



Façades, floors and maps – Influence of exposure measurement error on the association between transportation noise and myocardial infarction



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ABSTRACT

Background: Epidemiological research on transportation noise uses different exposure assessment strategies based on façade point estimates or regulatory noise maps. The degree of exposure measurement error and subsequent potentially biased risk estimates related to exposure definition is unclear. We aimed to evaluate associations between transportation noise exposure and myocardial infarction (MI) mortality considering: assumptions about residential floor, façade point selection (loudest, quietest, nearest), façade point vs. noise map estimates, and influence of averaging exposure at coarser spatial scales (e.g. in ecological health studies).

Methods: L_{den} from the façade points were assigned to > 4 million eligible adults in the Swiss National Cohort for the best match residential floor (reference), middle floor, and first floor. For selected floors, the loudest and quietest exposed façades per dwelling, plus the nearest façade point to the residential geocode, were extracted. Exposure was also assigned from 10×10 m noise maps, using “buffers” from 50 to 500 m derived from the maps, and by aggregating the maps to larger areas. Associations between road traffic and railway noise and MI mortality were evaluated by multi-pollutant Cox regression models, adjusted for aircraft noise, NO_2 and socio-demographic confounders, following individuals from 2000 to 2008. Bias was calculated to express differences compared to the reference.

Results: Hazard ratios (HRs) for the best match residential floor were 1.05 (1.02–1.07) and 1.03 (1.01–1.05) per IQR (11.3 and 15.0 dB) for road traffic and railway noise, respectively. In most situations, comparing the alternative exposure definitions to this reference resulted in attenuated HRs. For example, assuming everyone resided on the middle or everyone on first floor introduced little bias (%Bias in excess risk: –1.9 to 4.4 road traffic and –4.4 to 10.7 railway noise). Using the noise grids generated a bias of approximately –26% for both sources. Averaging the maps at a coarser spatial scale led to bias from –19.4 to –105.1% for road traffic and 17.6 to –34.3% for railway noise and inflated the confidence intervals such that some HRs were no longer statistically significant.

Conclusion: Changes in spatial scale introduced more bias than changes in residential floor. Use of noise maps to represent residential exposure may underestimate noise-induced health effects, in particular for small-scale heterogeneously distributed road traffic noise in urban settings.

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1. Introduction

The last decade has seen an increase in the number of studies on health effects of transportation noise, in particular on road traffic and aircraft noise in relation to cardiovascular outcomes (Foraster et al., 2017; H eritier et al., 2017; Munzel et al., 2017; van Kempen et al., 2018; Vienneau et al., 2015). Noise nuisance and exposure have typically been evaluated through self-reported annoyance surveys (Brink et al., 2016; Guski et al., 2017), noise measurements (Belojevic et al., 2012; Lercher et al., 2011; Quehl et al., 2017) or exposure calculations (Garg and Maji, 2014; Karipidis et al., 2014), with the latter approach particularly appealing for large population studies.

The gold standard for exposure modelling in noise and health research, and used in our previous studies in Switzerland (Eze et al., 2017; Foraster et al., 2017; H eritier et al., 2017; H eritier et al., 2018b), is two or three-dimensional source-propagation noise models (e.g. CNOSSOS-EU, Harmonise (Kephalopoulos et al., 2014)) and software frameworks such as SoundPLAN and CadnaA. These techniques can be used to estimate exposure at defined fa ade points on buildings, or at a lattice of receptor locations used to generate noise maps. Sufficiently detailed input data, computer resources and skill for undertaking this type of modelling, however, can be a limitation for some health studies. As such, an open-source simplified version of the CNOSSOS-EU modelling method, evaluating different spatial resolutions of input data, has been developed for deriving noise exposures for large geographic areas where harmonized noise exposure estimates at address locations are needed (Morley et al., 2015).

When noise mapping is used in epidemiology, exposure is often assigned on the basis of the grid cell in which the residential address falls (Jarup et al., 2008). Noise mapping may also be used to calculate area-level average exposures in ecological or small-area studies, or in health impact assessments (Hansell et al., 2013). In comparing results of health studies, differences in the spatial averaging of noise levels must therefore also be taken into consideration. In general, spatial scale is also likely important when attempting to disentangle noise and air pollution effects (H eritier et al., 2018a).

While different exposure assessment strategies or decisions may influence comparability of epidemiological results, there is also uncertainty about the extent of exposure measurement error in each of these studies compared to the gold standard. For objectively evaluated exposure based on measurements or models, the measurement error is most likely to be non-differential and not related to disease status. Errors may relate, for example, to uncertainty in the exposure proxy, inaccuracies in residential geocodes or the activity of study participants who naturally do not spend all their time at home leading to incorrect assignment of exposure. In this situation, and if there is a true association with disease status, the effect estimates are typically biased toward the null for classical errors (R o sli and Vienneau, 2014) whereas no bias is expected for Berkson type errors although confidence intervals are inflated (Armstrong, 1998). In reality, error in exposure assessment is a mixture of both types of errors.

Against this background, a critical evaluation of the potential measurement error for noise and health studies is needed. Previous analyses in the Swiss National Cohort, a census-based cohort of virtually all adults in Switzerland, found source-specific transportation noise exposure to be associated with cardiovascular mortality, with most consistent results across noise sources for myocardial infarction (MI) (H eritier et al., 2017). Using these detailed data from Switzerland, we aimed to compare associations between road traffic and railway noise exposure and MI mortality considering different exposure definitions to evaluate: assumptions about residential floor, fa ade point selection (i.e. maximum (“loudest”), minimum (“quietest”) and nearest), difference in using estimates from fa ade points vs. noise maps, and influence of different spatial scales (i.e. individual vs. small area-level measures).

2. Methods

2.1. Study population

The Swiss National Cohort (SNC) links national census data with mortality and emigration records (Spoerri et al., 2010). The census data contain personal, household and building information. Cause and date of death is included in the mortality records. This analysis was based on the 4 December 2000 census and on mortality and emigration data for the period 5 December 2000 to 31 December 2008, which included 7.28 million observations. Observations below 30 years of age ($n = 2.59$ million), observations for which residential coordinates were missing ($n = 0.19$ million), those living in an institution ($n = 0.25$ million), and observations for which the cause of death was imputed (0.03 million) were excluded. Unlike our previous analyses on transportation noise and cardiovascular mortality (H eritier et al., 2017; H eritier et al., 2018b), we also excluded observations residing within 500 m of the Swiss country border to prevent edge effects in the analyses exploring spatial scale ($n = 0.06$ million). In total, 4.35 million observations (i.e. individuals) were included in the analyses.

The SNC was approved by the cantonal ethics boards of Bern and Zurich.

2.2. Noise exposure data

Within the framework of the SiRENE project (Short and Long Term Effects of Transportation Noise Exposure), an exposure database was constructed for the census years for Switzerland. It included the three major transportation noise sources: road traffic, railway and aircraft noise. Here we used the 2001 exposure data which coincided with the cohort baseline. Residential geocodes were also from the 2000 census, and address history during follow-up was not available.

The noise exposure database is detailed elsewhere (Karipidis et al., 2014). In brief, road traffic noise emissions were calculated using sonROAD (Heutschi, 2004) while propagation was computed via the StL-86 model (OFPE, 1987). Traffic volumes for minor roads were based on the traffic volume of the nearby arterial roads and an estimate of the local traffic depending on population and type of businesses, scaled to match the total sum of the nation-wide statistics. For railway noise, the emissions were calculated using sonRAIL (Thron and Hecht, 2010) and propagation was computed using the Swiss railway noise model SEMIBEL (OFE, 1990). For aircraft noise, the three major civil airports (Z urich, Geneva and Basel) and the military airport (Payerne) were considered. Noise exposure estimates were calculated via FLULA2 (Thomann and Buetikofer, 1999), based on radar data for Z urich while for Geneva and Basel exposure was calculated using traffic statistics from the Federal Office of Civil Aviation along with available acoustic footprints from the years 2000 and 1999, respectively. For the military airport, noise exposure estimates were computed based on idealised flight paths, number of flights and approximate operation times.

For each building in Switzerland, noise exposure was estimated at pre-defined fa ade points. A maximum of three fa ade points, spaced by at least 5 m, were assigned to each building fa ade by floor. We calculated the L_{den} (defined as the weighted energetic average of Leq,day (07:00–19:00), $Leq,evening$ (19:00–23:00) and $Leq,night$ (23:00–07:00) with a respective penalty of 5 and 10 dB applied to the evening and night) for each noise source. The energetic sum of the three source-specific L_{den} values was also computed to derive total transportation noise, at every fa ade point (Eze et al., 2018). We further obtained the Swiss sonBase (same calculation models as above/SiRENE) noise maps (10 × 10 m; 4 m above ground) for road traffic and railway for year 2010 from the Federal Office for the Environment. These were based on maps for the day (06:00–22:00) and night (22:00–06:00) periods, with the daytime noise also applied for the evening interval to calculate the 24 h weighted average.

To align to our previous studies (H eritier et al., 2017; H eritier et al.,

Table 1
Description of the exposure definitions.

Definition	Code	Description
Best match max (reference)	BestMax	Used the residential floor information where available (see Fig. S1). A combination of 1 = exact floor (69.1%), 2 = middle floor (unknown floor; 29.2%), and 3 = top floor (floor known but higher than what was available in the noise database; 1.7%). Loudest façade point L_{den} value.
Middle floor max	MidMax	Assumed all individuals lived on the middle floor of the building. Loudest façade point L_{den} value.
1st floor max	1stMax	Assumed all individuals lived on the 1st floor of the building (substituted with the ground floor for buildings with no 1st floor). Loudest façade point L_{den} value. Note that floor numbers in European buildings start at 0 for the ground level.
Best match min	BestMin	Same floor linkage explained for BestMax (see Fig. S1). Quietest façade point L_{den} value.
1st floor near	1stNear	Assumed all individuals lived on the 1st floor of the building. The nearest façade point to the residential geocode location was selected. A mix of loudest and quietest L_{den} values.
Grid	Grid ₁₀	Based on the 10 × 10 m noise maps. Value from the cell in which each residential address fell was assigned.
	Grid ₁₀₀	Aggregated the 10 × 10 m noise maps (calculating the mean) to a coarser spatial resolution (100 × 100, 200 × 200 and 1000 × 1000 m grids). Value from the cell in which each residential address fell was then assigned.
	Grid ₂₀₀	
	Grid ₁₀₀₀	
Buffer	Buf ₅₀	Averaged noise in “buffers” (using focalmean) with a radius 50, 100 and 500 m around each location, based on the 10 × 10 m noise maps. Value from the cell in which each residential address fell was then assigned.
	Buf ₁₀₀	
	Buf ₅₀₀	

2018b), we included aircraft noise as an adjustment in the multi-pollutant models. We did not specifically evaluate aircraft noise in the analyses described here. This is because aircraft noise is modelled at gridded receptors at a resolution of 100 × 100 m with no differentiation by residential floor. Further, most dwellings fall entirely into one grid cell thus there is no within-dwelling exposure contrast (i.e. no distinction between loud and quiet sides).

2.3. Exposure assignment definitions

Different exposure definitions were evaluated, based either on façade point estimates or those from the 10 × 10 m noise maps. These are detailed below and summarised in Table 1. Spatial analysis and exposure linkage were conducted using the geographical information system ArcGIS10.5 and R statistical software.

a) Exposure definitions using façade points

Using the available geocodes, participants in the SNC were attached to their respective dwelling unit in the SiRENE database to assign noise exposure. In our original studies (Héritier et al., 2017; Héritier et al., 2018b), the floor linkage involved first searching for exact residential floor matches between the SNC data and the noise database. If floor was not available in the SNC, exposure from the same façade point on the middle floor of the building was linked. If there was a mismatch such that SNC floor > noise database floor, exposure from the same façade point on the highest available floor in the noise database was linked. This is illustrated in Fig. S1 in the supplementary information, and is further considered the “Best” match in these analyses.

Exposure was assigned on the basis of the single façade point per dwelling with the highest (i.e. “loudest” or maximum L_{den} value) in accordance with the recent WHO Environmental Noise Guidelines for the European Region (WHO, 2018). The lowest exposure level (i.e. “quietest” or minimum L_{den} value which may be more relevant for exposure during sleep) was also assigned. For the “Best” match floor linkage, these definitions are respectively referred to as the BestMax (which is the reference in all subsequent analyses) and BestMin.

To evaluate assumptions about residential floor, exposure by floor was assigned three ways: i) using the residential floor information as described above (BestMax); and ignoring floor information where we assumed all individuals lived ii) on the middle floor of the building (MidMax), and iii) on the first floor of the building (1stMax).

To evaluate the influence of façade point selection (i.e. loudest, quietest, nearest), comparisons were made between the BestMax and BestMin mentioned above. The distance from the residential geocode and nearest façade point per dwelling was also determined. In strategic noise mapping for Europe, noise levels are estimated at a height of 4 m

(EC, 2002; Höin et al., 2009). This is approximately the height of the first floor in Swiss buildings and homes. We thus extracted all first floor façade points ($n > 16$ mil) in the SiRENE database, plus the ground floor façade points ($n > 400,000$) for buildings with no first floor. Exposure was then assigned from the nearest façade point, i.e. without distinguishing between loudest or quietest exposure by dwelling (1stNear). A priori in the SiRENE study we decided this was likely to be a poor approach for exposure assignment, as it is highly dependent on the position of the residential geocode. Here we wanted to test this hypothesis.

b) Exposure definitions using noise maps

The main exposure definition from the noise maps was derived directly from the original 10 × 10 m noise maps for Switzerland. We extracted the value from the 10 m cell in which each residential address fell (Grid₁₀). This is how exposure would typically be assigned from a strategic noise map. This exposure definition was used in the comparison of façade points vs. noise maps.

To evaluate the influence of averaging exposure at coarser spatial scales, we defined three more spatial scales to mimic a less precise exposure assessment sometimes used in epidemiological studies (Casey et al., 2017; Joost et al., 2018). First, we calculated an ecological or area-level noise estimate based on a pre-defined geographic unit, such as census block, in which all participants in the area were assigned the same noise level. The smallest common administrative unit in Switzerland is the community which was considered too large for this analysis. The 10 × 10 m noise maps were instead aggregated to a spatial resolution of 100 × 100, 200 × 200 and 1000 × 1000 m, calculating the arithmetic average of noise for each rectangular area (e.g. Grid_{100–1000}). Second, the focal statistics tool in ArcGIS was used to calculate the arithmetic average of noise in circular windows to approximate “buffers” to derive an estimate of surrounding neighbourhood noise. This was done for a radius 50, 100 and 500 m (i.e. half the diameter of the aggregated grids). We then extracted the values from each of the resulting grids (e.g. Buf_{50–500}) to obtain the neighbourhood average for each residential address. Fig. S2 illustrates and describes the GIS operations used to create the larger grids and “buffers” from the original noise maps.

2.4. Statistical analysis

Prior to epidemiological analyses, noise levels for all exposure definitions were censored at 35 dB for road noise and at 30 dB for railway and aircraft noise. This was done to account for background noise from diffuse sources in this lower range of exposures.

Analyses focused on primary cause of death from myocardial

infarction (MI, ICD10: I21–I22). It was not possible in the SNC to exclude persons with prior, non-fatal MI. Cox proportional hazards models, with age as the underlying time variable, were used to analyse the data. Right censoring was applied at the age of emigration, age of death from another cause, or the end of follow-up. Linear Hazard Ratios (HRs) were computed using multi-pollutant models (including transportation noise from road traffic, railways and aircraft) adjusted for potential confounders and NO₂ concentrations from a dispersion model (FOEN, 2013). HRs and 95% confidence intervals (CI) were expressed per interquartile range (IQR) or per 10 dB. Additional confounders included in the model were sex (female/male), neighbourhood index of socio-economic position (SEP) which included income, education, occupation and housing condition domains (Panczak et al., 2012) (low, medium, high), civil status (single, married, widowed, divorced), education level (compulsory education or less, upper secondary level education, tertiary level education, not known), mother tongue (German and Rhaeto-Romansh, French, Italian, other language) and nationality (Swiss, rest of Europe, rest of the world/unknown). To satisfy the Cox proportional hazard assumption, we used the Stratified Cox procedure to stratify the baseline hazard function for the following variables: sex, neighbourhood SEP, civil status, education level, and NO₂ exposure.

The main analysis compared HRs across all exposure definitions and spatial scales for the full population. Percent bias, based on excess risk and in relation to our reference definition BestMax, was also calculated using Eq. (1).

$$\text{Bias (\%)} = 100 \times \frac{(HR_{\text{obs}} - 1) - (HR_{\text{ref}} - 1)}{(HR_{\text{ref}} - 1)} \quad (1)$$

where: *HR* is the hazard ratio for the exposure definition of interest i.e. observed (*obs*) and the reference definition (*ref*) which was BestMax.

Two sensitivity analyses were conducted. The first focused on the influence of available floor information in the SNC in defining the best floor linkage. For this, the population was stratified into: 1 = exact floor, 2 = middle floor (unknown floor), 3 = top floor (floor known but higher than what was available in the noise database) (Fig. S1). The second sensitivity analysis explored potential differences in risk by type of area, where the population was stratified by urban, intermediate and rural areas.

Statistical analyses were conducted in Stata version 14.0.

3. Results

The study population included 4.35 million observations, with 33.24 million person-years for the period 5th December 2000 to 31st December 2008. During this period, there were 19,022 deaths from MI.

The study characteristics are presented in Table S1 for the full population and the subsets explored in the two sensitivity analyses. Following the floor linkage approach (Fig. S1), an exact match between residential floor and floor in the noise database was possible for 69.1% of the study population. For 1.7% of the cohort, the residential floor indicated exceeded the number of floors in the noise database thus top floor linkage was performed. The remaining 29.2%, which were primarily single-family dwellings, were linked on the basis of the middle floor (see Fig. S1 for calculation of middle floor). Compared to the total population, the subset living mainly in single-family dwellings had higher proportions of higher education, married, Swiss nationals, German speaking and high socio-economic position, while exposure to road traffic and railway noise as well as air pollution was lowest. The percentage of participants respectively living in urban, intermediate and rural areas were 63.8, 23.0 and 13.2%. Those living in urban areas had higher proportions of higher education, single and divorced, and high socio-economic position. As expected, exposures to transportation noise and air pollution were higher for those living in urban areas. Across the whole population, exposure to road traffic noise was most

prevalent with almost 90% exposed to L_{den} above 45 dB. Substantially fewer individuals (25%) were exposed to railway noise at the same level.

3.1. Correlation between and distributions for the different exposure definitions

Correlations (Fig. S3, Table S2) and boxplots (Fig. S4, including IQR values) comparing the different exposure definitions are shown in the supplementary information. For the comparison of floors, exposures using BestMax, MidMax and 1stMax were almost perfectly correlated ($r > 0.99$). Correlations at the loudest (BestMax) vs. quietest (BestMin) or nearest (1stNear) façade point were higher for railway noise ($r = 0.91$ or 0.94) than road traffic noise ($r = 0.65$ or 0.77). Despite the difference in model year and metric definition, correlations were moderately high ($r > 0.67$ road traffic noise) to high ($r > 0.88$ railway noise) comparing estimates from façade points and those assigned from the original 10×10 m noise maps (BestMax and BestMin vs. Grid₁₀). Correlations between exposure from Grid₁₀ and noise averaged at different scales were higher for railway noise than for road traffic noise ($r > 0.47$ road Buf_{50–500}; $r > 0.36$ road Grid_{100–1000}; $r > 0.86$ rail Buf_{50–500}; $r > 0.72$ rail Grid_{100–1000}). For each group of “buffers”/grids, as expected, the correlation with Grid₁₀ and BestMax decreased as the size of the spatial unit for averaging increased.

3.2. Influence of residential floor and selected façade point

The HRs for the reference (BestMax) for the full population were 1.05 (1.02–1.07) and 1.03 (1.01–1.05) per IQR for road traffic and railway noise, respectively (Fig. 1). For other exposures assigned based on the loudest façade points (MidMax, 1stMax), assumptions about floor of residence had little influence on the HRs for MI mortality. Bias ranged from -4.4 to 10.7% , with more bias for railway compared to road traffic noise (Table 2). Using the quietest compared to loudest façade points (BestMin vs. BestMax), however, attenuated HRs and introduced substantial bias for road traffic noise (-56.2%) but not for railway. Using the nearest façade point (1stNear), which included a mix of loudest and quietest façade points per dwelling depending on location of the geocode, also reduced the HRs for road traffic noise with a bias of -40.6% . For railway noise, the HR and bias for 1stNear were similar to that for the other exposure definitions using façade points.

3.3. Façade points vs. maps and influence of spatial averaging

As shown in Fig. 1, the HRs for the original 10×10 m noise map (Grid₁₀) were 1.03 (1.01–1.06) and 1.02 (1.00–1.04) per IQR for road traffic and railway noise, respectively. Compared to the reference, the attenuation introduced a downward bias (i.e. negative sign indicating attenuation) of -25.5 to -28.2% (Table 2). Compared to road traffic noise, HRs for railway noise were more robust to changes in spatial scale. While the Grid_{100–1000} HRs for railway noise compared to the BestMax were slightly attenuated, the changes were not as marked as for road traffic noise (maximum bias of -34.3% and -105.1% for railway and road traffic noise, respectively). The HRs for railway noise further did not exhibit a clear downward trend with increasing spatial scale, and all exposure definitions except Grid₁₀₀₀ remained statistically significant. Compared to the 10×10 m noise map (Grid₁₀), the average noise calculated in the smaller “buffers” (Buf_{50–100}) did not materially affect the HRs for either source. On the other hand, HRs for the larger 500 m “buffer” were notably changed for the two sources, especially when compared to the BestMax.

3.4. Sensitivity analysis 1: influence of residential floor

Fig. 2 illustrates the results stratified by the floor linkage used in our previous studies. The HRs are presented per IQR exposure to highlight

Full population (n=4,347,902)

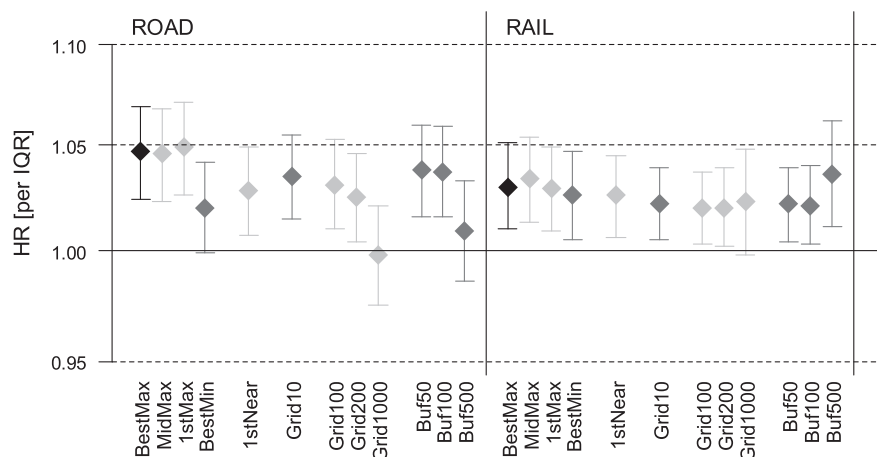


Fig. 1. Comparison of exposure definitions on the relationship between noise exposure and MI mortality. Hazard ratios (HR) and 95% confidence intervals (CI) per IQR increase in noise exposure. Adjusted for sex, neighbourhood SEP, civil status, education, mother tongue, nationality and NO₂. ROAD and RAIL are multi-pollutant models including road traffic, railway and aircraft noise. Results are for the full population. See Table 1 for exposure definitions and Fig. S4 for IQR values.

Table 2
Percentage bias in excess risk for each of the exposure definition compared to the reference.

Exposure definition	% Bias	
	Road traffic noise	Rail noise
BestMax	Reference	Reference
MidMax	-1.9	10.7
1stMax	4.4	-4.4
BestMin	-56.2	-15.0
1stNear	-40.6	-16.2
Grid ₁₀	-25.5	-28.2
Grid ₁₀₀	-33.2	-34.3
Grid ₂₀₀	-47.4	-33.0
Grid ₁₀₀₀	-105.1	-25.4
Buf ₅₀	-19.4	-28.7
Buf ₁₀₀	-20.1	-29.8
Buf ₅₀₀	-80.6	17.6

the patterns in risk between the different subsets. Similar HRs were found for road traffic noise in the multi-pollutant and single pollutant models. For road traffic noise, the subset where exact floor linkage was possible had slightly lower HRs compared to the full population, while those with middle floor linkage, mainly single family dwellings, had slightly higher HRs. The small subset for which a top floor linkage was used resulted in the highest point estimate, though with larger 95% CIs and a loss of statistical significance. Differences between subsets were not as marked for railway noise. Overall, there were no significant differences between subsets as indicated by the overlapping 95% CIs; all p-values for interaction were > 0.07 (Table S3).

3.5. Sensitivity analysis 2: influence of type of area

HRs for populations residing in urban, intermediate and rural areas for the different exposure definitions are presented in Table 3. Here the results are reported per 10 dB to facilitate comparisons with other studies and populations. Across exposure definitions, the HRs were higher and confidence intervals larger for road traffic noise in the subset residing in intermediate areas compared to urban and rural areas. The lowest HRs were found for those living in rural areas. Though less distinct, a similar pattern was apparent for railway noise for most exposure definitions. Overall, the overlapping 95% CIs and p-values for interaction indicated no significant differences between subsets (Table S4).

4. Discussion

Transportation noise risk estimates for MI mortality, especially for road traffic, were found to be sensitive to exposure definition such as differences in façade point selection (e.g. loudest vs. quietest façade estimates), exposure prediction format (e.g. façade estimates vs. noise maps), and spatial scale (e.g. in aggregated grids and “buffers”).

Compared to the reference method (BestMax), downward bias increased with increasing spatial aggregation. The effect on study results was found to be more of a problem for road traffic than for railway noise, in particular in urban settings. HRs for railway noise were more robust when comparing across exposure definitions, with a maximum bias of -34.3% (Fig. 1, Table 2). Switzerland has a very developed and dense railway network covering > 5000 km and ~1800 stops (FSO, 2016), however, railway infrastructure in general connects towns and cities and is less complex than road networks. As a consequence the geometry of railways and propagation from this source are relatively simple, typically with one set of adjacent rail lines passing through an area. Further, the traffic data for the rail noise model is more accurate than for road traffic noise (Karipidis et al., 2014) and approximately 45% of the study participants were not exposed to railway noise (Héritier et al., 2017). Taken together this indicates that with increasing spatial heterogeneity of noise distribution the effect of aggregation becomes more crucial.

Assumptions about floor of residence had little influence for both road and railway noise (Fig. 1). Noise levels do not decrease substantially with floor height if no obstacles or noise barriers are on the propagation path (which is the most often the case). Further, high-rise residential buildings are not typical in Switzerland. On the other hand, orientation of the façades to the source may play an important role (Karipidis et al., 2014), with windows at the quiet side in sonBASE having approximately 10 to 20 dB(A) less road traffic noise than the loudest side (Höin et al., 2009). One would expect that most people have their sleeping room at the less exposed noise façade and thus to observe stronger association with BestMin than BestMax. For railway noise, effect estimates were similar; but for road traffic noise, effect estimates using the full population for BestMin were substantially lower than for BestMax (Fig. 1). This may be explained by the simple noise distribution situation for railway noise mentioned above, also reflected in the high correlations between the loudest and quietest exposures and subsequent similar HRs for railway noise. The situation for road traffic noise, however, is geometrically more complex and dense (e.g. with residences surrounded by roads in built up areas). This led to lower exposure correlations and attenuation in the HRs for the BestMin compared to BestMax.

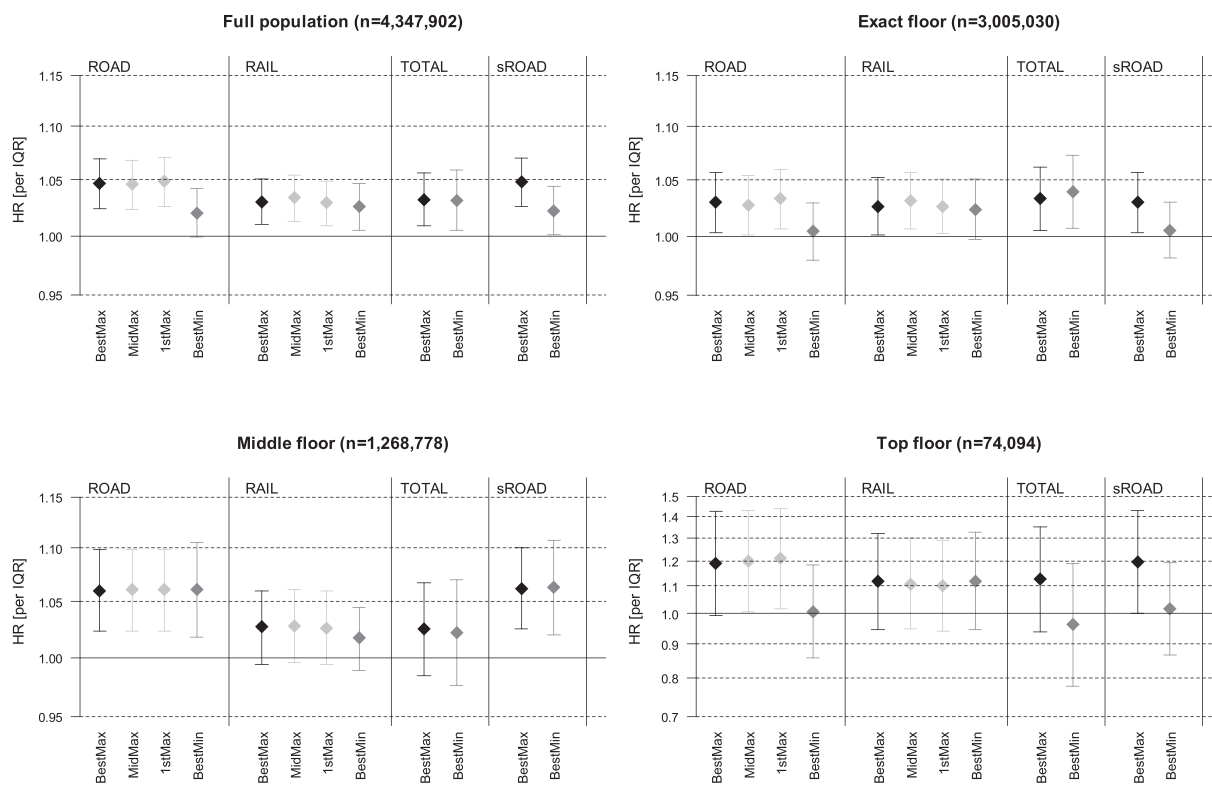


Fig. 2. Sensitivity analysis 1: influence of residential floor decisions on MI mortality. Hazard ratios (HR) and 95% confidence intervals (CI) per IQR increase in noise exposure. Adjusted for sex, neighbourhood SEP, civil status, education, mother tongue, nationality and NO₂. ROAD and RAIL are multi-pollutant models including road traffic, railway and aircraft noise; TOTAL is a single pollutant model for the combined exposure from all three transportation sources; and sROAD is a single pollutant model for road traffic noise. See Table 1 for exposure definitions and Fig. S4 for IQR values. Results are for the full population, and population stratified by the floor linkage (i.e. exact floor, middle floor, and top floor linkage; see Fig. S1).

Strikingly, the second sensitivity analysis indicated that the attenuation for BestMin compared to BestMax was in the urban population only; for those living in intermediate and rural areas, HRs for BestMin were similar or slightly higher. In intermediate settings building density is lower and the geometry less complex (mean population density in intermediate and urban areas is 245 and 1095 persons/km², respectively). Further, the road network may show the most pronounced mix between arterial and small roads and thus the most reliable traffic counts. These combined factors may result in little exposure misclassification. In urban settings, building configurations and the road network are more complex and denser. Thus, for small

distances even small uncertainties in the building footprints will produce larger errors in exposure assessment and more complex reflection patterns may not be adequately accounted for by the model. It should also be noted that noise estimates are not available at inner courtyard façades and city trams, which tend to follow roads, were not included in the version of sonRail used in the SiRENE study. This may also have contributed to increased exposure misclassification in urban areas (Karipidis et al., 2014). Masking of effects due to other noise sources (i.e. neighbour, community noise or church bells) is also likely increased in urban settings, in particular at the façades with minimum noise level, introducing additional exposure misclassification. Taken

Table 3
Sensitivity analysis 2: influence of type of area (urban, intermediate, rural) on MI mortality, HRs per 10 dB.

Exposure definition	Road				Rail			
	Full population n = 4,347,902	Urban n = 2,771,961	Intermediate n = 1,001,542	Rural n = 574,399	Full population n = 4,347,902	Urban n = 2,771,961	Intermediate n = 1,001,542	Rural n = 574,399
BestMax	1.04 (1.02–1.06)	1.04 (1.01–1.06)	1.07 (1.03–1.11)	1.01 (0.97–1.06)	1.02 (1.01–1.03)	1.02 (1.01–1.04)	1.03 (1.00–1.06)	1.01 (0.97–1.05)
MidMax	1.04 (1.02–1.06)	1.04 (1.01–1.06)	1.06 (1.02–1.11)	1.01 (0.97–1.06)	1.02 (1.01–1.04)	1.02 (1.01–1.04)	1.03 (1.00–1.06)	1.01 (0.97–1.05)
1stMax	1.04 (1.02–1.06)	1.04 (1.01–1.06)	1.06 (1.03–1.11)	1.01 (0.97–1.06)	1.02 (1.01–1.03)	1.02 (1.00–1.04)	1.03 (1.00–1.06)	1.01 (0.97–1.05)
BestMin	1.03 (1.00–1.05)	1.00 (0.97–1.04)	1.08 (1.02–1.14)	1.04 (0.97–1.11)	1.03 (1.01–1.05)	1.03 (1.00–1.05)	1.03 (0.98–1.08)	1.00 (0.93–1.08)
1stNear	1.03 (1.01–1.04)	1.01 (0.98–1.03)	1.08 (1.04–1.13)	1.00 (0.95–1.05)	1.02 (1.01–1.04)	1.03 (1.01–1.05)	1.02 (0.98–1.05)	1.01 (0.96–1.06)
Grid ₁₀	1.03 (1.01–1.05)	1.03 (1.00–1.05)	1.07 (1.03–1.12)	1.00 (0.96–1.05)	1.02 (1.00–1.04)	1.03 (1.01–1.05)	1.02 (0.99–1.06)	1.00 (0.94–1.05)
Grid ₁₀₀	1.03 (1.01–1.06)	1.02 (0.99–1.05)	1.06 (1.01–1.11)	1.03 (0.97–1.09)	1.02 (1.00–1.04)	1.03 (1.01–1.05)	1.02 (0.98–1.05)	0.98 (0.93–1.03)
Grid ₂₀₀	1.03 (1.00–1.06)	1.01 (0.97–1.04)	1.08 (1.02–1.14)	1.05 (0.99–1.12)	1.02 (1.00–1.04)	1.03 (1.00–1.05)	1.01 (0.98–1.05)	0.98 (0.93–1.04)
Grid ₁₀₀₀	1.00 (0.96–1.03)	0.97 (0.93–1.01)	1.06 (1.00–1.14)	1.01 (0.93–1.10)	1.02 (1.00–1.04)	1.02 (1.00–1.05)	1.01 (0.96–1.05)	1.00 (0.94–1.07)
Buf ₅₀	1.04 (1.02–1.07)	1.03 (1.00–1.06)	1.09 (1.04–1.14)	1.01 (0.96–1.07)	1.02 (1.00–1.04)	1.03 (1.00–1.05)	1.02 (0.98–1.06)	0.99 (0.94–1.05)
Buf ₁₀₀	1.05 (1.02–1.08)	1.03 (0.99–1.07)	1.10 (1.04–1.16)	1.03 (0.97–1.10)	1.02 (1.00–1.04)	1.03 (1.00–1.05)	1.02 (0.98–1.06)	0.99 (0.94–1.05)
Buf ₅₀₀	1.01 (0.98–1.05)	0.99 (0.94–1.04)	1.05 (0.98–1.13)	1.05 (0.96–1.15)	1.03 (1.01–1.05)	1.04 (1.01–1.06)	1.02 (0.97–1.06)	0.99 (0.93–1.06)

Hazard ratios (HR) and 95% confidence intervals (CI) per 10 dB increase in noise exposure. Adjusted for sex, neighbourhood SEP, civil status, education, mother tongue and nationality. These are multi-pollutant models including road traffic, railway and aircraft noise.

together, these considerations may explain the absence of an exposure-response association for the BestMin in urban settings and also explain the lower risk increase per 10 dB for BestMax in the urban compared to the intermediate population (sensitivity analyses 2). Studies in Austria by Lercher et al. (2011) support this notion, suggesting that the moderating effect of bedroom location may depend on the combination of sources. Exposure misclassification may also explain why no association was seen between MI mortality and BestMax of road and railway noise in rural settings. At low noise levels masking effects from other noise sources including natural sources (birds and animals) may have contributed to model uncertainty at lower noise levels (Karipidis et al., 2014).

To date strategic noise maps available for select populations, specifically residents in larger agglomerations, have become a resource for epidemiological studies (Dzhambov et al., 2017a; Dzhambov et al., 2017b; Eriksson et al., 2013; Fuks et al., 2017). We found the use of exposure at the residential geocode derived from noise maps instead of noise modelled directly at the façade resulted in 25.5 to 28.2% downward bias compared to the reference. A study from Sweden also highlights the importance of precision in geocodes when using strategic noise maps. After manually and automatically positioning location at the most exposed dwelling façade, residential address, and most exposed façade of the building, they found the highest concordance between observed and predicted annoyance when the most exposed dwelling façade was used (Eriksson et al., 2013). In our study, the greater bias value of -40.6% when using the 1stNear exposure definition, particularly for road traffic noise, echoes this point.

Our results highlight that the spatial resolution is important for detecting health effects. In short, the finer scale introduced less measurement error and subsequent bias. Several studies on noise mapping in Europe have compared different noise models and resulting exposure estimates (Garg and Maji, 2014; Morley et al., 2015; Murphy and Douglas, 2018; Nijland and Van Wee, 2005). Interestingly, in comparing models applicable to the Environmental Noise Directive or END (2002/49/EC) adopted in 2002 (EC, 2002), Murphy and Douglas (2018) found that the exposure estimation approach (e.g. using noise levels from the loudest vs. quietest receivers) had more influence on the derived L_{den} population exposure than the noise model itself. The present study, however, is the first thorough evaluation of the influence of exposure definition on epidemiological findings in a large population.

4.1. Strength and limitations

We developed an extensive and detailed noise exposure model, which allowed for individual exposure linkage at the address and floor level. This has rarely been done in previous large population studies. Analysis of our data indicates that linkage by floor may not be crucial for reducing measurement error, at least for Switzerland which has relatively few high-rise residential buildings. Floor linkage was not relevant for almost a third of our participants, i.e. those living in single family homes. For single family home with two stories (ground and first), the upper floor is considered the middle floor in our database. This is a reasonable assumption as most houses would have bedrooms upstairs. The sensitivity analyses pointed to differences in effects by floor linkage and type of area, with higher HRs in the subset living predominantly in single family homes. This is perhaps the result of a more homogenous dataset with less exposure misclassification due to simpler geometry compared to multi-level apartment buildings. This subset may also be more homogeneous in terms of confounding factors such as socio-economic status and thus yield less downward biased risk estimates.

A validation study for Switzerland, comparing 99 weekly measurements in 2016 to the 2011 SiRENE model, reported good agreement (mean + 0.5 dB(A); standard deviation 4.0 dB(A)) and temporal stability if no substantial changes in infrastructure (Schlatter et al., 2017). We could also compare noise exposure calculated at façade points with

estimates assigned at the residential geocode from noise maps developed using similar input data and models. These were not fully comparable, however, due the difference in years and in the metric definition (i.e. 2001 and L_{den} for the façade points vs. 2010 and a composite metric for the maps based on Swiss definitions for day and night). We thus cannot rule out that the noted differences between the BestMax and $Grid_{10}$ were in part due to these aspects.

Attempts at land use regression (LUR) for statistically estimating noise exposure at the homes of study participants have been made for cities/municipalities, with a growing number of examples from around the world (Aguilera et al., 2015; Harouvi et al., 2018; Ragetti et al., 2016; Sieber et al., 2017; Walker et al., 2017; Xie et al., 2011). To further compare to a noise LUR model was not practical given that our study area was the whole of Switzerland. As LUR gains popularity, studies will be needed to evaluate how they compare to source-propagation models. To run and incorporate estimates from the simplified CNOSSOS-EU model by Morley et al. (2015) was also beyond our scope, though such an endeavor may be interesting in future work.

Most studies to date, similar to ours, report risk estimates based on outdoor noise exposure when indoor noise may be more relevant for health. The few studies evaluating indoor noise or other proxy variables such as bedroom location and window opening habits point to an increased risk once these factors are taken into account. A 10-year follow-up study by Babisch et al. (1999) reported an increase of adjusted odds ratio (OR) for ischemic heart disease of 1.3 (95% CI: 0.8–2.2) when considering room orientation and window opening habits, and after duration of residence > 15 years was considered the OR increased to 1.6 (0.9–3.0). In a more recent study, Babisch et al. (2012) assessed the impact of exposure modifiers on the relationship between transportation noise and blood pressure. They reported significant effect modification for factors such as type of housing, length of residence, location of the living room, and noise barriers. In line with these results, Foraster et al. (2014) reported increased estimates for systolic blood pressure from -0.20 ($-1.25; 0.84$) for outdoor road traffic noise to 0.72 ($0.29; 1.15$) for indoor noise per 5 dB increase, after considering attenuation rates for bedroom position, types of window and opening window habits. Additionally Seidler et al. (2016) reported increased risk of heart failure/hypertensive heart disease diagnosis for indoor vs. outdoor sound pressure noise levels by traffic source. While correction factors have been studied for Switzerland (Locher et al., 2018), detailed information on noise attenuation factors such as window opening behavior, side of bedroom, or window construction/glazing is not available in the SNC. Thus, our results might be different in a study that has such kind of information available.

5. Conclusion

In a study on the whole adult population in Switzerland, the spatial scale for averaging of road traffic and railway noise exposure was shown to be important, with larger scales introducing more bias and consequent attenuated health estimates. We found the association between road traffic noise and MI mortality was more sensitive to measurement error in exposure definition than for railway noise. This indicates, in particular for numerous and dispersed sources resulting in complex exposure situations, that the use of noise maps may produce biased risk estimates and the finest resolution maps or façade modelling should be applied.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.envint.2018.12.015>.

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