[b] Variables logarithmically transformed to fit a crude linear regression model. Their coefficients and confidence intervals are interpreted as the change in NO_2 or $L_{24\,h}$ for a 10% increase in the potential determinant.
[c] The coefficient and confidence intervals are interpreted as the increase in NO_2 or $L_{24\,h}$ for an increase of one bus stop.
[d] Variables calculated in a street segment centred in front of the location, e.g. we count the number of gaps 50 m in one direction of the street and 50 m in the other direction of the street.

Likewise, we also examined the following traffic and urban configuration variables used as input data of the noise model: height of the building of the sampling location, height of the opposite building, height of the location, average daily traffic (ADT) density, average night-time traffic density and average day- and night-time density of heavy duty vehicles.

All considered variables are shown in Table 1.

2.5. Statistical analysis

To identify the main determinants of the spatial distribution of NO_2 and of noise and to evaluate the contribution of these determinants to the levels of both factors, regression models for NO_2, $L_{24 h}$ and L_{night} were fitted according to the following steps. Firstly, we carried out a systematic univariate analysis of all variables, with an evaluation of their normality. Secondly, linearity of the associations between each potential predictor and each environmental factor (NO_2, $L_{24 h}$ and L_{night}) was assessed graphically. In those instances where linearity did not hold, predictors were log-transformed (namely, the traffic density variables and the *distance to the sidewalk*) or categorised according to meaningful categories, e.g. *open areas* ("few", "some" and "many") and *gaps* ("no gaps", "some" and "many"). NO_2, $L_{24 h}$ and L_{night} were normally distributed, thus not transformed. In addition, to assess potential collinearity between predictors, we computed Spearman correlation coefficients for the continuous variables and the percentage of agreement for categorical variables. Due to their high collinearity, separate models were derived for the different traffic density variables and also for the degree of urbanisation variables (namely *building density, open areas* and *gaps*). Regarding the degree of urbanisation variables, we only present the models for *building density*. Separate models were also derived for the 50 and 150 m radius buffers. Thirdly, all variables associated with the outcomes with a *p*-value < 0.2 entered the initial saturated models. Backward regression was then performed, excluding the variable with the highest *p*-value at each step. The final model included those variables with a *p*-value < 0.05. Regression diagnostic tests included residuals' normality, homoscedasticity, linear relationships, multicollinearity and influential data. The model residuals were tested for spatial correlation using Moran's I.

The correlation between NO_2 and $L_{24 h}$ was evaluated graphically and with Pearson correlation coefficient. We also evaluated whether there was a remaining statistically significant correlation between the residuals from the NO_2 multivariate linear regression model and those from the $L_{24 h}$ models, which would indicate that the final predictors of the models explain only part of the spatial correlation between the two factors. We also stratified the correlation NO_2-$L_{24 h}$ by high and low *ADT* using the median as cut-off point (1000 vehicles/day). The difference between strata was tested by including an interaction term between the binary variable and NO_2 in a linear model for $L_{24 h}$. The residual spatial correlation was tested using Moran's I.

In a subsequent analysis we considered that the correlation of traffic-related noise and NO_2 may be different throughout the city due to differences in the urban characteristics. Therefore, we stratified the NO_2-$L_{24 h}$ correlation by 'city centre' and 'outskirts' and also tested the differences between strata. Secondly, we used a model-based exploratory analysis, the geographically weighted regression (GWR), to visualise the change in the space of the relationship NO_2-$L_{24 h}$. The GWR computed a linear regression model between $L_{24 h}$ and NO_2 (dependant variable: $L_{24 h}$) for each location, giving an inverse distance-weighting to the surrounding locations up to a distance predefined by the GWR itself (652 m). Afterwards we tested the spatial variation comparing this model with a traditional regression model (with one coefficient for the entire city; Fotheringham et al., 2002). Finally, we stratified the NO_2-$L_{24 h}$ correlation according to the patterns obtained from the GWR model results and tested the differences between these strata as described above.

The analyses were performed with Stata 8.2 and R 2.6.0.

3. Results

In the city of Girona, on average, the $L_{24 h}$ was 63.3 dBA (range: 47.9–72.9 dBA, IQR: 8.22 dBA), the L_{night} was 55.7 dBA (range: 40.3–66.3 dBA, IQR: 8.30 dBA) and the annual concentration of NO_2 was 26.9 $\mu g/m^3$ (range: 6.5–53.0 $\mu g/m^3$, IQR: 12.16 $\mu g/m^3$). A total of 62 locations were in streets with buildings at both sides. All traffic and land-use variables are described in Table 1.

The *Palmes* diffusion tubes had a good precision with a coefficient of variation lower than 5% across all duplicate measurements. The bivariate analyses showed a significant linear association and a rather high R^2 between NO_2 and *height of the opposite building*, traffic density and degree of urbanisation variables (Table 1). For $L_{24 h}$ the relationship was stronger with the traffic density variables, but the number of *crossroads within* 150 m, and *bus lines in front of the site* had also rather high R^2 values. Results for L_{night} and $L_{24 h}$ were very similar (data not shown).

The Pearson correlation coefficient between the long-term estimates of NO_2 and $L_{24 h}$ was 0.62 (95% CI: 0.46; 0.73). The multivariate linear regression models are presented in Table 2. *Building density*, *distance to the sidewalk* and *ADT* explained 73% of the variability of $L_{24 h}$ whereas these same determinants plus *height of the opposite building* and *street width* explained 46% of the variability of the annual averages of NO_2. The models satisfied the regression diagnostics.

We also analysed the relevance of the variables common to both models, namely *building density, ADT* and *distance to the sidewalk*. Those explained 73% and 36% of the variability of $L_{24 h}$ and NO_2, respectively (data not shown). The models in Table 2 indicated that locations with "2 sides built" had 6.13 $\mu g/m^3$ higher NO_2 levels and 4.61 dBA higher $L_{24 h}$ levels as compared with "isolated houses". With every 10% increase in the *distance to the sidewalk*, noise levels were reduced by 0.07 dBA and NO_2 levels by 0.17 $\mu g/m^3$. An increase in 10% in *ADT* was associated with a 0.18 $\mu g/m^3$ increase in NO_2 and a 0.27 dBA increase in noise, respectively. Regarding the determinants that differed between models, an interquartile range increase in *height of the opposite building* (IQR = 11.2 m) and in *street width* (IQR = 3.80 m) resulted in NO_2 increases of 4.76 and 2.25 $\mu g/m^3$, respectively. These determinants were not associated with $L_{24 h}$.

Table 2

Regression coefficients (β) and 95% confidence intervals (CI) of the multivariate linear regression models for NO_2, $L_{24 h}$ and L_{night} including all significant urban determinants at $\alpha = 0.05$.

Urban determinants	Model for NO_2		Model for $L_{24 h}$		Model for L_{night}	
	$R^2 = 0.46$		$R^2 = 0.73$		$R^2 = 0.79$	
	Regression coefficient[a]	**95% confidence interval**	**Regression coefficient**[a]	**95% confidence interval**	**Regression coefficient**[a]	**95% confidence interval**
Building density						
Isolated house	Ref.		Ref.		Ref.	
1 side built	2.76[c]	−4.09; 9.61	4.36	1.75; 6.98	4.79	2.38; 7.20
2 sides built	6.13	0.58; 11.69	4.61	2.55; 6.68	5.05	3.15; 6.95
Height opposite building (m)	4.76	1.70; 7.83	–	–	–	–
Average daily traffic (veh./day)	0.18[b]	0.04; 0.32	0.27[b]	0.22; 0.32	–	–
Distance location to sidewalk (m)	−0.17[b]	−0.32; −0.01	−0.07[b]	−0.13; −0.02	−0.05[b]	−0.11; −0.003
Street width (m)	2.25	0.30; 4.19	–	–	–	–
Average night-time traffic (veh./day)	–	–	–	–	0.29[b]	0.25; 0.34
Constant	13.50	7.20; 19.79	58.64	56.48; 60.81	64.86	61.75; 67.98

R^2: coefficients of determination, veh.: vehicles.

[a] For continuous determinants, the regression coefficients and confidence intervals are expressed as the increase in NO_2 or $L_{24 h}$ for an interquartile range increase in the potential determinant, except in coefficients marked with [b] and [c].

[b] Variables logarithmically transformed to fit a crude linear regression model. Their coefficients and confidence intervals are interpreted as the change in NO_2 or $L_{24 h}$ for a 10% increase in the potential determinant.

[c] All regression coefficients are significant for a *p*-value < 0.05 in a multivariate linear regression model, except coefficient c.
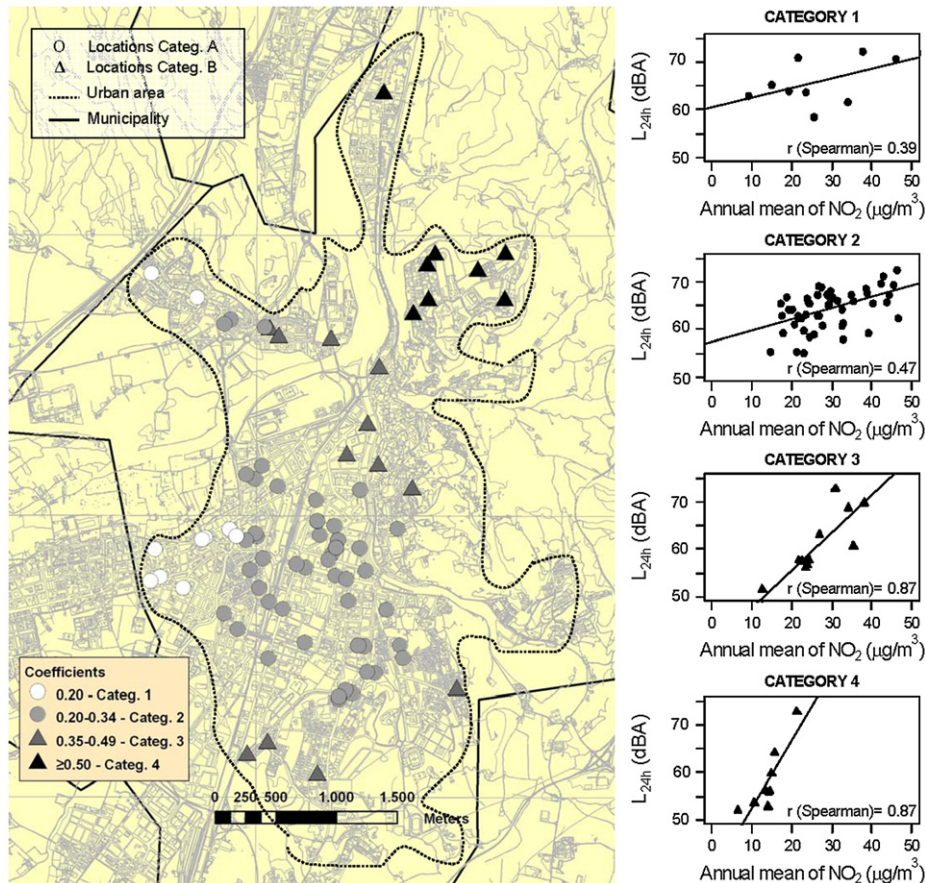
Fig. 1. Spatial variation of the regression coefficient based on a geographically weighted regression between NO$_2$ and L$_{24 h}$ in the study locations of the city of Girona. In the map, the continuous black line represents the municipality border of Girona, whereas the dotted line represents the urban area. The locations are represented by points (Category A) and triangles (Category B) and by a colour scale of 4 categories depending on the regression coefficient at the location. The scatter plots on the right represent the correlation between the 24 h long-term average sound level (L$_{24 h}$) and the annual mean concentration of NO$_2$ for each of the four categories in the map.

The Pearson correlation between the residuals of the models for NO$_2$ and L$_{24 h}$ shown in Table 2 was 0.27 (95% CI: 0.06; 0.46).

The correlation of NO$_2$ with L$_{night}$ ($r = 0.61$) was almost identical to the one with L$_{24 h}$. The model for L$_{night}$ included the variables *building density, average night-time traffic density* (instead of ADT) and *distance to the sidewalk,* which explained 79% of the variability of this acoustic indicator (Table 2). This model showed some heteroscedasticity. When computing robust standard errors, the variable *distance to the sidewalk* slightly decreased its significance (p-value = 0.06, 95% CI: -0.11; 0.003).

The alternative models for NO$_2$, L$_{24 h}$ and L$_{night}$ using the variables *gaps* or *open areas* (50 and 150 m) instead of *building density* gave similar results with similar R^2s to the ones described above. Overall, these models had lower R^2s when using the *density of heavy duty* variables (instead of the total traffic density), and *buses in front of the location* was retained in the L$_{24 h}$ model when using this variable. No spatial correlation was found in the residuals of these models (p-values > 0.25).

In a next step, we stratified the L$_{24 h}$–NO$_2$ correlation by traffic density (cut-off: 1000 vehicles/day). The correlation between L$_{24 h}$ and NO$_2$ was stronger in the low *ADT* group (Spearman rank = 0.60 versus 0.47), but the interaction did not reach statistical significance (p-value for interaction = 0.247). The same held for the apparent difference in the Pearson correlation in downtown sites (0.48) versus outskirts sites (0.63) (p-value for interaction = 0.247).

The comparison between the GWR model and the traditional regression model for the entire city revealed a statistically significant spatial variation in the relationship between NO$_2$ and L$_{24 h}$

(p-value = 0.007) as Fig. 1 shows. The interaction analyses were statistically significant for a variable with four categories corresponding to the four areas of the figure (p-value of interaction = 1.517×10^{-5}). The plots in Fig. 1 depict differences in the NO$_2$–L$_{24 h}$ relationship mainly in categories 1 and 2 (category A) versus categories 3 and 4 (category B). The Spearman correlation stratifying by these two areas were 0.46 (95% CI: 0.24;0.64) and 0.79 (95% CI: 0.54; 0.91), respectively—a statistically significant interaction (p-value = 0.003).

No residual spatial correlation was found in any of the interaction analyses (p-value > 0.1 for Moran's I test).

4. Discussion

This is the first study assessing the spatial correlation between NO$_2$ and noise and its determinants in Spain and, to our knowledge, in a Mediterranean town where dense urban structures with street canyons and high traffic prevail. In this study, the correlation between modelled L$_{24 h}$ and measured NO$_2$ was 0.62; degree of urbanisation, traffic density and distance to the street were common determinants of L$_{24 h}$ and NO$_2$ explaining part of the moderate to high correlation between noise and NO$_2$.

4.1. Determinants and correlation between NO$_2$ and L$_{24 h}$

The moderately high correlation between long-term average levels of modelled L$_{24 h}$ and of measured NO$_2$ was in line with the spatial correlations found in Chicago, Riverside and Vancouver for

5- min noise measurements and 2-week NO_2 measurements (Allen et al., 2009; Davies et al., 2009). Our correlation was higher than that found in a Dutch study for modelled black smoke and modelled noise ($r=0.24$; Beelen et al., 2008) and in Oslo for 24 h modelled noise and the 3-month mean of modelled NO_2 ($r=0.46$) (Klaeboe et al., 2000). Finally, our correlation was lower than that reported in Groningen for modelled noise and modelled PM_{10} ($r=0.72$) (de Kluizenaar et al., 2007). To what extent the models for noise and pollutants were based on the same input variables was not always described in these studies. Local factors may be important determinants of the heterogeneity in these correlations. Moreover, studies used different markers of traffic-related pollution, e.g. NO_2 or PM_{10}, whose local spatial patterns do however differ substantially. Thus, correlations with noise are expected to differ as well.

Building density nearby is clearly relevant for both NO_2 and noise in our models. While there are many ways to characterize building density, results with the simpler metrics were very similar to those based on the more complex and not readily available data of number and size of open spaces.

The common determinants found for $L_{24\,h}$ and NO_2 (namely *building density*, *ADT* and *distance to the sidewalk*) explained only part of the correlation between these exposures (Table 2). These variables had been previously associated with the increase of NO_2 concentrations (Rijnders et al., 2001). In our study, they also explained most of the $L_{24\,h}$ levels. Similar variables were retained in a NO_2 model in Vancouver, where the correlation noise–NO_2 was similar to ours (Davies et al., 2009). In that study, the noise–NO_2 relationship was mainly explained by the number of lanes on the nearest road, the presence of a major intersection and the traffic density. However, they used a single multivariate linear regression model with noise as independent variable and NO_2 as dependant variable. They did not evaluate separately the determinants of the spatial variability of each environmental factor. Similar to our study, the proximity to traffic (*distance from the sidewalk*) was a good determinant. These results support the statement of Allen et al. (2009), who cautioned that the simple proximity measures may similarly be a surrogate of noise and air pollution, posing challenges in the investigation of health effects possibly caused by one or the other factor.

The only determinants differing between the $L_{24\,h}$ and the NO_2 model were the *height of the opposite building* and the *street width* which were significant positive predictors of the NO_2 concentrations. In Girona, building height may be a good proxy for street canyons, which have known effects on local pollution (Tang and Wang, 2007), while street width might be related to the number of street lanes and, thus, to some patterns of traffic, such as traffic jams in wider main roads, which may not be well reflected in the *ADT* variable. Therefore, in street canyon conditions and in some traffic patterns, the correlation between noise and NO_2 may differ. Alternatively, as suggested by Davies et al. (2009), these two predictors may indicate differences in the dispersion properties of sound waves and gaseous pollutants (NO_2).

From a health perspective, exposure to night-time noise may be of particular relevance (Jarup et al., 2008). In our study, the correlation between L_{night} and $L_{24\,h}$ was so high ($r=0.99$) that results for L_{night} were very similar to those of $L_{24\,h}$, thus, conclusions related to $L_{24\,h}$ also apply to the night conditions.

4.2. Modifiers of the correlation $L_{24\,h}$–NO_2

Unlike the street canyon effect described above, to our knowledge, no studies assessed the effect of traffic density on the correlation between noise and air pollution. Our data suggest that the noise–NO_2 correlation may be stronger at locations with low traffic density. Although differences were not significant, probably due to the limited number of sites, this interaction has some plausibility as noise, unlike

NO_2, has a logarithmic trend with increase in traffic density. More research is needed to understand such interactions as they may be very relevant in epidemiological studies.

The GWR approach indicated that the NO_2–$L_{24\,h}$ relationship varied in space (Fig. 1). However, as this method computes regression coefficients at each point based mainly on data from nearby locations, coefficients and R^2s in more isolated sites may have more variability and be misleading. However, the spatial variation of the NO_2 and $L_{24\,h}$ relationship was not sensitive to the exclusion of the isolated point in the North (data not shown). Our category A represents the modern dense downtown area and category B the outskirts and the narrow urban layout of the small historical centre, with several pedestrian areas. This is in line with the different NO_2–$L_{24\,h}$ correlation found by the pre-defined categories of city centre/outskirts, although the interaction analyses were not statistically significant. Taking into account that the most important source of NO_2 in our study area is traffic, these differences may be due to land-use differences by areas, e.g. to the higher proportion of street canyons downtown compared to the outskirts. Further analyses by city areas may help to ascertain the spatial variation of the noise–NO_2 relationship and disentangle the cardiovascular long-term effects of noise and air pollution.

4.3. Strengths and limitations

The main strength of our study is that the analyses were based on a large number of NO_2 measurements covering the range of traffic density and urban space in Girona city. NO_2 outdoors is a good surrogate of traffic-related pollution (Beckerman et al., 2008). If our measurements were affected by other NO_2 sources, this would result in an underestimation of the NO_2–$L_{24\,h}$ correlation. However, we are not aware of other local sources of NO_2 in the area, thus the large local contrasts observed for NO_2 possibly reflect the impact of the dense urban structure with street canyons. This is supported by the lack of spatial autocorrelation in the NO_2 concentrations based on Moran's I test.

This study used the best information available and a large number of objective data on street configuration, traffic and proximity to the sites. With the exception of *ADT*, none of the covariates directly used to derive the parametric component of Girona's noise model were included in our analyses. In contrast to most other comparisons of noise and pollution in epidemiological studies, we used measured instead of modelled NO_2. Moreover, although noise values were based on a model, predictions and measurements have been shown to be highly correlated (R^2 of the validation$=0.93$). Furthermore, the noise estimations calculated at the façade and at the same height of the NO_2 sampler – similarly to the measured NO_2 concentrations – may be rather good estimates of the outdoor conditions found at residential sites. Studies deriving noise and air pollution from models using the same covariates, but from different sources or quality, may underestimate the true association between these environmental conditions. Thus, claims of low or moderate correlations with modelled data could be misleading.

Regarding our linear regression models, regression coefficients were of comparable precision even if we used NO_2 measurements and modelled noise and obtained different R^2s. The different measurement error in the response would not change the regression coefficients (Carroll et al., 2006), although it could change the *p*-values and R^2s.

It should be noted that the noise estimates, derived with a noise model of 2005, are representative of the period of the NO_2 measurements (2007/2008), because noise levels are rather constant over time (Beelen et al., 2008) and we are not aware of major changes in traffic organisation in Girona during this period.

The significant correlation found between the residuals of both models clearly indicates that noise and NO_2 have other common predictors, not identified in our data. Among the unavailable data, we lacked meteorological information which has been reported to affect the air pollution-noise correlation at the local level. In Essen, Germany, the highest correlation for these factors was found with weak air turbulences (Weber and Litschke, 2008). Another study in the USA reported changes in the correlation in streets' downwind and upwind but only had information on wind speed and direction at a single location in each city (Allen et al., 2009). While local wind patterns in street canyons may explain some of the residual correlation between noise and NO_2, availability of local street-level wind information may be extremely difficult to get.

Finally, our study – like many others before – evaluated outdoor levels of the environmental factors, whereas people spend most of their time indoors. In fact, what matters in epidemiological research is people exposure rather than noise or air pollution at the outdoor façade. These results are relevant for many epidemiological studies that can only describe exposure based on models of outdoor traffic-related noise and air pollution. However, further research is needed to better understand the association between ambient conditions and personal exposure to transportation-related noise, which – for the effects of noise on CVD – most strongly relates to the night-time.

5. Conclusions

The substantial correlation found between the long-term average of traffic-related 24 h noise levels and the annual average of NO_2 concentrations, as well as the many common determinants of the spatial distribution of both factors, suggests that noise could confound the long-term effects of road traffic air pollution on cardiovascular health and vice-versa. Apparent 'low correlations' between these factors, as published in some studies, may be due to differences in urban structure or to the different indicators used compared with our study, but modelling artefacts may also play a role in studies that did not have measurements available. Our results suggest that epidemiological studies should include a detailed local assessment of both environmental factors. Further efforts to disentangle noise and air pollution effects should focus on the spatial determinants of the correlation between the two and on validation studies with personal exposure. In the case of noise, adaptive behaviour need to be carefully integrated as well, as it may substantially alter the true exposure to noise while indoors (e.g. wearing ear plugs during sleep).

Competing interests declaration

This study has no competing interests.

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