The influence of long term traffic-related noise exposure on cardiovascular mortality

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Harris Héritier

Aus Molondin VD und Mauritius

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Prof. Dr. Martin Röösli und Dr. M. Sørensen

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Prof. Dr. Martin Spiess
Dekan der
Philosophisch-Naturwissenschaftlichen Fakultät

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List of abbreviations

AOD:	Aerosol optical depth
BP:	Blood pressure
CI:	Confidence interval
CVD:	Cardiovascular diseases
DALY:	Disability-adjusted life year
HR:	Hazard ratio
HRQOL:	Health related quality of life
ICD-10:	International classification of disease, 10^{th} revision
IHD:	Ischemic heart disease
IR:	Intermittency ratio
Lden:	Time averaged noise levels with 5 dB penalty between 19:00 and 23:00 and 10 dB penalty between 23:00 and 07:00
Ldn:	Time averaged noise levels with 10 dB penalty between 23:00 and 07:00
Leq:	Equivalent noise levels
MI:	Myocardial infarction
p:	p-value
$PM_{2.5}$	Particular matter smaller or equal to 2.5 µm in size
PM_{10}	Particular matter smaller or equal to 10 µm in size
SEM:	Structural equation modelling
SNC:	Swiss national cohort
Swiss TPH	Swiss tropical and public health institute, Basel, Switzerland
WHO:	World health organization
YLL:	Years of life lost

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Summary

Background

Cardiovascular diseases are the leading cause of death globally and transportation noise has been reported to be associated with cardiovascular diseases. A meta-analysis reported a pooled risk ratio for cardiovascular diseases of 1.08 (95% CI: 1.04-1.13) per 10 dB of increase of road traffic noise levels. Some relationships between noise, annoyance to noise and health need to be clarified. There are still uncertainties about the exposure-response relationship for the major transportation noise sources from road traffic, railway and aircraft. In addition, most studies have used averaged noise indicators to define exposure, not taking the variation of noise into account. Further the timing of exposure has not been investigated enough. Finally there is still no agreement about the impact of confounding by air pollution in noise studies.

Objectives

The aim of the present thesis was to better understand

1) The relationship between annoyance to road traffic noise exposure, road traffic noise exposure and heath-related quality of life,

2) To better understand the mutual independent link between cardiovascular mortality in adults and residential exposure to transportation noise sources,

3) To evaluate the impact of noise characteristics such as number of events and intermittency on the association between transportation noise and cardiovascular diseases

4) To evaluate whether the association between noise and cardiovascular mortality is modified by factors such as age, gender, socio-economic status, nationality and degree of urbanity

5) To investigate the effects of transportation noise in different time windows during day and night on cardiovascular mortality.

6) To build a nationwide NO_2 and $PM_{2.5}$ air pollution model in order to assess the mutual impact of noise and traffic related air pollution on cardiovascular mortality.

Methods

The first study presented is based on the QUALIFEX prospective cohort study on health-related quality of life and radiofrequency electromagnetic field exposure. Questionnaires were sent at baseline and follow-up one year later. The questionnaire consisted of a battery of validated scores about general health status, socio-demographic and lifestyle factors. Noise annoyance due to transportation, industry and neighbours noise was evaluated using a validated question. Road traffic noise exposure was

assigned at the place of residence of the cohort participants based on their geocodes. For each respondent, time-weighted daily average noise levels L_{dn} were calculated for road traffic noise. Baseline and follow-up survey data were combined and analysed with multivariable mixed-effects regression models to investigate the association between annoyance to each noise source, noise exposure, and the health indicators. Upon identification of sleep disturbance as the main effect modifier, a structural equation model was built to explore the interdependencies between the variables road traffic noise, annoyance to road traffic noise, sleep disturbance and HRQOL.

The second, third and fourth studies were designed as follows:

Data from the Swiss national cohort (SNC) was used. The data contain 7.28 million observations and information from census data from 2000, and mortality and emigration records for the period 2000 to 2008. The outcomes under investigation were primary causes of death from various cardiovascular outcomes. Within the framework of this project, a high definition Swiss-wide noise database for the three transportation noise sources and intermittency ratio (IR) for the year 2001 was built. The IR quantifies the number of noticeable noise events of a noise source based on the sound energy of events that surpasses background noise by 3 dB. Each SNC participant was assigned exposure from the noise database according to their geocodes and floor of residence.

In the second study, L_{den} for each noise source was assigned at the most exposed façade to each participant of the cohort. The association between noise and CVD was investigated by Cox regression. Hazard ratios for the outcomes of interests were computed using multipollutant models. Each model was adjusted for potential confounders and IR. Stratification analysis was conducted by sex, age, building age, movers, number of events exceeding the background level, socio-economic position, and degree of urbanity.

In the third study, combined noise exposure ($L_{eq,Comb}$) from all three sources was calculated (i.e. the energetic sum of road, railway, and aircraft noise) in different time windows. IR from all transportation sources combined was estimated in the same time periods as the $L_{eq,Comb}$ as well as separately for 07 to 23h and 23 to 07h. The data was analyzed by Cox regression. In order to calculate comparable hazard ratios (HR), the $L_{eq,Comb}$ for each time window of interest were standardized and HRs per one standard deviation (SD) of exposure were reported. Each model was adjusted for IR and potential confounders. The impact of IR in different time windows during day and night adjusted for $L_{den,Comb}$ and the other potential confounders mentioned above was examined.

In the fourth study L_{den} for each noise source was assigned at the most exposed façade to each participant of the cohort. In addition, NO_2 (for the years 2000 to 2008) and $PM_{2.5}$ (for the years 2003 to 2008) concentrations were predicted at the residential address. The yearly predictions were averaged in order to determine long-term exposure. The association between noise and MI was examined by Cox regression. Hazard ratios for the outcomes of interests were computed using

multipollutant models. Models were adjusted for potential confounders, without adjusting for air pollution, with adjustment for NO_2 or $PM_{2.5}$, and adjusted for both air pollutants.

Conclusion

The first study demonstrates that annoyance to road traffic noise is more strongly associated with health-related quality of life than road traffic noise itself. This pattern was confirmed by structural equation modelling that revealed an indirect link between road traffic noise and health-related quality of life via annoyance to road traffic noise. Annoyance was found to be the mediator of further health effects to which sleep disturbance belongs.

The second study confirmed previous results from international studies on transportation noise and strengthens the existing evidence. Road traffic noise appeared to be the most widespread noise source and the most detrimental for health. Road traffic noise studies conducted to date have found inconclusive results with regards to stroke. In the second study, road traffic noise was significantly associated with ischemic stroke but not with haemorrhagic stroke. Impact of railway noise exposure was comparable with road traffic noise but at a smaller magnitude. For railway noise the risk started to increase from levels as low as 30 dB. Aircraft noise was associated with heart failure and myocardial infarction. Exposure to mid-range noise intermittency tended to increases the risk of cardiovascular disease by 1%.

In the third study, different timing of exposure yielded diurnal variation of risk for cardiovascular diseases, ischemic heart diseases, blood pressure and heart failure. Stroke was neither associated with noise nor a diurnal variation of risk could be observed. For ischemic heart diseases, the risk was greater during the night whereas for heart failure daytime noise appeared to be more relevant. For blood pressure-related death and nighttime noise, the risk started to increase from levels as low as 35 dB.

Adjustment for NO_2 and $PM_{2.5}$ did not induce much change of the risk estimate for all noise sources and MI. Categorical models showed that the increase of the risk estimates for road traffic noise and MI was more pronounced in the higher noise categories 60-65dB and >60 dB. The increase was mainly driven by adjustment for NO_2. For railway noise, MI risk estimates were robust to adjustment to air pollution.

1. Introduction and background

1.1. Cardiovascular diseases (CVD): a public health burden

In 2012, CVD were the leading cause of death globally and its death toll amounted to around 17.5 million people, representing 31% of all deaths (1). Yearly the CVD subgroups coronary heart disease and stroke cause around 7.4 and 6.7 million deaths (1). Over three quarters of the deaths caused by CVD occur in low- and middle income countries (1). While a decrease in age-standardized mortality rate has been observed in high income countries for decades (2), mortality rates have been reported to be higher in low- and middle income countries than in the western world (3). Quality of life is affected too; disability-adjusted life years (DALYs) is an indicator combining the years of healthy life lost due to disability and the years of life lost due to premature death. CVD are responsible for 10% and 18% of DALYs in low- and middle income countries, and in high-income countries, respectively (1). The worldwide distribution of DALYs for CVD is depicted in Figure 1. In 2010, the global costs of CVDs were estimated at US$ 863 billion (4).

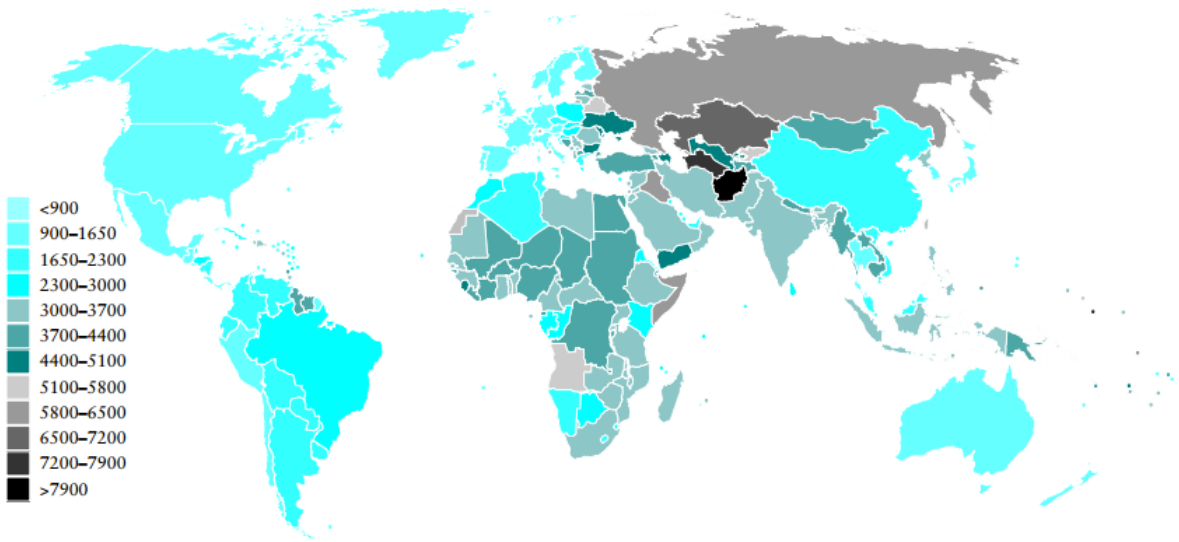

Figure 1: Map of the worldwide distribution of DALYs for CVD in 2004 per 100'000 (5)

In 2014, CVD was the number one killer in Switzerland and accounted for 34.8% and 30.6% of the total deaths for females and males (6). In 2011, CVD is attributed to CHF 10.3 mio and CHF 6.4 mio of direct and indirect costs (7).

Risk factors for CVD can be classified in three different subgroups; physiological, behavioral, and socioeconomic and environmental (8). Physiological factors include hypertension, high blood lipids, diabetes and high blood glucose, and thrombosis and inflammation. These factors act directly on the

cardiovascular system by favoring atherosclerosis leading to CVD. The status of physiological factors is directly influenced by behavioral factors from one side, and by socioeconomic and environmental factors from the other. Socioeconomic and environmental factors also indirectly influence physiological pathways via behavioral factors. Behavioral factors comprise physical inactivity, smoking, alcohol overconsumption, and unhealthy diet, while socioeconomic and environmental factors encompass exposure to air pollution and transportation noise, lack of green space, socioeconomic status, education level and inequalities. This framework is displayed in Figure 2.

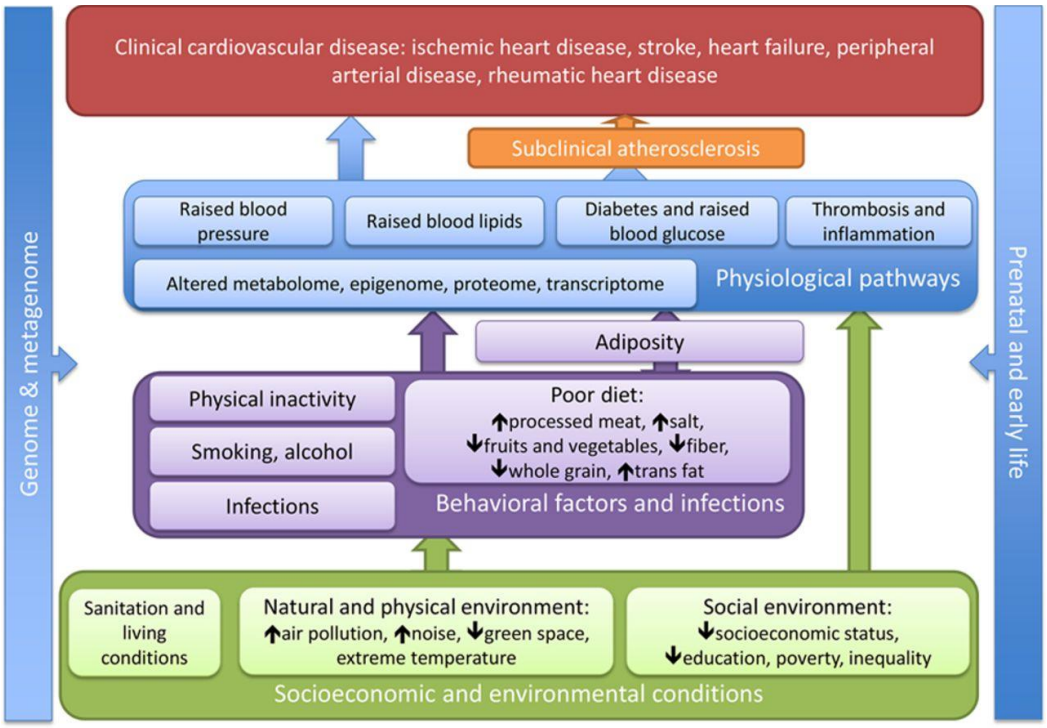

Figure 2: Schematic diagram of proposed determinants of and risk factors for CVD by Tzoulaki et al. (8)

A joint-effect analysis of those risk factors conducted in a comparative risk assessment study by Ezzati, Hoorn (9) showed that an estimated 80% of deaths from IHD and 70% of deaths from stroke globally were attributable to only a few physiological and behavioral factors. Most environmental factors contribute little to the etiology of CVD compared to physiological and behavioral factors. However, exposure to air pollution and transportation noise is very common and even a marginal risk increase can have a significant impact in terms of public health.

1.2. Noise

Sound is a wave travelling through the air by small fluctuations of atmospheric pressure. A sound is made of different tones, each of these tones having a particular frequency. The human ear perceives frequencies ranging from 20Hz to 20’000Hz. When these frequencies are harmonically ordered we hear pleasant sounds such as the chirping of birds or music. When these frequencies are chaotically

ordered, we perceive unpleasant sound referred to as noise. However, the way each of us perceives noise depends on our individual habits and preferences, situational conditions, and meaning and impact of noise.

Noise is measured in sound pressure level *Lp*, which is a logarithmic measure of the effective pressure of a sound *p* relative to a reference value p_0 (equation 1).

Equation 1 $$L_p = 10 \log_{10} \left(\frac{p^2}{p_0^2}\right) \text{[dB]}$$

The sound pressure levels tells how much larger the measured sound pressure is compared to the pressure at the hearing threshold ($2*10^{-5}$ Pa @ 1000Hz) and it is measured in decibel (dB). An overview of the decibel scale is depicted in Figure 3.

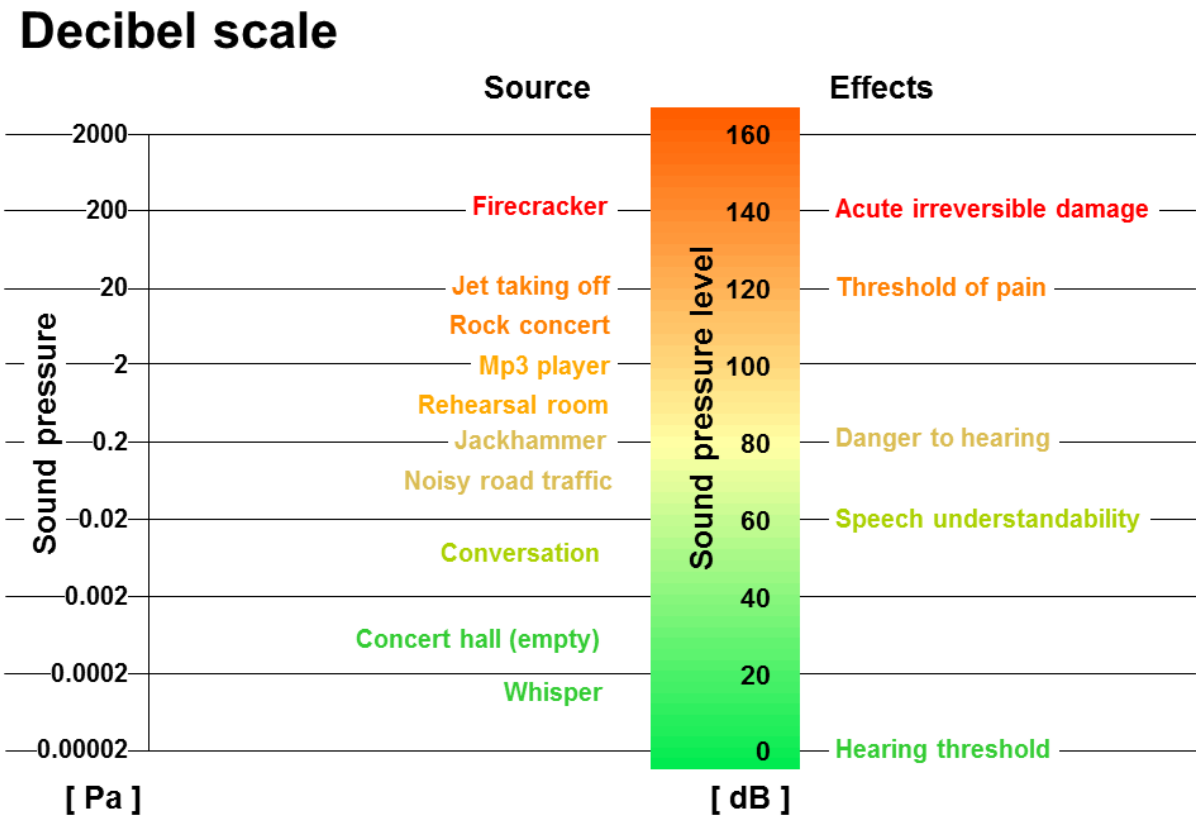

Figure 3: The decibel scale

Sound level meter devices can measure the sound pressure level in real time very accurately. However, in order to evaluate long-term noise exposure, an energetic average of the sound pressure level over the timeframe of interest coined the equivalent continuous sound level L_{eq} is preferred (10). Two further noise metrics often used in epidemiology are based on the L_{eq}; L_{day}, L_{night}, L_{dn} and L_{den}. L_{day} and L_{night} metrics are the average equivalent sound level over the daytime and nighttime hours. The definition of daytime and nighttime hours can vary depending on the country. The L_{dn} is the average equivalent sound level over 24h including a penalty of 10 dB for noise during the nighttime hours

from 22:00 to 06:00 in Europe (10) and 23:00 to 07:00 in Switzerland. The L_{den} is the average sound level over 24h including a penalty of 5 dB for noise during the evening hours from 18:00 to 22:00 and a penalty of 10 dB during the nighttime hours from 22:00 to 06:00 (10). In Switzerland, the L_{den} is calculated based on evening hours from 19:00 to 23:00 and nighttime hours from 23:00 to 07:00.

1.3. Noise exposure

Currently, more than half of the world's population live in urban areas where noise levels exceed the WHO guideline for night-time noise of 45 dB (11). According to projections from the WHO, by 2050 seven out of ten people will live in cities (12). A growing world population and increasing global traffic will lead to more noise and therefore expose a bigger share of the population to harmful noise levels.The most prevalent noise globally originates from transportation noise sources: road traffic, railway and aircraft noise. Studies from Vietnam (13), Brazil (14), India (15) and Nigeria (16) have reported noise levels far above the 45 dB threshold. In China, in order to manage the 10% annual growth in air traffic, six to seven new airports are built each year in addition to the 180 that already exist (17). A noise impact study around Guangzhou Baiyun airport in China has reported weighted effective continuous perceived noise levels L_{WECPN} above 70 dB affecting 400'000 people (18).

The situation is similar in industrialized countries. In 2012 in the European Union, 125 and 37 mio people were respectively affected by road traffic noise L_{den} above 55 and 65 dB (19). For railway and aircraft noise 8 mio and approximately 3 mio people are exposed to L_{den} noise levels above 55 dB (19). In Switzerland, 50% of the population is exposed to $L_{eq,day}$ above 55 dB from road traffic noise (20). Further, 85, 90 and 95% of people affected by road traffic, railway and aircraft noise live in cities, where the co-exposure to two or more noise sources is common (20).

1.4. Noise and health

Unlike other environmental pollutants, noise does not physically interact with the organs of the exposed individuals. Rather it interacts with the body via nervous stimulation of the brain following direct and indirect pathways (21). The direct pathway involves the acoustic nerve and the central nervous system, while the indirect pathway involves the cognitive perception of the noise and triggers cortical activation that is linked with emotional responses. Both pathways trigger stress responses translating into an increase in heart rate and increase in the concentration of cortisol, adrenaline and noradrenaline that in turn may lead to inflammation. It is important to note that noise exposure during daytime triggers the above mentioned mechanism, while nighttime noise induces additional stress via sleep disturbances.

1.3.1. Noise and annoyance

Noise as "unwanted sound" (22) is mostly perceived as a stressor and triggers annoyance which may lead to further health effects . Annoyance by transportation noise is widespread and, in Switzerland,

about two-thirds of the population is annoyed by transportation noise (23). Noise annoyance defines the level of disturbance induced by noise exposure (24) and is driven by psychosocial factors. This includes fear of danger from the noise source, beliefs about the value of the noise source, decisional freedom regarding exposure and general noise sensitivity (25, 26). The proportion of strongly annoyed individuals in a population tends to increase when the same individuals are exposed to higher noise levels (27). According to the framework of Recio et al. (28), constant or repeated exposure to noise induces sustained psychological stress that is transferred to the somatic and physiological levels. Therefore, annoyance can be seen as the first health effect of noise exposure and the mediator of further health effects such as stress (29).

1.3.2. Noise and sleep

Many studies have shown that exposure to noise causally disturbs sleep (30-32). Sleep deprived subjects tend to display alterations of the sympathetic system, that in turn trigger changes in cardiac contractility, cardiac output, blood volume and peripheral resistance, thus inducing an elevation of the blood pressure (33). Further, sleep indebted subjects present higher concentrations of inflammatory markers such as high leukocytes count and IL-6 and TNF-α concentration in blood (33). Thus, through mechanisms such as raised blood pressure and inflammation, sleep deprivation has a direct impact on CVD health. Indeed, a meta-analysis reported that short sleep duration (less than 7h per night) was associated with an increased risk of developing or dying of coronary heart diseases (RR 1.48, 95% CI: 1.22-1.80) and stroke (RR 1.15, 95% CI: 1.00-1.31) (34).

1.3.3. Noise and CVD

The mechanisms presented in the previous sections 1.3.1. and 1.3.2. lead to disturbances of the homeostasis of the exposed individuals. In the long-term, those disturbances lead to hypertension or atherosclerosis and more severe events such as stroke or myocardial infarction. Indeed, two meta-analyses found a relative risk for hypertension of 1.13 (95% CI: 1.00-1.28) per 10 dB(A) increase in aircraft noise (35), while odds ratio of 1.03 (95% CI: 1.01-1.06) was found for 5 dB(A) increase of road noise (36). The hazard ratio for mortality from myocardial infarction was found to be 1.48 (95% CI: 1.01–2.18) for subjects living since more than 15 years in an area where aircraft noise levels were above 60 dB(A) (37). Another study reported a relative risk of 1.21 (95% CI: 0.98-1.49) and 1.23 (95% CI: 1.02-1.49) for stroke mortality in relation to daytime and nighttime noise levels (38).

2. State of research and open questions

Most studies use models with a single-source noise exposure and some uncertainty still remains in the mutual independent exposure-response curve when considering multiple sources. Further, the majority of studies use the time-averaged noise exposure variables L_{den}, L_{dn} or $L_{eq,Day}$ or $L_{eq,Night}$ which do not inform neither about the peaking characteristics of the noise sources nor about the number of noisy events. Therefore there is a need for an exposure variable describing the variability of noise. Indeed, it

has been shown that the probability of event-related, body movements and cardiovascular arousals depends on the sound pressure level and the slope of rise of individual noise events (39-41). Further, the impact of the time of exposure has rarely been investigated and only a few studies report different estimates for the day and the night (42-44). Finally, there is still a lack of agreement on the question whether noise and air pollution trigger independent effects. Many studies have assessed the mutual independent impact of transportation noise and air pollution on CVD with mixed results (45-50).

3. Methods

This thesis is part of the interdisciplinary SiRENE project (Short and Long Term Effects of Transportation Noise Exposure) involving a team of acousticians, psychologists, chronobiologists and epidemiologists. The goal of SiRENE was to investigate acute, short- and long-term effects of road, railway and aircraft noise exposure on annoyance, sleep disturbances and cardio-metabolic risk. The project was financed by the Swiss National Science Foundation and conducted together with the Swiss Federal Laboratories for Materials Science and Technology (EMPA), the Centre of Chronobiology of the University of Basel, N-sphere and the Federal Office for the Environment (FOEN). The noise data generated in the framework of the SiRENE project is described in the publication of Karipidas et al. in Annex A, while the derivation of the intermittency ratio is described in the publication of Wunderli et al. in Annex B.

3.1. Aims of this thesis

In this section, the different aims of this thesis and the description of the studies are presented. Each aim refers to a particular chapter.

Aim 1: To investigate the association between road traffic noise exposure and annoyance, and health indicators.

Aim 2: To better understand the mutual independent link between cardiovascular mortality in adults and the residential exposure to transportation noise sources

Aim 3: To evaluate how relevant are noise characteristics such as number of events and intermittency of the noise source

Aim 4: To evaluate whether the association between noise and cardiovascular mortality is modified by factors such as age, gender, socio-economic status, nationality and degree of urbanity.

Aim 5: To investigate the effects of transportation noise in different time windows during day and night on cardiovascular mortality.

Aim 6: To build a nationwide NO_2 and $PM_{2.5}$ air pollution model in order to assess the mutual impact of noise and traffic related air pollution on cardiovascular mortality.

3.2. Study description

The first study (described in detail in section 4. of this work) answers aim 1 and is based on the QUALIFEX prospective cohort study on HRQOL and radiofrequency electromagnetic field exposure. Questionnaires were sent to 4000 randomly selected residents from the region of Basel, Switzerland,

aged between 30 and 60 years. After one year, a follow-up was conducted by resending the same questionnaire to the respondents of the baseline survey. The questionnaire consisted of a battery of validated scores about their general health status (general health indicator from the European health survey (51)), physical health (von Zerssen (52)), mental health (SF-36 mental health component (53)), sleep disturbances (sleep disturbance score from the Swiss Health Survey 2007 (54)), socio-demographic and lifestyle factors. Noise annoyance due to road traffic, trains, aircrafts, industry and neighbours was evaluated using a validated question. Road traffic noise exposure was assigned at the place of residence of the cohort participants based on their geocodes. For each respondent, time-weighted daily average noise levels L_{dn} were calculated for road traffic noise. Baseline and follow-up survey data were combined and analysed with multivariable mixed-effects regression models to investigate the association between annoyance to each noise source, noise exposure, and the health indicators. Interaction tests were conducted to identify effect modifiers. Upon identification of sleep disturbance as the main effect modifier, a structural equation model was built to explore the interdependencies between the variables road traffic noise, annoyance to road traffic noise, sleep disturbance and HRQOL.

Answers to aims 2 to 6 were provided based on the same health data with a different set of exposure variables for each study.

The Swiss National Cohort (SNC) contains 7.28 million observations and information from census data from 2000, and mortality and emigration records for the period 2000 to 2008. The outcomes under investigation were primary causes of death from various cardiovascular outcomes. Within the framework of the SiRENE project, a high definition Swiss-wide noise database for the three transportation noise sources and intermittency ratio for the year 2001 was built. The IR quantifies the number of noticeable noise events of a noise source based on the sound energy of events that surpasses background noise by 3 dB. Each SNC participant was assigned exposure from the noise database according to their geocodes and floor of residence.

In the second study (described in details in section 5. of this work) aims 2-4 were answered. L_{den} for each noise source was assigned at the most exposed façade to each participant of the cohort. The association between noise and CVD was investigated using the Cox proportional hazards model with age as the underlying time variable. Hazard ratios for the outcomes of interests were computed using multipollutant models, i.e. including linear terms for each noise source. Each model was adjusted for potential confounders and IR. Stratification analysis was conducted by sex, age, building age, movers, number of events that exceed the background level, socio-economic position, and urbanization.

The study on diurnal variation of noise exposure (described in section 6. of this work) answered aim 5. For this analyses combined noise exposure ($L_{eq,Comb}$) from all three sources was calculated (i.e. the energetic sum of road, railway, and aircraft noise) for the following time windows: 07 to 19h, 19 to

23h, 23 to 01h, 01 to 05h, 05 to 06h, and 06 to 07h. IR from all transportation sources combined was estimated in the same time periods as the $L_{eq,Comb}$ as well as separately for 07 to 23h and 23 to 07h. The data was analyzed by Cox regression with age as the underlying time variable. In order to calculate comparable hazard ratios (HR), the $L_{eq,Comb}$ for each time window of interest were standardized and HRs per one standard deviation (SD) of exposure were reported. Each model was adjusted for IR and potential confounders. In order to explore the exposure-response relationship, categorical analyses were conducted for all time windows. Further, the impact of IR in different time windows during day and night adjusted for $L_{den,Comb}$ and the other adjustment variables mentioned above was examined.

The study focusing on the confounding effect of air pollution (described in section 8. of this work) answered aim 6. For this analysis two distinct air pollution models for NO_2 and $PM_{2.5}$ were built. More details on the modelling procedure for $PM_{2.5}$ and NO_2 can be found in section 7. and section 8. of this work. The NO_2 model was based on passive sampler data collected from 2000 to 2008 from the air pollution monitoring authorities from the cantons: Genève, Vaud, Neuchâtel, Jura, Fribourg, Berne, Basel-City, Basel-Country, Solothurn, Aargau, Ticino, the Inluft and Ostluft network, and the Sapaldia team. Building footprint area, population density, land use area, total length of roads, traffic density, altitude, and PolluMap estimates (55) were extracted in various buffer sizes around the sampling point. Model selection was conducted for each year from 2000 to 2008 with the elastic net algorithm and the residuals were kriged. The prediction for the SNC at residential address was based on the selected elastic net model and the smoothed residuals obtained by kriging. Finally, the yearly final predictions were then averaged to obtain a long-term NO_2 exposure value.

The $PM_{2.5}$ model was built based on the method published by Kloog et al. (56). In brief, the ratio between $PM_{2.5}$ and PM_{10} data from 10 monitoring sites of the NABEL network was used to predict $PM_{2.5}$ concentrations at sites where only PM_{10} measurements were available. Aerosol optical depth (AOD) data for the period of 2003−2008 at 1km resolution was used. Spatial and temporal predictor data was extracted at different scales; the global (1x1km) and local (100x100m; nested in the 1x1km grid) scales as well as at the point location of the monitoring sites. Both global and local predictors included the following: $PM_{2.5}$ emissions from agriculture, households, industry, traffic and wood smoke for the years 2005 and 2010, distance to nearest main road, elevation, land use, meteorological data including daily modelled planetary boundary layer data, daily temperature, wind speed, wind direction and precipitation at a ~10x10km resolution. A 4-staged modelling approach, described in previous work (56) was adapted and further developed to ultimately estimate $PM_{2.5}$ concentrations at both 1km and 100m grid cells across Switzerland from 2003 to 2008. In stage 1, $PM_{2.5}$ measurements were regressed against 1km predictors using mixed effects models. In stage 2, the models were then used to predict $PM_{2.5}$ in the cells where AOD was available. In stage 3 a generalized additive mixed model with spatial smoothing was applied to generate $PM_{2.5}$ predictions for those grid cells where

AOD was missing (stage 3). Finally, to estimate 100 m localized $PM_{2.5}$ predictions, the residuals from the stage 1 model at each monitoring site were regressed against the local spatial and temporal variables at each monitoring site (stage 4) using the support vector machine algorithm.

L_{den} for each noise source was assigned at the most exposed façade to each participant of the cohort. In addition, NO_2 (for the years 2000 to 2008) and $PM_{2.5}$ (for the years 2003 to 2008) concentrations were predicted at the residential address. The yearly predictions were averaged in order to determine long-term exposure. The association between noise and MI was examined by Cox regression with age as the underlying time variable. Hazard ratios for the outcomes of interests were computed using multipollutant models, i.e. including linear terms for each noise source 1) without adjusting for air pollutants 2) adjusted for $PM_{2.5}$ 3) adjusted for NO_2 and 4) adjusted for both. Each model was adjusted for potential confounders and IR. Categorical noise analyses were performed using L_{den}(Road), L_{den}(Rail) and L_{den}(Air) in 5dB categories to explore the effect of combined exposure of noise with NO_2 and $PM_{2.5}$ respectively. Synergistic effects were searched by including an interaction term for each noise source and NO_2 and $PM_{2.5}$.

3.3. The relevance of this thesis

By examining the relationships between noise, annoyance, sleep disturbance and HRQOL, the effects of noise events, the impact of diurnal variability of noise and the extent of confounding by air pollution, many open questions in the area of epidemiological noise research have been explored. This thesis will help the community of noise researchers to better understand the impacts of noise on CVD. In addition, the introduction of a novel noise metric termed intermittency ratio opens new research avenues in environmental noise epidemiology.

The expected findings of this thesis should inform governments and ministries regarding the extent of the risk of noise exposure for cardiovascular diseases. These results will help decision-makers in determining noise protection policies for better protection of the affected populations. In addition, this thesis may offer valuable information for urban planners and architects willing to protect the populations at risk. Further, physicians may also benefit from the knowledge generated by this thesis. A better understanding of the risk of transportation noise exposure and knowledge about the most vulnerable populations may help to develop targeted prevention measures.

4. Paper 1

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The Association between Road Traffic Noise Exposure, Annoyance and Health-Related Quality of Life (HRQOL)

Harris Héritier [1,2], Danielle Vienneau [1,2], Patrizia Frei [3], Ikenna C. Eze [1,2], Mark Brink [4],

Nicole Probst-Hensch [1,2] and Martin Röösli [1,2]

[1] Swiss Tropical and Public Health Institute, Basel, Switzerland;

[2] University of Basel, Petersplatz 1, Basel, Switzerland

[3] Krebsliga Schweiz, Bern, Switzerland;

[4] Federal Office for the Environment, Bern, Switzerland;

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Article

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Harris Héritier [1,2], Danielle Vienneau [1,2], Patrizia Frei [3], Ikenna C. Eze [1,2], Mark Brink [4], Nicole Probst-Hensch [1,2] and Martin Röösli [1,2,*]

1 Swiss Tropical and Public Health Institute, Socinstr. 57, P.O. Box, CH-4002 Basel, Switzerland; E-Mails: harris.heritier@unibas.ch (H.H.); danielle.vienneau@unibas.ch (D.V.); ikenna.eze@unibas.ch (I.C.E.); nicole.probst@unibas.ch (N.P.-H.)

2 University of Basel, Petersplatz 1, CH-4003 Basel, Switzerland

3 Krebsliga Schweiz, 3001 Bern, Switzerland; E-Mail: patrizia.frei@krebsliga.ch

4 Federal Office for the Environment, 3003 Bern, Switzerland; E-Mail: mark.brink@bafu.admin.ch

* Author to whom correspondence should be addressed; E-Mail: martin.roosli@unibas.ch; Tel.: +41-(0)-61-284-8383; Fax: +41-61-284-8105.

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Abstract: The aim of this study is to investigate the relationships between road traffic noise exposure, annoyance caused by different noise sources and validated health indicators in a cohort of 1375 adults from the region of Basel, Switzerland. Road traffic noise exposure for each study participant was determined using modelling, and annoyance from various noise sources was inquired by means of a four-point Likert scale. Regression parameters from multivariable regression models for the von Zerssen score of somatic symptoms (point symptom score increase per annoyance category) showed strongest associations with annoyance from industry noise (2.36, 95% CI: 1.54, 3.17), neighbour noise (1.62, 95% CI: 1.17, 2.06) and road traffic noise (1.53, 95% CI: 1.09, 1.96). Increase in modelled noise exposure by 10 dB(A) resulted in a von Zerssen symptom score increase of 0.47 (95% CI: −0.01, 0.95) units. Subsequent structural equation modelling revealed that the association between physical noise exposure and health-related quality of life (HRQOL) is strongly mediated by annoyance and sleep disturbance. This study elucidates

the complex interplay of different factors for the association between physical noise exposure and HRQOL.

Keywords: noise; exposure; annoyance; health indicators; von Zerssen; SF-36; quality of life

1. Introduction

Annoyance is one of the numerous health effects related to noise exposure and affects a large share of the population worldwide. Annoyance, often also triggered at low noise levels, has been the focus of previous environmental noise research [1,2]. Numerous studies found a positive exposure-response relationship for annoyance with increasing noise exposure from various sources [3–5]. In 2011, the WHO estimated that the share of the European population highly annoyed by road traffic noise at levels >55 dB(A) was 25% [6]. Upon extrapolation, it was estimated that annoyance induces losses in the range of 0.32–3.92 million disability adjusted life years or DALYs/year in the European Union [6].

In recent years, the evidence linking noise exposure and indicators of annoyance-mediated degradation of quality of life has accumulated. Studies have shown marked associations between noise exposure and annoyance with disturbance [2,7,8], reduced wellbeing [2,7] and reduced health-related quality of life (HRQOL) [2,8,9].

According to the Burden of Disease Report of the WHO [6], people annoyed by noise may experience a range of negative responses such as depression, anxiety or exhaustion, thus augmenting stress which is a recognised risk factor for cardiovascular diseases. For this reason, a better understanding of annoyance and its influence on health may help to prevent future health degradation. As stated in the theoretical framework of Stallen [10] and Soames [11], annoyance plays a role in mediating the further development of noise-induced health effects. Indeed, an internal mechanism of appraisal based on a set of non-acoustical factors such as attitude towards the noise source [10] or noise sensitivity [11] modify the annoyance reaction. Thus, subjects lacking the internal resource to overcome noise-induced stress and annoyance are more likely to present signs of health degradation in the long-term, although noise effects on sleep have also been observed in people who are not annoyed by noise [12]. In previous work [13–15] structural equation models have been used to disentangle the complex interplay between noise and noise-related variables such as annoyance, sleep disturbance, noise sensitivity and HRQOL.

Further, the association between annoyance and any health outcome may be modified by factors such as sleep deprivation or body mass index (BMI). Indeed, the recent study of Sørensen *et al.* [16] indicate that BMI may play a role in noise induced health effects. A recent analysis using the same data as the present paper found that the association between road traffic noise and sleep was modified by gender [17].

The present study investigated the association between road traffic noise exposure and annoyance, and health indicators. It is based on a cohort study on HRQOL in relation to environmental factors conducted in the Basel area in Switzerland [18]. Whereas a previous analysis focussed on noise induced sleep effects [12], the present paper addresses the interplay between noise, annoyance to noise, sleep disturbance and HRQOL, and explores potential modifying factors such as socio-demographic

factors, BMI, comorbidity and noise exposure level. We further investigate the importance of annoyance and sleep disturbance as mediators of the association between physical noise and HRQOL indicators by structural equation modelling (SEM).

2. Methods

We used data from the QUALIFEX study (HRQOL and radio frequency electromagnetic field (RF-EMF) exposure: prospective cohort study), which focussed on health effects of RF-EMF and various other environmental exposures [18,19]. In May 2008, questionnaires entitled "environment and health" were sent to 4000 randomly selected residents from the region of Basel (2000 each from the cantons of Basel-City and Basel-Country), Switzerland, aged between 30 and 60 years. Reasons of non-eligibility in the cohort were severe disabilities, death, incorrect addresses (no possible matching with modelled noise exposure), absence during the time of the survey, and problems understanding the questionnaire due to language. After one year, a follow-up was conducted by sending the same questionnaire to the respondents of the baseline survey. Ethical approval for the conduct of the study was received from the ethics committee of Basel on 19 March 2007 (EK: 38/07).

2.1. Outcome Variables

The questionnaire consisted of a battery of validated scores about health in general, major health outcomes (current treatment for diabetes, stroke), and various non-specific symptoms of health (sleep quality, headaches) as well as socio-demographic (sex, age, marital status) and lifestyle (alcohol consumption, smoking, physical activity) factors. Respondents were requested to assess their health status on a categorical scale which was transformed into a binary variable (0 = "very good" and "good"; 1 = "fair", "bad" and "very bad") and used as an indicator of general health status as described in the methodological manual of the European Health Interview Survey [20]. We additionally used the von Zerssen 24 item list of somatic complaints [21] such as tiredness, loss of appetite, abdominal pain, cold feet; these are not specific to any diseases and can therefore be used for broad patients groups or, as in this study, for a population to estimate HRQOL. For each participant, answers to all 4-point Likert scale questions have been summed resulting in a continuous score ranging from 0 (no health complaints) to 96 (maximum health complaints). Mental health was assessed using the mental health section of the SF-36 questionnaire [22], which is an indicator used for evaluating individual patients health status. We recalculated the norm-based score for each participant, where high values reflected low mental health. Respondents had to state their feeling of nervousness, depression, relaxation, demoralisation and happiness on a five point scale. Sleep disturbances were assessed using the sleep disturbance score from the Swiss Health Survey 2007 [23] which addresses difficulties to fall asleep, troubled sleep, frequency of spontaneous awakening, and waking up too early in the morning.

2.2. Noise Annoyance and Noise Exposure

Noise annoyance at home due to road traffic, trains, aircrafts, industry and neighbours was evaluated using a four-points Likert scale with categories "no", "slight", "considerable", and "heavy" [24].

Noise exposure assessment was conducted using the same procedure described elsewhere [12]. In brief, the Swiss Federal Statistical Office provided geocodes for each respondent address. Both geocodes were provided for participants who moved between the baseline and follow-up (n = 65). Based on their geocodes, noise exposure was assigned from one of two available models depending on whether study participants resided in Basel-City (urban) or in Basel-Country (suburban). In Basel-City we used a road traffic noise cadaster provided by the Basel-City Office for the Environment and Energy. It is based on a detailed 3D city model that was developed by the land surveying office using photogrammetrically analysed aerial photographs. The road traffic data were derived from a traffic model from the year 2008 [12]. In Basel-Country, values were derived from the nationwide SonBASE model [25,26]. Respondents were assigned average traffic noise values for the day (L_{day} 06:00–22:00) and the night (L_{night} 22:00–6:00). Time-weighted daily average noise levels L_{dn} were calculated for rail and road traffic noise including a 10 dB(A) penalty for the nighttime [27]. Values were censored at 30 db(A), and 10 dB(A) increments of L_{dn} were used in the analysis. In order to rule out selection bias, exposure values extracted for the geocodes of participants and non-participants were compared.

2.3. Statistical Analysis

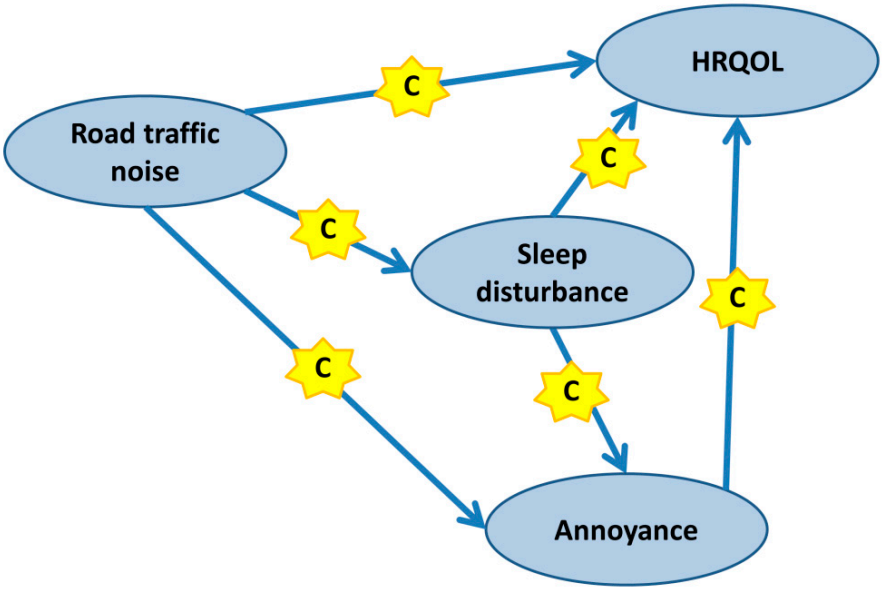
Baseline and follow-up survey data were combined and analysed with multivariable mixed-effects regression models with random intercept, clustered at the level of the individual to investigate the association between annoyance to each noise source, noise exposure, and the health indicators. The relationships with the von Zerssen symptom score and the SF-36 mental health score were analysed using linear regression, while logistic regression was used for self-reported health status. All models were adjusted for age, age as quadratic polynomial, sex, physical activity (frequency of exercise-induced sweating per week), smoking (current smoker *vs*. non or former smoker), education level (low, middle, high), and marital status (single, married, divorced/widowed). A further adjustment was conducted to account for urban *vs*. suburban region, where the two different noise models (3D city model *vs*. SonBASE) have been used.

In order to evaluate potential effect modification, stratified analyses and interaction tests with annoyance to noise source or noise exposure were conducted by sex, age (subjects aged below and above median = 47 years), noise exposure level (subjects exposed below and above median = 46 dB(A)), BMI (cut-off value = 25), and sleep disturbance score from the Swiss Health Survey 2007 [23] (subjects below and above median = 5.61, where individuals scoring higher than median had the most sleep disturbances). A further stratification was conducted for self-reported doctor-diagnosed comorbidity, defined as suffering two or more diseases (arthritis, bronchitis, myocardial infarction, stroke, kidney disease, cancer, osteoporosis or diabetes).

2.4. Structural Equation Model (SEM)

Upon identification of sleep disturbance as the main effect modifier, a structural equation model was built to explore the interdependencies between the variables road traffic noise, annoyance to road traffic noise, sleep disturbance and HRQOL. SEM allows for gathering in-depth knowledge on the direct and indirect effects variables may have on each other. As displayed in Figure 1, we specified the SEM in sequential steps based on the literature focussing on the relationships (1) road traffic noise → HRQOL, (2) road traffic noise → sleep disturbance, (3) road traffic noise → annoyance to road traffic noise, (4) sleep disturbance → HRQOL, (5) sleep disturbance → annoyance to road traffic noise and (6) annoyance to road traffic noise → HRQOL. We then built two distinct SEMs for each HRQOL indicator (von Zerssen and SF-36 score) by incrementally increasing their complexity. Relationships (1), (2), (4) and (6) were adjusted for gender, age, physical activity, smoking and education, while relationships (3) and (5) were adjusted for gender, age, urban/suburban and awareness about environmental issues (e.g., fear from car exhaust, sceptical to new technologies) [28]. All variables were z-normalised to obtain comparable regression coefficients. We ran a separate model for baseline and follow-up data. Missing values were excluded yielding 1307/1357 baseline and 1064/1074 follow up observations for SEMs including the von Zerssen/SF-36 mental health indicator. In subsequent steps, non-significant exogenous/endogenous and endogenous/endogenous relationships between variables were constrained to zero. Search for missing paths was conducted using modification indices, and significant paths consistent with the direction of effect were added to the model. Model selection was based on χ^2, Aikaike Information Criterion (AIC), Tucker-Lewis, Root Mean Squared Error of Approximation (RMSEA) and Standardized Root Mean squared Residuals (SRMR) values. Statistical analyses were carried out using STATA version 13.0 (StataCorp, College Station, TX, USA).

Figure 1. Theoretical model used for the construction of subsequent SEMs for the relationships between road traffic noise, sleep disturbance, annoyance to road traffic noise and HRQOL. The "C" indicates additional factors (confounders) relevant for an association.



3. Results

Out of 3743 eligible study participants, 1375 individuals participated in the baseline investigation (participation rate of 37%) and, of these, 1122 (82%) returned a follow-up questionnaire one year later accounting for a total of 2497 observations. The socio-demographic characteristics of the study sample are displayed in Table 1.

Table 1. Socio-demographic characteristics of the 2497 observations.

Age Categories	**In %**
30–34 Years	13.3
35–39 Years	13.5
40–44 Years	17.7
45–49 Years	17.7
50–54 Years	18.0
>55 Years	19.9
Sex	**In %**
Female	59.1
Male	40.9
Educational level	**In %**
Low (primary school)	5.9
Medium (apprenticeship)	48.4
High (higher education)	45.7
Lifestyle characteristics	
Mean BMI (SD)	24.2 (4.2)
Smokers (%)	27.3
Comorbidity * (%)	11.5

Note: * At least two chronic diseases in the same subject (see text).

In terms of potential selection bias, road traffic and rail noise exposure was not significantly different between participants (mean L_{dn} road: 52.02 ± 6.18 dB(A) and mean L_{dn} railway: 23.59 ± 10.44 dB(A)) and non-participants (52.45 ± 6.28 dB(A) and 24.67 ± 11.10 dB(A)). Figure 2 shows the proportion of the study sample exposed to road and rail noise in 5 dB(A) L_{dn} categories. We decided not to conduct analysis on modelled noise exposure to rail noise due to the small number of highly exposed persons (94% and 95% exposed to L_{day} and L_{night} noise levels <40 dB(A), respectively).

Figure 3 shows the distribution of annoyance to various noise sources. The proportion of respondents that reported considerable and heavy annoyance was highest in relation to aircraft noise (21.4%), road traffic noise (13.8%) and neighbour noise (10.2%) and less so for rail (2.4%) and industry noise (1.9%). Univariable regression parameters for annoyance to neighbour noise were found to be strongly associated with annoyance to road (0.21, 95% CI: 0.17, 0.25) and industry (0.17, 95% CI: 0.09, 0.25) noise.

Figure 2. Proportion of the study sample in relation to L_{dn}.

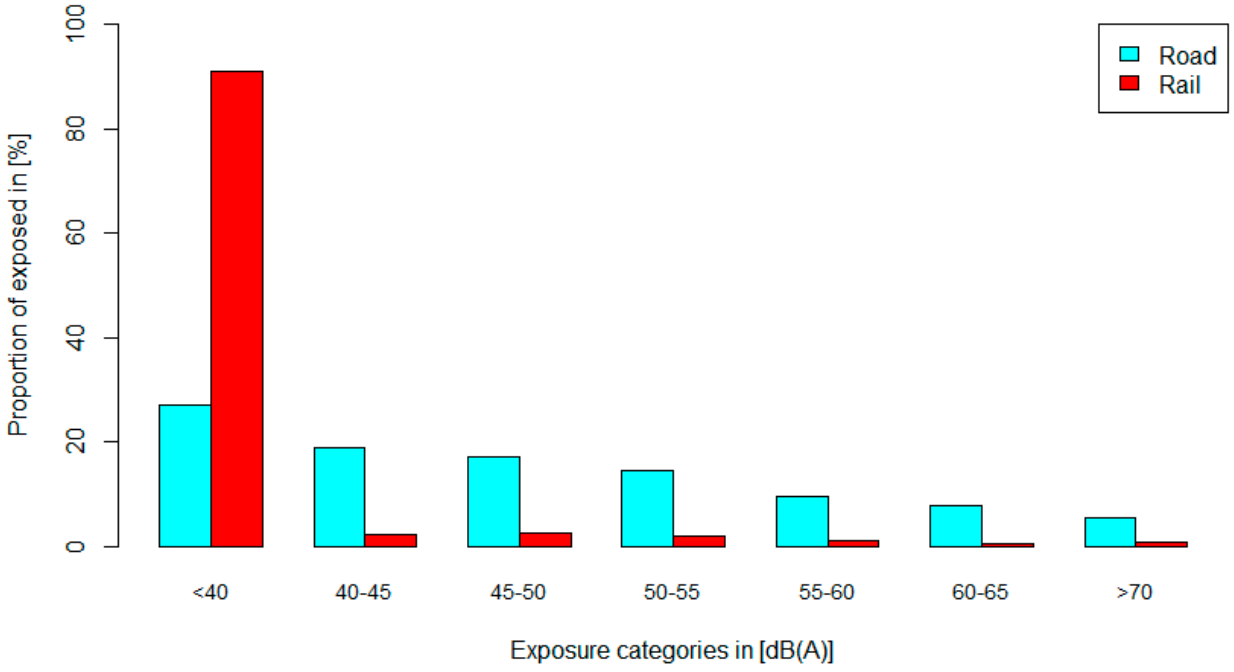


Figure 3. Proportion of the level of annoyance due to different noise sources for the study sample.

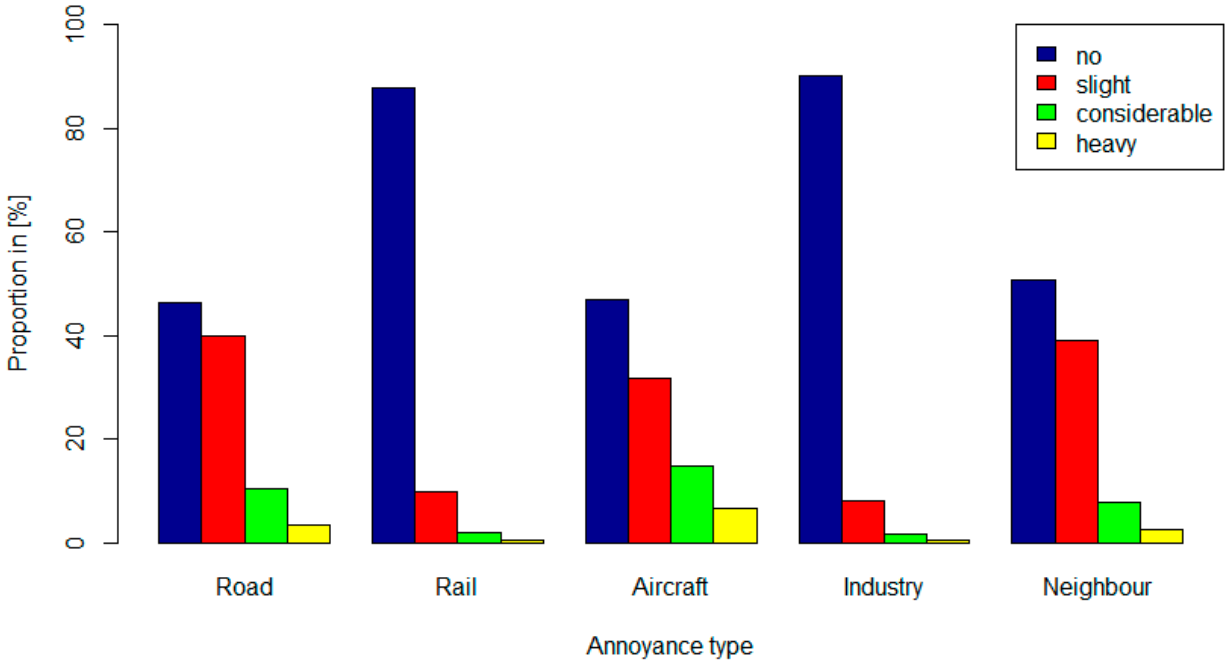
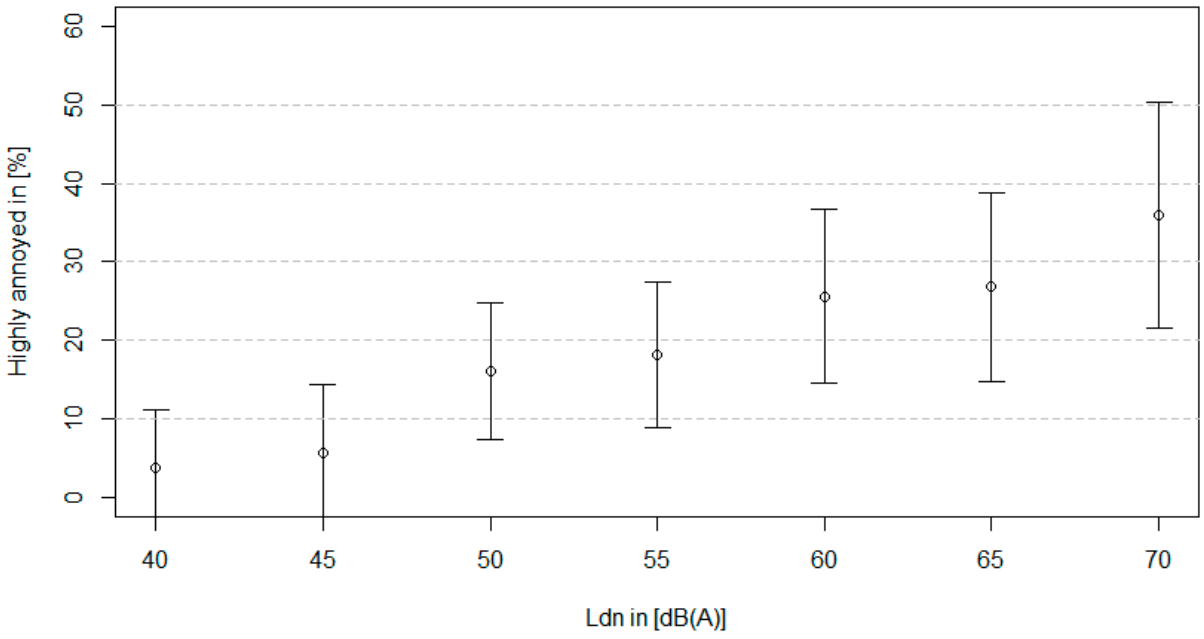


Figure 4 shows the relationship between modelled road traffic noise and annoyance. The proportion of the study sample highly (considerable + heavy) annoyed by road traffic noise reaches 36% at an L_{dn} of 70 dB(A).

Figure 4. Proportion of the study sample highly (considerable + heavy) annoyed in relation to road traffic noise levels.



Crude and adjusted regression parameters for the von Zerssen symptom and SF-36 mental health score are displayed in Table 2.

Table 2. Crude and adjusted increase of the von Zerssen symptom score and the SF-36 mental health score in relation to modelled noise (per 10 dB(A) Ldn) or source specific annoyance (per rating category).

Von Zerssen	**β [95% CI] Crude**	***p*-Value**	**β [95% CI] Adjusted ***	***p*-Value**
Road traffic noise 10 dB(A)	0.59 [0.09, 1.09]	0.02	0.47 [−0.01, 0.95]	0.05
Annoyance road	1.50 [1.06, 1.94]	<0.001	1.53 [1.09, 1.96]	<0.001
Annoyance rail	1.03 [0.22, 1.84]	0.01	0.84 [0.06, 1.63]	0.04
Annoyance aircraft	0.76 [0.35, 1.18]	<0.001	0.73 [0.33, 1.14]	<0.001
Annoyance industry	2.14 [1.30, 2.97]	<0.001	2.36 [1.54, 3.17]	<0.001
Annoyance neighbour	1.61 [1.16, 2.07]	<0.001	1.62 [1.17, 2.06]	<0.001
SF-36 Mental Health	**β [95% CI] Crude**	***p*−Value**	**β [95% CI] Adjusted ***	***p*-Value**
Road traffic noise 10 dB(A)	0.47 [−0.05, 0.98]	0.08	0.09 [−0.43, 0.61]	0.73
Annoyance road	1.16 [0.66, 1.66]	<0.001	1.03 [0.54, 1.52]	<0.001
Annoyance rail	1.49 [0.60, 2.37]	<0.01	1.22 [0.34, 2.10]	0.01
Annoyance aircraft	0.12 [−0.33, 0.58]	0.60	0.21 [−0.25, 0.67]	0.37
Annoyance industry	2.16 [1.22, 3.10]	<0.001	2.20 [1.27, 3.12]	<0.001
Annoyance neighbour	1.47 [0.96, 1.98]	<0.001	1.34 [0.83, 1.84]	<0.001

Note: * Adjusted for age, age^2, sex, physical activity, smoking, education, marital status, region.

After adjusting for covariates, a 10 dB(A) increase of the road traffic noise L_{dn} was associated with a 0.47 (95% CI: −0.01, 0.95) point increase of the von Zerssen symptom score. A substantial increase in the von Zerssen symptom score for annoyance to road, industry and neighbour noise was observed

(>1.5 per unit increase in annoyance rating category), while the link with annoyance to railway and aircraft noise was weaker (<1 point per unit increase).

After adjusting for covariates the SF-36 mental health score was not associated with road traffic noise, whereas it was positively associated with most annoyance types with the exception of annoyance to aircraft noise (Table 2). In the crude and adjusted logistic regression models presented in Table 3, self-reported health status was strongly associated with road traffic noise and annoyance to road traffic and neighbour noise. Annoyance to neighbour noise was positively associated with health indicators in all models.

Table 3. Crude and adjusted odds ratio for decrease of self-reported health status in relation to modelled noise (per 10 dB(A) Ldn) or source specific annoyance (per rating category).

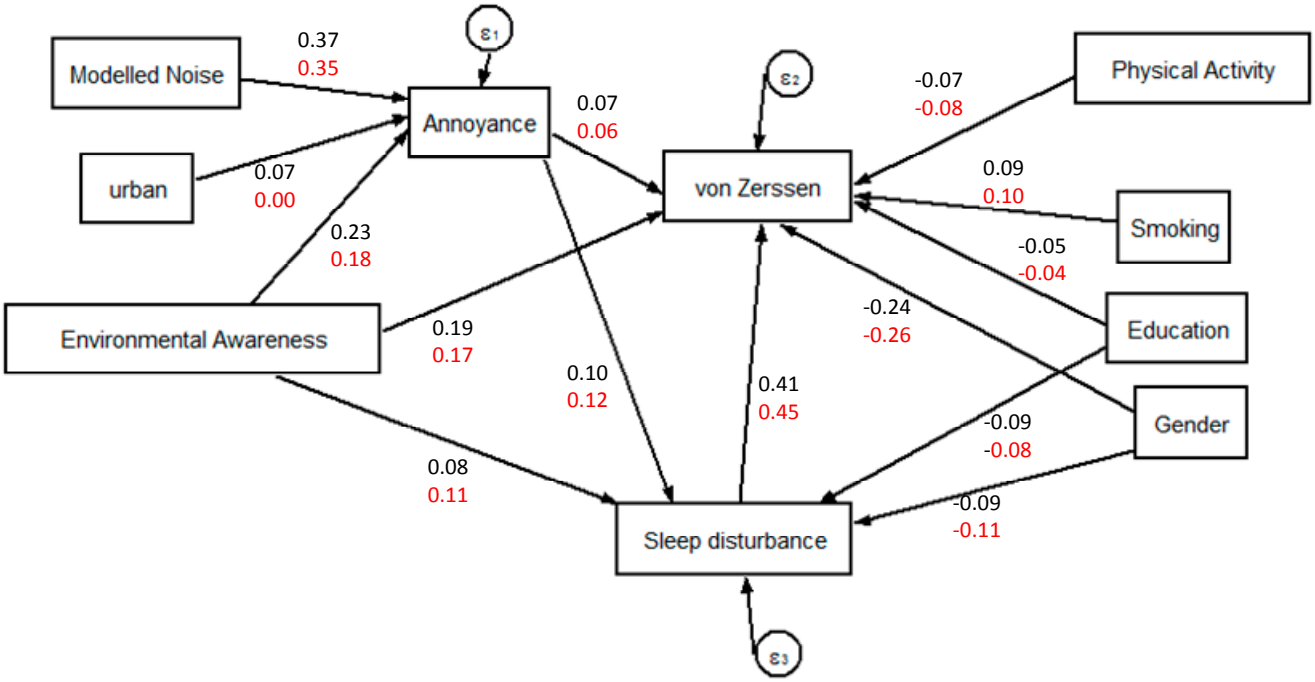
Self-Reported Health Status	OR [95% CI] Crude	*p*-Value	OR [95% CI] Adjusted *	*p*-Value
Road traffic noise 10 dB(A)	1.36 [1.19, 1.55]	<0.001	1.28 [1.12, 1.48]	<0.001
Annoyance road	1.52 [1.32, 1.77]	<0.001	1.45 [1.25, 1.70]	<0.001
Annoyance rail	1.22 [0.95, 1.58]	0.12	1.07 [0.83, 1.40]	0.58
Annoyance aircraft	0.98 [0.85, 1.13]	0.78	0.99 [0.86, 1.15]	0.96
Annoyance industry	1.43 [1.11, 1.88]	0.01	1.28 [0.97, 1.68]	0.08
Annoyance neighbour	1.79 [1.52, 2.08]	<0.001	1.75 [1.49, 2.08]	<0.001

Note: * Adjusted for age, age^2, sex, physical activity, smoking, education, marital status, noise model used.

To test effect modification, stratified analyses were conducted for gender, age, noise exposure levels, BMI, sleep disturbance score and occurrence of comorbidity. In general, we found little indication that these factors act as effect modifiers. Sleep disturbance, however, was found to modify the relationship between road traffic noise and the von Zerssen score (p-value < 0.001), annoyance to aircraft noise and von Zerssen score (p-value = 0.017), annoyance to industry noise and the von Zerssen score (p-value < 0.01), and annoyance to neighbour noise and the von Zerssen score (p-value < 0.01). These associations were stronger for those people who had a higher sleep disturbance score. Stratified analysis conducted for the SF-36 mental health score and the self-reported health status yielded no results and is therefore not shown.

Figure 5 shows the final SEM (Model A) and the Z-normalised parameters for the relationships between road traffic noise, annoyance to road traffic noise, sleep disturbance, the von Zerssen score and their confounders. Separate models for the baseline and the follow-up data yielded equivalent results. The assumed direct relationship between road traffic noise and the von Zerssen score lost significance with the addition of paths between the von Zerssen and other explanatory variables. Path parameters between road traffic noise and annoyance to road traffic noise, and between sleep disturbance and the von Zerssen score display the highest values.

Figure 5. Model A, SEM describing the relation between road traffic noise, annoyance to road traffic noise, sleep disturbance, the von Zerssen score and their confounders. Z-normalised model parameters based on baseline (displayed in black) and follow up (in red) observations.



The path estimates and model fit indices for Model A are shown in Table 4. Both baseline and follow up subsets of Model A have low and non-significant χ^2 test values indicating a good fit of the model parameters with the observed covariance matrix. Other model fit indices confirm this diagnostic.

Table 4. Estimated parameters of SEM, 95% confidence intervals and *p*-values for all relationships and model fit indices for baseline and follow up observations in Model A.

Relationship	Baseline (n = 1307)		Follow up (n = 1064)	
	β [95% CI]	*p*-Value	β [95% CI]	*p*-Value
Direct effects				
Road traffic noise → Annoyance to road traffic	0.37 [0.32, 0.42]	<0.001	0.35 [0.29, 0.40]	<0.001
Degree of urban → Annoyance to road traffic	0.07 [0.02, 0.12]	0.007	0.00 [−0.05, 0.06]	0.929
Environmental Awareness → Annoyance to road traffic	0.23 [0.18, 0.28]	<0.001	0.18 [0.12, 0.23]	<0.001
Annoyance to road traffic → von Zerssen	0.07 [0.02, 0.11]	0.003	0.06 [0.01, 0.11]	0.021
Sleep disturbance → von Zerssen	0.41 [0.36, 0.45]	<0.001	0.45 [0.40, 0.50]	<0.001
Environmental Awareness → von Zerssen	0.19 [0.15, 0.24]	<0.001	0.17 [0.12, 0.22]	<0.001
Physical activity → von Zerssen	−0.07 [−0.11, −0.02]	0.002	−0.08 [−0.13, −0.03]	0.003
Smoking → von Zerssen	0.09 [0.04, 0.13]	<0.001	0.10 [0.05, 0.15]	<0.001
Education → von Zerssen	−0.05 [−0.10, −0.01]	0.022	−0.04 [−0.09, 0.01]	0.097
Gender → von Zerssen	−0.24 [−0.28, −0.19]	<0.001	−0.26 [−0.31, −0.21]	<0.001
Annoyance to road traffic → Sleep disturbance	0.10 [0.05, 0.15]	<0.001	0.12 [0.06, 0.18]	<0.001
Environmental Awareness → Sleep disturbance	0.08 [0.03, 0.14]	0.003	0.11 [0.05, 0.17]	<0.001
Education → Sleep disturbance	−0.09 [−0.15, −0.04]	0.001	−0.08 [−0.14, −0.02]	0.011
Gender → Sleep disturbance	−0.09 [−0.15, −0.04]	0.001	−0.11 [−0.17, −0.05]	<0.001

Table 4. *Cont.*

Relationship	Baseline (n = 1307)		Follow up (n = 1064)	
	β [95% CI]	*p*-Value	β [95% CI]	*p*-Value
Indirect effects				
Annoyance to road traffic → von Zerssen	0.04 [0.02, 0.06]	<0.001	0.05 [0.03, 0.08]	<0.001
Road traffic noise → von Zerssen	0.04 [0.02, 0.06]	<0.001	0.04 [0.02, 0.06]	<0.001
Degree of urban → von Zerssen	0.01 [0.00, 0.01]	0.022	0.00 [−0.01, 0.01]	0.929
Environmental Awareness → von Zerssen	0.06 [0.03, 0.08]	<0.001	0.07 [0.04, 0.10]	<0.001
Education → von Zerssen	−0.04 [−0.06, −0.01]	0.001	−0.03 [−0.06, −0.01]	0.011
Gender → von Zerssen	−0.04 [−0.06, −0.01]	0.001	−0.05 [−0.08, −0.02]	<0.001
Road traffic noise → Sleep disturbance	0.04 [0.02, 0.06]	<0.001	0.04 [0.02, 0.06]	<0.001
Degree of urban → Sleep disturbance	0.01 [0.00, 0.01]	0.031	0.00 [−0.01, 0.01]	0.929
Environmental Awareness → Sleep disturbance	0.02 [0.01, 0.04]	0.001	0.02 [0.01, 0.03]	0.001
Model fit indices	**Baseline**		**Follow up**	
χ^2	3.62		13.42	
p-value χ^2	0.963		0.201	
RMSEA	0.000		0.018	
AIC	36278		28878	
Tucker-Lewis	1.017		0.989	
SRMR	0.006		0.014	

Figure 6. Model B, SEM describing the relation between road traffic noise, annoyance to road traffic noise, sleep disturbance, the SF-36 mental health score and their confounders. Z-normalised model parameters based on baseline (displayed in black) and follow up (in red) observations.

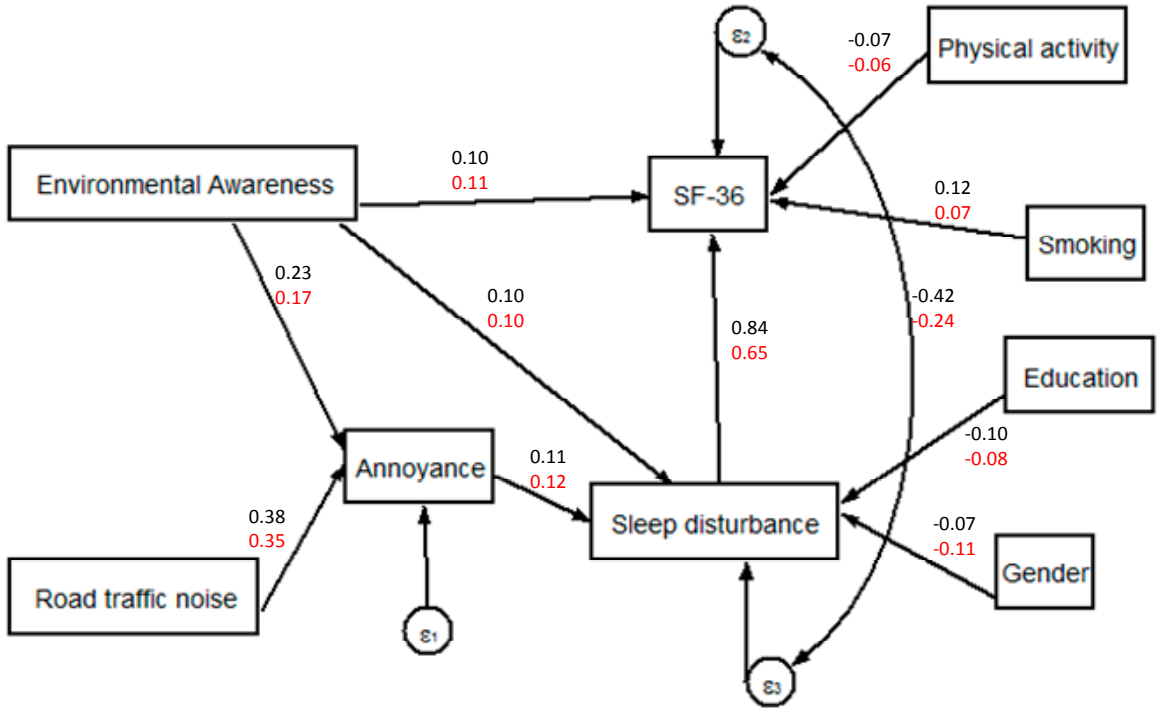


Figure 6 shows the final SEM (Model B) and the Z-normalised parameters for the relationships between road traffic noise, annoyance to road traffic noise, sleep disturbance, the SF-36 mental health score and their confounders. Again, there was no direct relationship between road traffic noise and the

SF-36 mental health score upon addition of paths better explaining the latter variable. In contrast to Figure 5, no significant path between annoyance to road traffic noise and the SF-36 mental health score was identified. An indirect link between road traffic noise and SF-36 was found via annoyance to road traffic noise and sleep disturbance.

The path estimates and model fit indices for Model B are shown in Table 5. Both baseline and follow up subsets of Model B have low and non-significant χ^2 test values indicating a good fit of the model parameters with the observed covariance matrix. Other model fit indices confirm this diagnostic.

Table 5. Estimated parameters of SEM, 95% confidence intervals and *p*-values for all relationships and model fit indices for baseline and follow up observations in Model B.

Relationship	Baseline		Follow_up	
	β [95% CI]	*p*-Value	β [95% CI]	*p*-Value
Direct effects				
Road traffic noise → Annoyance to road traffic	0.38 [0.33, 0.43]	<0.001	0.35 [0.30, 0.41]	<0.001
Environmental Awareness → Annoyance to road traffic	0.23 [0.18, 0.28]	<0.001	0.17 [0.12, 0.22]	<0.001
Sleep disturbance → SF-36	0.84 [0.54, 1.14]	<0.001	0.65 [0.35, 0.95]	<0.001
Environmental Awareness → SF-36	0.10 [0.04, 0.17]	0.003	0.11 [0.04, 0.18]	0.002
Physical activity → SF-36	−0.07 [−0.11, −0.02]	0.004	−0.06 [−0.12, −0.01]	0.025
Smoking → SF-36	0.12 [0.08, 0.17]	<0.001	0.07 [0.02, 0.13]	0.011
Annoyance to road traffic → Sleep disturbance	0.11 [0.06, 0.16]	<0.001	0.12 [0.06, 0.18]	<0.001
Environmental Awareness → Sleep disturbance	0.10 [0.04, 0.15]	<0.001	0.10 [0.05, 0.16]	<0.001
Education → Sleep disturbance	−0.10 [−0.15, −0.05]	<0.001	−0.08 [−0.14, −0.02]	0.007
Gender → Sleep disturbance	−0.07 [−0.12, −0.02]	0.007	−0.11 [−0.17, −0.06]	<0.001
Indirect effects				
Annoyance to road traffic → SF-36	0.09 [0.05, 0.14]	<0.001	0.08 [0.04, 0.12]	<0.001
Road traffic noise → SF-36	0.04 [0.02, 0.06]	<0.001	0.03 [0.01, 0.05]	0.001
Environmental Awareness → SF-36	0.10 [0.04, 0.17]	0.001	0.08 [0.03, 0.13]	0.003
Education → SF-36	−0.09 [−0.13, −0.04]	<0.001	−0.05 [−0.09, −0.01]	0.015
Gender → SF-36	−0.06 [−0.1, −0.02]	0.006	−0.07 [−0.12, −0.03]	0.002
Road traffic noise → Sleep disturbance	0.04 [0.02, 0.06]	<0.001	0.04 [0.02, 0.07]	<0.001
Environmental Awareness → Sleep disturbance	0.03 [0.01, 0.04]	<0.001	0.02 [0.01, 0.03]	0.001
Model fit indices				
χ^2	3.724		10.094	
p-value χ^2	0.959		0.432	
RMSEA	0.000		0.003	
AIC	33991		26379	
Tucker-Lewis	1.018		1.000	
SRMR	0.006		0.012	

4. Discussion

In our analysis using multiple linear models, modelled road traffic noise exposure was strongly associated with self-reported health status but not with the SF-36 mental score and borderline significant with the von Zerssen symptom score. The associations with noise annoyance tended to be stronger and more consistent for all three health indicators, although the pattern was more pronounced for annoyance from road, industry or neighbour noise than for annoyance from railway and aircraft noise.

The SEMs revealed no direct associations linking modelled road traffic noise to the von Zerssen symptom score and the SF-36 mental health score. This finding is in line with previous work [14] where the link between modelled road traffic noise and health outcomes vanished after inclusion of additional variables. However, for both HRQOL indicators we could demonstrate the existence of an indirect path via annoyance and sleep disturbances in both surveys (baseline and follow-up). These indirect paths indicate that annoyance and sleep disturbance act as a mediator for the association between noise exposure and health related quality of life. Interestingly, no direct relationship between annoyance and the SF-36 mental health score was found. According to the work of Stansfeld [29] such a direct relationship may have been observable when including the noise sensitivity in the model, since noise sensitivity affects the psyche and annoyance. Unfortunately, this information is not available in our study.

In both SEMs, the path linking road traffic noise to sleep disturbance vanished after inclusion of the variables education and gender. This is in line with a previous analysis conducted on the present cohort [12] that found no association between road traffic noise exposure and subjective sleep quality. However, a significant association was found between road traffic noise and objective sleep parameters measured by actimetry. The lack of association between road traffic noise and subjective sleep quality implies that people may not be aware of the objective effect of noise on their sleep. This is of particular relevance for research looking at the link between noise and cardiovascular diseases. This further raises the question of the accuracy of annoyance as an indicator for the most severe health effects of noise.

The von Zerssen score is a HRQOL indicator which, to the best of our knowledge, has not yet been used in noise research. Although we used different health indicators our study results are comparable with previous research on this topic [2,7–9]. The direction and magnitude of the observed associations are consistent with the theoretical framework of Soames *et al.* [11]. This demonstrates that noise annoyance and sleep disturbances play important mediating roles for noise induced effects on HRQOL. Our SEMs confirm the statements made by different authors that HRQOL is more closely correlated with reaction and coping of noise exposure than with the physical noise exposure itself [10,11]. The mediator effect of annoyance indicates that both individual coping behaviour and the real noise exposures are important, at least for a common source like road traffic. Conversely, hidden factors triggering annoyance may explain why the proportion of persons highly (considerable and heavy) annoyed by aircraft noise is substantially higher than for any other noise source (Figure 2), although exposure to aircraft noise is relatively low in our study area. According to noise contour maps from the Federal Office of Civil Aviation, no subject in our study sample lived in area with noise ratings [27] $L_{r,\,day}$ exceeding 57 dB(A) [30] and $L_{r,\,23:00\text{-}24:00}$ exceeding 47 dB(A), whereas 19 percent of the study sample is exposed to road traffic above 57 dB(A). Such a high annoyance to aircraft noise could, for instance, be explained by increased awareness to this particular noise source through the controversies on the night traffic bans. This phenomenon for example has been previously observed in Switzerland with respect to shooting noise, where only a low correlation with actual exposure values was observed [3]. Alternatively the few aircraft operations taking place between 23:00 and 24:00 at Basel airport may be triggering annoyance because they are well observable due to the generally low background noise levels. Yet, the weaker association observed between annoyance to aircraft noise and

the three health scales could be attributable to lower aircraft noise exposure or show that high annoyance does not necessarily translate into a decrease of HRQOL.

Exposure to industry and railway noise is also expected to be low in our sample although modelling data to confirm this was only available for railway noise. Contrary to aircraft noise, for both of these sources the proportion of annoyed persons is also low. Nevertheless, the associations of the three health indicators with railway noise annoyance and with industry noise annoyance are quite different, with considerably stronger associations for the latter indicating that annoyance from a specific noise sources is mediated by additional factors.

We investigated whether the low response rate of 37% could lead to bias in our analysis. We found similar noise exposure of non-respondents compared to respondents, ruling out bias for the relationship between road traffic noise and HRQOL. In terms of annoyance, it was not possible to undertake a non-responder analysis thus it is conceivable that more environmentally concerned people have taken part in this study yielding to an overestimation of the proportions of annoyed people. However, associations between annoyance and HRQOL would only be biased if these people also differ in terms of HRQOL.

Potential limitations when dealing with self-reported annoyance and health outcomes include information bias and confounding. People more susceptible to all kinds of environmental and other factors may express more noise annoyance and more symptoms. We adjusted for relevant confounding factors which, in most cases, decreased the association indicating that residual confounding still might play a role, although unlikely to explain the full association. However, in these regards, the absence of adjustment for noise sensitivity and possible exposure misclassification (façade insulation, location of the bedroom and window opening/closing behaviour could not be considered) is a shortcoming of this study. The cross-sectional analysis also did not allow us to address the timing issue; which comes first, the increase in annoyance, the sleep disturbance or the decrease in HRQOL? As in other studies, we did not have the possibility to assess the proportion of people who moved out of noisy areas because of annoyance. Although our analysis is based on a cohort study, a longitudinal analysis was not possible since only about 65 subjects, those who moved in the one year between baseline and follow-up, had a change of exposure between baseline and follow-up. We further saw no significant difference in the L_{dn} values for participants that moved between baseline and follow-up indicating that self-selection is not expected to play a major role.

5. Conclusions

This study demonstrates that sleep disturbances and annoyance play an important role for the effects of road traffic noise on HRQOL.

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Author Contributions

Harris Héritier conducted the analysis and wrote the first paper draft. Martin Röösli designed the study, obtained funding, supervised the analysis and revised the paper draft. Patrizia Frei collected the data. All authors contributed to the interpretation of the data analysis, revised the paper draft and approved the final manuscript.

Conflicts of Interest

The authors declare no conflicts of interest.

References

1. Bluhm, G.; Nordling, E.; Berglind, N. Road traffic noise and annoyance—An increasing environmental health problem. *Noise Health* **2004**, *6*, 43–49.
2. Öhrström, E.; Skånberg, A.; Svensson, H.; Gidlöf-Gunnarsson, A. Effects of road traffic noise and the benefit of access to quietness. *J. Sound Vib.* **2006**, *295*, 40–59.
3. Brink, M.; Wunderli, J.-M. A field study of the exposure-annoyance relationship of military shooting noise. *J. Acoust. Soc. Am.* **2010**, *127*, 2301–2311.
4. Miedema, H.; Oudshoorn, C. Annoyance from transportation noise: Relationships with exposure metrics DNL and DENL and their confidence intervals. *Environ. Health Perspect.* **2001**, *109*, 409–416.
5. Miedema, H.M.; Vos, H. Exposure-response relationships for transportation noise. *J. Acoust. Soc. Am.* **1998**, *104*, 3432–3445.
6. WHO. *Burden of Disease from Environmental Noise*; WHO: Bonn, Germany, 2011.
7. Öhrström, E. Longitudinal surveys on effects of changes in road traffic noise—Annoyance, activity disturbances, and psycho-social well-being. *J. Acoust. Soc. Am.* **2004**, *115*, 719–729.
8. Schreckenberg, D.; Meis, M.; Kahl, C.; Peschel, C.; Eikmann, T. Aircraft noise and quality of life around Frankfurt Airport. *Int. J. Environ. Res. Public Health* **2010**, *7*, 3382–3405.
9. Dratva, J.; Zemp, E.; Dietrich, D.F.; Bridevaux, P.-O.; Rochat, T.; Schindler, C.; Gerbase, M.W. Impact of road traffic noise annoyance on health-related quality of life: Results from a population-based study. *Qual. Life Res.* **2010**, *19*, 37–46.
10. Stallen, P. A theoretical framework for environmental noise annoyance. *Noise Health* **1999**, *1*, 69–80.
11. Soames Job, R. Noise sensitivity as a factor influencing human reaction to noise. *Noise Health* **1999**, *1*, 57–68.
12. Frei, P.; Mohler, E.; Röösli, M. Effect of nocturnal road traffic noise exposure and annoyance on objective and subjective sleep quality. *Int. J. Hyg. Environ. Health* **2014**, *217*, 188–195.
13. Fyhri, A.; Aasvang, G.M. Noise, sleep and poor health: Modeling the relationship between road traffic noise and cardiovascular problems. *Sci. Total Environ.* **2010**, *408*, 4935–4942.

14. Fyhri, A.; Klæboe, R. Road traffic noise, sensitivity, annoyance and self-reported health—A structural equation model exercise. *Environ. Int.* **2009**, *35*, 91–97.
15. Kroesen, M.; Molin, E.J.; van Wee, B. Testing a theory of aircraft noise annoyance: A structural equation analysis. *J. Acoust. Soc. Am.* **2008**, *123*, 4250–4260.
16. Sørensen, M.; Andersen, Z.J.; Nordsborg, R.B.; Becker, T.; Tjønneland, A.; Overvad, K.; Raaschou-Nielsen, O. Long-term exposure to road traffic noise and incident diabetes: A cohort study. *Environ. Health Perspect.* **2013**, *121*, 217–222.
17. Röösli, M.; Mohler, E.; Frei, P.; Vienneau, D. Noise-related sleep disturbances: Does gender matter? *Noise Health* **2014**, *16*, 197–204.
18. Frei, P.; Mohler, E.; Braun-Fahrländer, C.; Fröhlich, J.; Neubauer, G.; Röösli, M. Cohort study on the effects of everyday life radio frequency electromagnetic field exposure on non-specific symptoms and tinnitus. *Environ. Int.* **2012**, *38*, 29–36.
19. Mohler, E.; Frei, P.; Braun-Fahrländer, C.; Fröhlich, J.; Neubauer, G.; Röösli, M. Effects of everyday radiofrequency electromagnetic-field exposure on sleep quality: A cross-sectional study. *Radiat. Res.* **2010**, *174*, 347–356.
20. Eurostat. European Health Interview Survey (EHIS wave 2). In *Methodological Manual*; Publications Office of the European Union: Luxembourg, Luxembourg, 2013.
21. Von Zerssen, D.; Petermann, F. *B-LR—Beschwerden-Liste—Revidierte Fassung*; Hogrefe: Göttingen, Germany, 2011.
22. Ware, J.E.; Kosinski, M.; Dewey, J.E.; Gandek, B. *SF-36 Health Survey: Manual and Interpretation Guide*; Quality Metric Inc.: Montréal, Canada, 2000.
23. SFSO. *Swiss Health Survey 2007*; SFSO: Neuchâtel, Switzerland, 2012.
24. Wirth, K.; Brink, M.; Rometsch, R.; Schierz, C. *Lärmstudie 2000 Zusammenfassung*; Zentrum für Organisations-und Arbeistswissenschaften ETH: Zürich, Switzerland, 2005.
25. Höin, R.; Ingold, K.; Köpfli, M.; Minder, T. SonBase—Die GIS-Lärmdatenbank der Schweiz. Grundlagen. In *Umwelt-Wissen Nr. 0908*; Bundesamt für Umwelt BAFU: Bern, Switzerland 2009.
26. Karipidis, I.; Vienneau, D.; Habermacher, M.; Köpfli, M.; Brink, M.; Probst-Hensch, N.; Röösli, M.; Wunderli, J.-M. Reconstruction of historical noise exposure data for environmental epidemiology in Switzerland withing the SiRENE project. *Noise Mapp.* **2014**, *1*, 3–14.
27. Noise Abatement Ordinance of 15 December 1986. Available online: http://www.admin.ch/opc/en/classified-compilation/19860372/index.html (accessed on 1 December 2014).
28. Kannapin, O.; Pawlik, K.; Zinn, F. The pattern of variables predicting self-reported environmental behavior. *Z. Exp. Psychol.* **1997**, *45*, 365–377.
29. Stansfeld, S.A. Noise, noise sensitivity and psychiatric disorder: Epidemiological and psychophysiological studies. *Psychol. Med. Monogr. Suppl.* **1992**, *22*, 1–44.
30. BAZL. Lärmbelastungskataster Landesflughafen Basel-Mulhouse. In *Zivilluftfahrt*; Bundesamt für Zivilluftfahrt BAZL: Bern, Switzerland, 2009.

5. Paper 2

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Transportation noise exposure and cardiovascular mortality: a nationwide cohort study from Switzerland.

Harris Héritier[1,2]*; Danielle Vienneau[1,2]*; Maria Foraster[1,2], Ikenna C. Eze[1,2], Emmanuel Schaffner[1,2] Laurie Thiesse[3], Franziska Rudzik[3], Manuel Habermacher[4], Micha Köpfli[4], Reto Pieren[5], Mark Brink[6], Christian Cajochen[3], Jean Marc Wunderli[5], Nicole Probst-Hensch[1,2], Martin Röösli [1,2] for the SNC study group

[1] Swiss Tropical and Public Health Institute, Basel, Switzerland

[2] University of Basel, Basel, Switzerland

[3] Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Basel, Switzerland

[4] N-sphere AG, Zürich, Switzerland

[5] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Dubendorf, Switzerland.

[6] Federal Office for the Environment, Bern, Switzerland

* both authors contributed equally

Transportation noise exposure and cardiovascular mortality: a nationwide cohort study from Switzerland.

Harris Héritier[1,2]*; Danielle Vienneau[1,2]*; Maria Foraster[1,2], Ikenna C. Eze[1,2], Emmanuel Schaffner[1,2] Laurie Thiesse[3], Franziska Rudzik[3], Manuel Habermacher[4], Micha Köpfli[4], Reto Pieren[5], Mark Brink[6], Christian Cajochen[3], Jean Marc Wunderli[5], Nicole Probst-Hensch[1,2], Martin Röösli [1,2] for the SNC study group

[1] Swiss Tropical and Public Health Institute, Basel, Switzerland

[2] University of Basel, Basel, Switzerland

[3] Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Basel, Switzerland

[4] N-sphere AG, Zürich, Switzerland

[5] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Dubendorf, Switzerland.

[6] Federal Office for the Environment, Bern, Switzerland

* both authors contributed equally

Correspondence:

Martin Röösli

Swiss Tropical and Public Health Institute

Socinstrasse 57

P.O. Box

CH-4002 Basel

E-Mail martin.roosli@unibas.ch

Tel. +41 (0)61 284 83 83

Fax +41 (0)61 284 85 01

Abstract

Background: Most studies published to date consider single noise sources and the reported noise metrics are not informative about the peaking characteristics of the source under investigation. Our study focuses on the association between cardiovascular mortality in Switzerland and the three major transportation noise sources – road, railway and aircraft traffic– along with a novel noise metric termed intermittency ratio (IR), expressing the percentage contribution of individual noise events to the total noise energy from all sources above background levels.

Methods: We generated Swiss-wide exposure models for road, railway and aircraft noise for 2001. Noise from the most exposed façade was linked to geocodes at the residential floor height for each of the 4.41 million adult (>30y) Swiss National Cohort (SNC) participants. For the follow-up period 2000 to 2008, we investigated the association between all noise exposure variables (L_{den}(Road), L_{den}(Rail), L_{den}(Air), and IR at night) and various cardiovascular primary causes of death by multipollutant Cox regression models adjusted for potential confounders including NO_2.

Results: The most consistent associations were seen for myocardial infarction: adjusted hazard ratios (HR) (95% CI) per 10 dB increase of exposure were 1.038 (1.019-1.058), 1.018 (1.004-1.031), and 1.026 (1.004-1.048) respectively for L_{den}(Road), L_{den}(Rail), and L_{den}(Air). In addition, total IR at night played a role: HRs for CVD were non-significant in the 1st, 2nd and 5th quintiles whereas they were 1.019 (1.002-1.037) and 1.021 (1.003-1.038) for the 3rd and 4th quintiles.

Conclusion: Our study demonstrates the impact of all major transportation noise sources on cardiovascular diseases. Mid-range IR levels at night (i.e. between continuous and highly intermittent) are potentially more harmful than continuous noise levels of the same average level.

Keywords: road traffic, railway, aircraft, noise, exposure, Lden, cardiovascular mortality, intermittency ratio

Introduction

There is growing epidemiological evidence that transportation noise affects health in various ways. Transportation noise induces sympathetic and endocrine arousals that release stress hormones such as catecholamine and cortisol in the blood stream [1]. Further it induces annoyance [2], affects health-related quality of life [3-5], reduces sleep quality [6], and has been shown to affect physical activity levels [7]. All these effects are expected to increase the incidence and progression of hypertension and myocardial infarction (MI) [8, 9].

Blood pressure has been the most studied outcome in noise research so far. In a meta-analysis, Van Kempen and Babisch report a pooled odds ratio for hypertension prevalence of 1.03 (95% CI: 1.01, 1.06) per 5 dB increase in road traffic noise levels [10]. With respect to aircraft noise, meta-analyses on hypertension reported risk increase of 1.63 (95% CI: 1.14-2.33) [11] and 1.13 (95% CI: 1.00-1.28) [12] per 10 dB increase in exposure. For MI, a recent meta-analysis reports a linear exposure-response relationship within the range of 52‑77 dB with a relative risk of 1.08 (95% CI: 1.04-1.13) per 10 dB increase in road traffic noise exposure [13], while another meta-analysis reported a relative risk for ischemic heart disease (IHD) of 1.04 (95% CI: 1.00-1.10) per 10 dB increase of road traffic noise exposure starting at a threshold of 50 dB [14]. For stroke, however, the few published studies do not show a consistent pattern; some point to an association with transportation noise [15, 16] whereas others do not [17-20].

Other cardiovascular outcomes such as heart failure have rarely been addressed, and studies on railway noise are scarce and restricted to outcomes like sleep medication intake [21], weight gain [22], waist circumference [23], blood pressure [24], heart failure [25] and hypertensive heart disease but cardiovascular mortality has not yet been investigated.

Further, most studies to date have considered a single noise source as the exposure variable. Thus, there is considerable uncertainty about the mutual independent exposure-response curve in the presence of multiple noise sources. Moreover, all previous studies on chronic diseases represent exposure using time-averaged noise levels, predominantly the L_{eq}-based metrics like Ldn, Lden, $L_{eq,Night}$ or $L_{eq,Day}$. However, there is evidence that temporal characteristics of noise also play an important role, e.g. when it comes to awakening reactions at night. There is evidence that the probability of event-related awakenings, body movements and cardiovascular arousals depend on the maximum sound pressure level and the slope of rise of individual events [26-28] . As a

consequence, such effects are less well predicted by average levels [29]. Since noise events during night have been linked with cardiac arousal on the short term [27, 30], we hypothesize that single but pronounced noise events during night are involved in the development of cardiovascular disease morbidity and mortality in the long run irrespective of the source of such events. Therefore, to describe the "eventfulness" of the noise exposure situation, we developed the noise metric “intermittency ratio” (IR in %), which quantifies the contribution of individual noise events above the background level to the total noise exposure [31].

By combining source specific nationwide noise models with the Swiss National Cohort (SNC) data, the aim of this study was to better understand the mutual independent link between cardiovascular mortality in adults and the residential exposure to the levels and intermittency of major transportation noise sources in Switzerland. In addition, we evaluated potential modifying effects of sex, age, socio-economic status, urbanization, building age, duration of residence and number of noise events.

Methods

Study population

The SNC probabilistically links national census data with mortality and emigration records [32]. The data used in our study is based on the 4 December 2000 census and on mortality and emigration data for the period 5 December 2000 to 31 December 2008 and contains 7.28 million observations. We excluded subjects below 30 years of age (n=2.59 million), observations for which residential coordinates were missing (n=0.19 million), subjects living in an institution (n=0.25 million), and observations for which the cause of death was imputed (n=0.03 million) leaving 4.41 million observations for the analyses.

The outcomes under investigation were primary causes of death from all cardiovascular diseases (CVD) (ICD-10: I00-I99), IHD (ICD-10: I20-I25), stroke in general (ICD-10: I60-I64), hemorrhagic stroke (ICD-10: I60-I62), ischemic stroke (ICD-10: I63), MI (ICD-10: I21-I22), heart failure (ICD-10: I50), and blood-pressure related death (BP) (ICD-10: I10-I15).

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

Noise exposure data

Within the framework of the SiRENE project (Short and Long Term Effects of Transportation Noise Exposure), we built a Swiss-wide noise exposure database for the year 2001 which includes the three major transportation noise sources in Switzerland: road traffic, railway and aircraft noise.

The noise exposure database is described in detail elsewhere [33]. In brief, road traffic noise emissions were calculated using sonROAD [34] while propagation was computed via the propagation model of StL-86 [35]. For railway noise, the emissions were calculated using sonRAIL [36] and propagation was computed using the Swiss railway noise model SEMIBEL [37]. For aircraft noise, the three major civils airports; Zürich, Geneva and Basel, and the military airport located in Payerne were considered. Noise exposure estimates were calculated via FLULA2 [38], based on radar data for Zürich while for Geneva and Basel exposure was calculated on the basis of 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 of Payerne, noise exposure estimates were computed based on idealized flight paths, number of flights and approximate operation times.

IR calculation has already been described elsewhere in detail [31]. In brief, the event-based sound pressure was computed for all vehicle pass-bys obtained from traffic flow statistics. All event-based sound pressure levels that exceed the background level, defined as the modeled hourly L_{eq}, by 3 dB were classified as perceivable events (level L_{eq}, T, Events). The ratio of these perceivable events and the overall sound pressure level $L_{eq,T,tot}$ is termed the intermittency ratio (IR) as follows:

$$IR \equiv \frac{10^{0.1L_{eq,T,Events}}}{10^{0.1L_{eq,T,tot}}} * 100$$

IR takes values in the range 0% to 100%, where 0% IR means that no single events can be perceived above background and 100% IR means that all noise energy is produced by “individual” noise events. In this study we only considered IR at night (23 to 7h) from all transportation sources combined.

For each building in Switzerland, noise exposure was estimated at pre-defined façade points. A maximum of three façade points, spaced by at least 5 meters, were assigned to each building façade by floor. For each façade point, we calculated the L_{den} (defined as the weighted logarithmic average of $L_{eq,day}$, $L_{eq,evening}$ and $L_{eq,night}$ with a penalty of 5 dB and 10 dB respectively applied to the latter exposure variables) for each noise source. Using the available geocodes and the information about floor of residence, we linked participants to their respective dwelling unit to assign noise exposure. Exposure, including IR, was assigned on the basis of the façade point per

dwelling unit with the highest L_{den} value. If information on floor of residence was not available, we assigned the noise estimates corresponding to the middle floor of the building.

Statistical Analysis

We analyzed the data using the Cox proportional hazards model with age as the underlying time variable. Participants were followed until emigration, death or end of follow-up. L_{den} exposure variables were censored at 35 dB (road traffic) or 30 dB (railway and aircraft noise). Hazard ratios for the outcomes of interests were computed using multipollutant models, i.e. including linear terms for each noise source shown in the following equation.

$$Outcome = L_{den}(Road) + L_{den}(Rail) + L_{den}(Air) + adjustments$$

The linearity assumption was evaluated in models with categorical exposure data. We included quintiles of IR night as a categorical variable in the model based on a significant log-likelihood ratio test. Each model was adjusted for sex (female/male), neighborhood index of socio-economic position (low, medium, high), civil status (single, married, widowed, divorced), education level (compulsory education or less, upper secondary level education, tertiary level education, not known), annual average NO_2 concentration ($\mu g/m^3$, estimated outdoor residential levels using PolluMap, a 100x100m dispersion model for Switzerland for year 2010 [39]), mother tongue (German and Rhaeto-Romansch, French, Italian, other language) and nationality (Swiss, rest of Europe (inclusive ex-USSR), rest of the world/unknown). Stratification analysis was conducted by sex (female vs. male), age (<65 vs. >=65 years old), building age (newer than 30 years or renovated vs. older than 30 years without renovation), movers (more than 5 years of residence vs. less than 5 years of residence), number of events that exceed the background level (median split), socio-economic position (median split), and urbanization (urban vs. rural).

Results

The study population amounted to 4.41 million observations, with 33.85 million person-years for the period 5th December 2000 to 31st December 2008. Characteristics of the study population are displayed in Table 1. In general, populations highly exposed to road and railway noise (L_{den} >55 dB, IR>80%) had a lower socioeconomic position and education than the total population, whereas it was the other way around for people highly exposed to aircraft noise. For all three highly noise exposed populations, the proportion of non-Swiss residents and the NO_2 concentration tended to be increased.

The cohort contained 142,955 deaths from CVD, of which 42.2%, 15.7% and 9.4% were deaths from IHD, stroke and BP-related diseases. Exposure to road traffic noise was most prevalent with >89% of the study population exposed to L_{den} above 45 dB. With respect to railway and aircraft noise, 25% and 14% of the study population were respectively exposed to L_{den} above 45 dB. The proportion of censored observations was 1.2%, 45.5% and 67.1% for road, railway and aircraft noise, respectively. The distribution of exposure to IR was skewed toward higher values with a mean of 69.5% and a median of 75.0%. More than 39% of the population was exposed to IR above 80%. These patterns are displayed on Figure 1.

Correlations between the three noise sources were low: Spearman's rank correlation for L_{den}(Air) vs L_{den}(Road): 0.09, L_{den}(Rail) vs L_{den}(Road): 0.12, L_{den}(Air) vs L_{den}(Rail): -0.04. Correlations for IR night with the transportation specific noise levels were also low: (L_{den}(Air): -0.07, L_{den}(Rail): 0.14 and L_{den}(Road): 0.04).

Table 2 shows the hazard ratios (HR) per 10 dB increase in L_{den} of all three transportation sources, included in one multipollutant model by outcome. The most consistent associations were observed for road traffic noise for which the HR for all CVD was 1.025 (95% CI: 1.018-1.032) per 10 dB. The only outcomes not related to road traffic noise were stroke in general (HR=1.011, 95% CI: 0.993-1.028) and hemorrhagic stroke (HR=1.004, 95% CI: 0.968-1.040). Ischemic stroke was significantly associated with road traffic noise (HR=1.050, 95% CI: 1.002-1.099). For railway noise exposure, significant positive linear associations were seen for IHD and MI with a borderline significant association for all CVD. Aircraft noise was not associated with all CVD (HR=0.994, 95% CI: 0.985-1.002). However, significantly elevated risks were observed for MI, heart failure and ischemic stroke. Categorical analyses demonstrated for most outcomes approximate linear associations, with the risk starting to increase as low as 40 dB (Supplementary Figures 2-9).

Night time IR was significantly associated with CVD (Table 3 and supplementary figures 10-17) in a bell shaped relationship where the HR for CVD were 1.019% (95% CI: 1.002-1.037) and 1.021% (95% CI: 1.003-1.038) in the 3rd and 4th quintiles. This trend was observed for all other outcomes although statistical significance was only reached for hemorrhagic stroke. Depending on the outcome, the highest HRs were located in the second (heart failure), third (stroke, ischemic stroke), or fourth (IHD, hemorrhagic stroke) quintile. This indicates that the health effects of noise are more pronounced in situations where the noise is between continuous and highly intermittent.

We evaluated a number of potential effect modifiers on all CVD, MI, ischemic stroke, BP and heart failure. For road and railway noise L_{den}, higher HRs for CVD were observed for men compared to women, although not

confirmed in the other outcomes. For all exposure sources and all outcomes, in tendency, HRs were larger in younger than in older individuals (≥65 year). For urbanization a mixed pattern was seen depending on outcome and exposure source. We did not find effect modification when considering building age, duration of living at the same residence, number of events that exceed background level, or socio-economic position (Supplementary Table 4-8).

Discussion

This study adds to the body of evidence linking transportation noise to CVD and, for the first time, addresses the impact of noise characteristics using a novel metric. In general, IR night appears to be non-linearly associated with the outcomes, which suggests that either continuous noise or highly intermittent noise with long quiet periods between events is less problematic than mid-range intermittency.

We observed a risk increase for heart failure, BP related, ischemic stroke and MI mortality in the road traffic noise levels independent of the other noise sources. Our exposure-response associations are fairly well in line with previous meta-analyses suggesting somewhat stronger risks but with a higher threshold [13, 10, 14]. Previous research on road traffic noise and stroke was inconsistent. Our study is the first to show that ischemic stroke but not hemorrhagic stroke is associated with road traffic noise. Of note, for hemorrhagic stroke IR was a significant predictor and we found borderline significant associations with railway noise, which is highly intermittent during night. This might indicate that for hemorrhagic stroke eventfulness of noise exposure is more relevant than the average level.

Most previous noise research on long-term transportation noise effects has not considered railway noise. In our study we observe an association between railway noise and death from IHD, MI and CVD as primary cause of death, but we cannot confirm a link with blood pressure and heart failure as seen in a German case control study [25]. Strikingly, categorical models suggest that the risk for CVD, IHD, and MI starts to increase at L_{den} levels as low as 30 dB (Supplementary Figures 2, 4, 5). Similar to Huss et al. [20], though with three additional years of SNC data and a greatly improved noise exposure model, we confirmed a significant association between aircraft noise and MI. In addition, aircraft noise was linked to heart failure and ischemic stroke. In general, effects of aircraft noise on all types of CVD were less pronounced than as seen for road and railway noise; a pattern which was also observed in the NORAH study [25]. In Switzerland there is a night ban on air traffic, meaning that during most of the night people are not exposed to aircraft noise, which may explain weaker associations for CVD. On the other hand this would imply that for MI, heart failure and ischemic stroke the

daytime, early morning or late night exposure to aircraft noise is involved in mechanisms triggering those particular outcomes. Further, self-selection may play a role in explaining the absence of association for aircraft noise exposure: subjects not able to cope with aircraft noise may tend to move away from noisy areas. Alternative explanations include residual confounding (socioeconomic position and educational levels tended to be higher in people exposed to aircraft noise) and exposure misclassification in the high exposure range (since airports have to subsidize the installation of sound-proof windows in areas exceeding certain noise levels).

The recently published ESCAPE study reports a weak association between road traffic noise and hypertension that tended to be attenuated after adjustment for $PM_{2.5}$ [41]. In our study, neither the direction nor the strength of association of any noise source substantially changed after adjustment for NO_2 (data not shown). However, we cannot rule out residual confounding from fine particles.

We did not find strong effect modification by sociodemographic factors. But higher relative risks observed for males, with respect to road and railway noise, are in line with an observational sleep study [42] and a recent meta-analysis on myocardial infarction [14].

Additional effect of noise intermittency

A particular asset of this study is the evaluation of the temporal noise exposure characteristics in addition to the noise levels. This IR night refers to the events above the background level and thus can be high even if L_{den} is low. The underlying hypothesis is that in quiet places even relatively low-level noise events during night may have an additional health effect compared to the average noise levels expressed as L_{den}. We found indications that a moderate IR night level (2^{nd}-4^{th} quintile) was more relevant than continuous noise (quintile 1) or highly variable noise (quintile 5) for increased risk of all CVD and IHD. This bell-shaped relationship indicates that subjects appear to be better able to cope with either continuous noise or highly variable noise. The latter may represent situations with very few distinct events and long quiet periods between. Interestingly, depending on the outcome different trends were apparent: for heart failure continuous levels with low level of intermittency seemed to be more problematic whereas for IHD it was the other way round.

Public health impact

While a risk increase of 2.5% per 10 dB road traffic noise exposure for cardiovascular mortality seems negligible, the public health impact is substantial. Transportation noise in 2010 in Switzerland was attributed to 6000 years of life lost which represents 1.8 billion CHF of external costs when also accounting for loss in

housing prices due to noise exposure [43]. Our study contributes to a better understanding of the role of noise characteristics; this should be considered in future noise regulation which currently focuses only on average noise levels.

Strengths and limitations

The strengths of this study are: the large study population without potential for selection bias; the large sample size combined with the long follow-up time which gives us high statistical power; and the very comprehensive noise modeling, for all three transportation sources, at the address level taking into account floor information. This allowed us to conduct an individual exposure assessment, and to assess the individual impact of all three sources concurrently which has rarely been done in previous large population studies. Finally, the IR metric allowed us to refine our exposure estimates, and in general this metric opens new research avenues in the field of noise epidemiology and in the regulation of noise.

While our models were adjusted for socioeconomic status and other demographic variables, there is no information about individual lifestyle and smoking in the SNC. Thus, we cannot rule out that residual confounding of lifestyle may have played a role for our analyses, although no indications for this were seen in a previous SNC noise study [20].

Our study further relied on death records, which may be considered a weakness as it does not represent incidence data. From a mortality perspective, bias would only be a problem if the quality of the death records is related to noise exposure. Given that noise exposure varies on a small scale this is unlikely. Furthermore, a meta-analysis of studies on MI and road traffic did not find systematic differences between mortality and incidence studies [14].

Although, substantial effort in historical exposure modeling was undertaken in the SiRENE project, the potential for exposure misclassification is a limitation in the present work, as it is in other epidemiological noise studies. Modeling errors of the three propagation models, e.g. from meteorology, was reported to be between 0.4 and 2.0 dB [44]. Additional uncertainty comes from the missing input data, missing information about residential history and location of bedroom, and personal behavior (such as window closing behavior and other coping strategies). However, these errors are more likely to be non-differential, thus resulting in an underestimation of the true risks.

In conclusion, this study confirmed previous results and generated new knowledge in the field of noise exposure epidemiology. Not only the nature of noise but its characteristics were found to be relevant for health. This finding may be of importance for noise protection policies.

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Disclosure

Conflict of interests: None

References

1. Münzel T, Gori T, Babisch W, Basner M. Cardiovascular effects of environmental noise exposure. European heart journal. 2014;35(13):829-36.
2. Miedema H, Oudshoorn C. Position Paper on Dose Response Relationships between Transportation Noise and Annoyance. EU's Future Noise Policy, WG2–Dose/Effect. 2002;20.
3. Héritier H, Vienneau D, Frei P, Eze IC, Brink M, Probst-Hensch N et al. The association between road traffic noise exposure, annoyance and health-related quality of life (HRQOL). International journal of environmental research and public health. 2014;11(12):12652-67.
4. Roswall N, Høgh V, Envold-Bidstrup P, Raaschou-Nielsen O, Ketzel M, Overvad K et al. Residential exposure to traffic noise and health-related quality of life—A population-based study. PloS one. 2015;10(3):e0120199.
5. Dratva J, Zemp E, Dietrich DF, Bridevaux P-O, Rochat T, Schindler C et al. Impact of road traffic noise annoyance on health-related quality of life: Results from a population-based study. Quality of life research. 2010;19(1):37-46.
6. Frei P, Mohler E, Röösli M. Effect of nocturnal road traffic noise exposure and annoyance on objective and subjective sleep quality. International journal of hygiene and environmental health. 2014;217(2):188-95.
7. Foraster M, Eze IC, Vienneau D, Brink M, Cajochen C, Caviezel S et al. Long-term transportation noise annoyance is associated with subsequent lower levels of physical activity. Environment International. 2016;91:341-9.
8. Basner M, Babisch W, Davis A, Brink M, Clark C, Janssen S et al. Auditory and non-auditory effects of noise on health. The Lancet. 2014;383(9925):1325-32.

9. Tzoulaki I, Elliott P, Kontis V, Ezzati M. Worldwide Exposures to Cardiovascular Risk Factors and Associated Health Effects Current Knowledge and Data Gaps. Circulation. 2016;133(23):2314-33.
10. Van Kempen E, Babisch W. The quantitative relationship between road traffic noise and hypertension: a meta-analysis. Journal of hypertension. 2012;30(6):1075-86.
11. Huang D, Song X, Cui Q, Tian J, Wang Q, Yang K. Is there an association between aircraft noise exposure and the incidence of hypertension? A meta-analysis of 16784 participants. Noise and Health. 2015;17(75):93.
12. Babisch W, Van Kamp I. Exposure-response relationship of the association between aircraft noise and the risk of hypertension. Noise and Health. 2009;11(44):161.
13. Babisch W. Updated exposure-response relationship between road traffic noise and coronary heart diseases: A meta-analysis. Noise and Health. 2014;16(68):1.
14. Vienneau D, Schindler C, Perez L, Probst-Hensch N, Röösli M. The relationship between transportation noise exposure and ischemic heart disease: a meta-analysis. Environmental research. 2015;138:372-80.
15. Hansell AL, Blangiardo M, Fortunato L, Floud S, de Hoogh K, Fecht D et al. Aircraft noise and cardiovascular disease near Heathrow airport in London: small area study. BMJ : British Medical Journal. 2013;347. doi:10.1136/bmj.f5432.
16. Sørensen M, Hvidberg M, Andersen ZJ, Nordsborg RB, Lillelund KG, Jakobsen J et al. Road traffic noise and stroke: a prospective cohort study. European heart journal. 2011;32(6):737-44.
17. Correia AW, Peters JL, Levy JI, Melly S, Dominici F. Residential exposure to aircraft noise and hospital admissions for cardiovascular diseases: multi-airport retrospective study. BMJ : British Medical Journal. 2013;347. doi:10.1136/bmj.f5561.
18. Evrard A-S, Bouaoun L, Champelovier P, Lambert J, Laumon B. Does exposure to aircraft noise increase the mortality from cardiovascular disease in the population living in the vicinity of airports? Results of an ecological study in France. Noise and Health. 2015;17(78):328.
19. Gan WQ, Davies HW, Koehoorn M, Brauer M. Association of long-term exposure to community noise and traffic-related air pollution with coronary heart disease mortality. American journal of epidemiology. 2012;175(9):898-906.
20. Huss A, Spoerri A, Egger M, Röösli M, Group SNCS. Aircraft noise, air pollution, and mortality from myocardial infarction. Epidemiology. 2010;21(6):829-36.
21. Lercher P, Widmann U, Kofler W, editors. Transportation noise and blood pressure: the importance of modifying factors. Proceedings of the 29th International Congress and Exhibition on Noise Control Engineering (Cassereau D, ed). InterNoise; 2000.
22. Christensen JS, Raaschou-Nielsen O, Tjønneland A, Nordsborg RB, Jensen SS, Sørensen TI et al. Long-term exposure to residential traffic noise and changes in body weight and waist circumference: A cohort study. Environmental research. 2015;143:154-61.
23. Pyko A, Eriksson C, Oftedal B, Hilding A, Östenson C-G, Krog NH et al. Exposure to traffic noise and markers of obesity. Occupational and Environmental Medicine. 2015;72(8):594-601.
24. Dratva J, Phuleria HC, Foraster M, Gaspoz J-M, Keidel D, Künzli N et al. Transportation noise and blood pressure in a population-based sample of adults. Environmental health perspectives. 2012;120(1):50.
25. Seidler A, Wagner M, Schubert M, Dröge P, Römer K, Pons-Kühnemann J et al. Aircraft, road and railway traffic noise as risk factors for heart failure and hypertensive heart disease—A case-control study based on secondary data. International Journal of Hygiene and Environmental Health. 2016;219(8):749-58. doi:http://dx.doi.org/10.1016/j.ijheh.2016.09.012.
26. Basner M, Samel A, Isermann U. Aircraft noise effects on sleep: Application of the results of a large polysomnographic field study. Journal of the Acoustical Society of America. 2006;119(5):2772-84.
27. Basner M, Muller U, Elmenhorst EM. Single and combined effects of air, road, and rail traffic noise on sleep and recuperation. Sleep. 2011;34(1):11-23.
28. Brink M, Lercher P, Eisenmann A, Schierz C. Influence of slope of rise and event order of aircraft noise events on high resolution actimetry parameters. Somnologie. 2008;12:118-28.
29. Griefahn B, Marks A, Robens S. Noise emitted from road, rail and air traffic and their effects on sleep. Journal of Sound and Vibration. 2006;295(1-2):129-40.

30. Hofman WF, Kumar A, Tulen JHM. Cardiac reactivity to traffic noise during sleep in man. Journal of Sound and Vibration. 1995;179(4):577-89. doi:http://dx.doi.org/10.1006/jsvi.1995.0038.
31. Wunderli JM, Pieren R, Habermacher M, Vienneau D, Cajochen C, Probst-Hensch N et al. Intermittency ratio: A metric reflecting short-term temporal variations of transportation noise exposure. J Expos Sci Environ Epidemiol. 2016;26(6):575-85. doi:10.1038/jes.2015.56.
32. Spoerri A, Zwahlen M, Egger M, Bopp M. The Swiss National Cohort: a unique database for national and international researchers. International journal of public health. 2010;55(4):239-42.
33. Karipidis I, Vienneau D, Habermacher M, Köpflii M, Brink M, Probst-Hensch N et al. Reconstruction of historical noise exposure data for environmental epidemiology in Switzerland within the SiRENE project. Noise Mapping. 2014;1(1).
34. Heutschi K. SonRoad: New Swiss road traffic noise model. Acta Acustica united with Acustica. 2004;90(3):548-54.
35. OFPE. Modèle de calcul de bruit du traffic routier pour ordinateur. Les cahiers de l'environment. Berne: Office Fédéral de la Protection de l'Environment; 1987.
36. Thron T, Hecht M. The sonRAIL emission model for railway noise in Switzerland. Acta Acustica united with Acustica. 2010;96(5):873-83.
37. OFE. SEMIBEL Modèle suisse des émissions et des immissions pour le calcul du bruit des chemins de fer. Berne1990.
38. Thomann G, Buetikofer R. FLULA2: Aircraft noise prediction and measurements. The Journal of the Acoustical Society of America. 1999;105(2):1065-.
39. FOEN. PM10 and PM2.5 Ambient Concentrations in Switzerland. Modeling Results for 2005, 2010 and 2020. Berne Federal Office for the Environment; 2013.
40. Seidler A, Wagner M, Schubert M, Dröge P, Römer K, Pons-Kühnemann J et al. Aircraft, road and railway traffic noise as risk factors for heart failure and hypertensive heart disease—A case-control study based on secondary data. International Journal of Hygiene and Environmental Health. 2016.
41. Fuks KB, Weinmayr G, Basagaña X, Gruzieva O, Hampel R, Oftedal B et al. Long-term exposure to ambient air pollution and traffic noise and incident hypertension in seven cohorts of the European study of cohorts for air pollution effects (ESCAPE). Eur Soc Cardiology; 2016.
42. Röösli M, Mohler E, Frei P, Vienneau D. Noise-related sleep disturbances: Does gender matter? Noise and health. 2014;16(71):197.
43. Vienneau D, Perez L, Schindler C, Lieb C, Sommer H, Probst-Hensch N et al. Years of life lost and morbidity cases attributable to transportation noise and air pollution: A comparative health risk assessment for Switzerland in 2010. International journal of hygiene and environmental health. 2015;218(6):514-21.
44. Schäffer B, Plüss S, Thomann G. Estimating the model-specific uncertainty of aircraft noise calculations. Applied Acoustics. 2014;84:58-72. doi:http://dx.doi.org/10.1016/j.apacoust.2014.01.009.

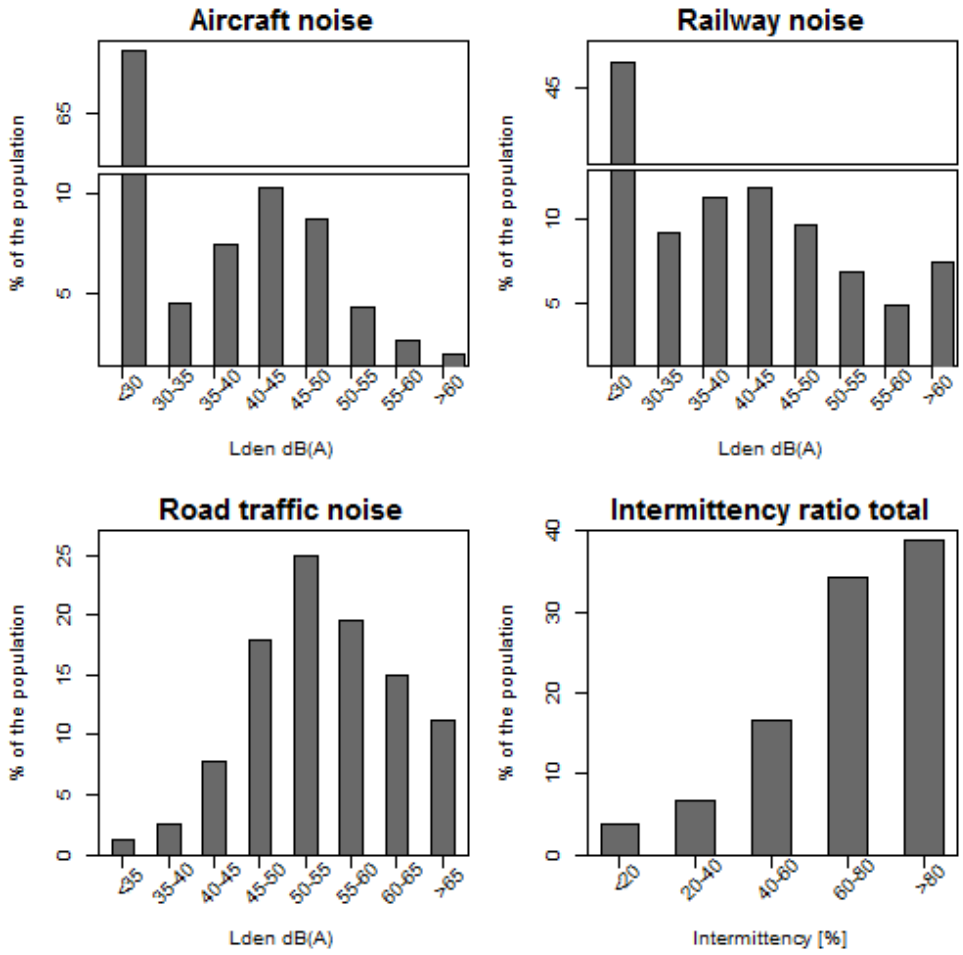

Fig 1: Distribution of noise exposure in the study population expressed as L_{den} in dB for aircraft, railway and road traffic noise levels and total IR at night in % (note different labelling for y-axis).

Table 1: Study population characteristics**.**

Characteristics at baseline	**Total cohort (n=4,415,206)**	**>55 dB Road traffic noise (n=2,018,090)**	**>55 dB Railway noise (n=459,408)**	**>55 dB Aircraft noise (n=115,463)**
Males (%)	47.9	47.9	48.2	48.7
Age: mean (SD)	52.4 (15.1)	52.4 (15.5)	52.1 (15.3)	51.1 (14.1)
Education level (%)				
• Compulsory education or less	23.8	26.7	27.9	20.7
• Upper secondary level education	51.6	49.9	50.5	51.7
• Tertiary level education	22.2	20.4	18.6	24.8
• Not known	2.2	2.8	2.9	2.6
Civil status (%)				
• Single	13.9	15.5	14.3	12.5
• Married	69.6	66.4	68.1	71.2
• Widowed	7.9	8.5	8.2	6.0
• Divorced	8.4	9.4	9.2	10.2
Socio-economic position (%)				
• low	50.4	54.5	61.0	35.2
• high	49.6	45.5	39.0	64.8
Mother tongue (%)				
• German and Rhaeto-Romansch	64.7	58.2	60.0	52.9
• French	19.4	22.6	19.5	28.4
• Italian	7.3	8.6	8.8	5.1
• Other	8.4	10.5	11.5	13.5
Nationality				
• Swiss	82.2	77.9	76.2	73.9
• Rest of Europe (inclusive ex-USSR)	15.9	19.5	21.3	22.2
• Other /unknown	1.9	2.5	2.4	3.9
NO_2 concentration µg/m3: mean (SD)	20.4 (6.2)	22.1 (6.6)	21.6 (5.2)	24.3 (4.6)

Table 2: Hazard Ratios (HR) and 95% confidence intervals (CI) per 10 dB increase in noise levels for cardiovascular causes of death.

Outcome	ICD-10 code	N Cases	HR L_{den}(Road) (95% CI)		HR L_{den}(Rail) (95% CI)		HR L_{den}(Air) (95% CI)	
			Crude	Adjusted	Crude	Adjusted	Crude	Adjusted
CVD	I00-I99	142955 (37.4%[a])	**1.012 (1.006-1.019)**	**1.025 (1.018-1.032)**	**1.006 (1.001-1.010)**	**1.005 (1.000-1.010)**	0.949 (0.942-0.956)	0.994 (0.985-1.002)
BP	I10-I15	13549 (9.4%[b])	1.02 (0.999-1.041)	**1.053 (1.030-1.075)**	1.008 (0.993-1.023)	1.011 (0.995-1.027)	0.930 (0.907-0.954)	1.012 (0.985-1.039)
IHD	I20-I25	60261 (42.2% [b])	1.008 (0.997-1.018)	**1.023 (1.012-1.034)**	**1.019 (1.011-1.026)**	**1.012 (1.005-1.020)**	0.958 (0.947-0.969)	0.991 (0.978-1.003)
MI	I21-I22	19313 (13.5% [b])	**1.038 (1.020-1.056)**	**1.040 (1.021-1.059)**	**1.023 (1.011-1.036)**	**1.020 (1.007-1.033)**	0.971 (0.952-0.990)	**1.027 (1.006-1.049)**
Heart failure	I50	12345 (8.6% [b])	1.021 (0.999-1.043)	**1.051 (1.027-1.074)**	0.964 (0.948-0.980)	0.997 (0.980-1.014)	0.907 (0.882-0.932)	**1.056 (1.028-1.085)**
Stroke	I60-I64	22377 (15.7% [b])	1.005 (0.989-1.022)	1.011 (0.993-1.028)	1.000 (0.988-1.012)	0.995 (0.983-1.008)	0.981 (0.963-0.999)	1.013 (0.993-1.033)
Hemorrhagic stroke	I60-I62	5354 (3.7% [b])	1.029 (0.995-1.063)	1.004 (0.968-1.040)	**1.028 (1.004-1.051)**	1.020 (0.996-1.045)	0.990 (0.955-1.026)	0.991 (0.951-1.032)
Ischemic stroke	I63	2991 (2.1% [b])	1.037 (0.992-1.083)	**1.050 (1.002-1.099)**	0.998 (0.966-1.031)	0.989 (0.956-1.023)	**1.052 (1.006-1.099)**	**1.074 (1.020-1.127)**

[a] % of total causes of death, [b]% of all CVD causes of death, Multipollutant models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO_2 exposure.

Table 3: Hazard Ratios and 95% confidence intervals (CI) for cardiovascular causes of death and categorical IR night by quintiles.

	1st quintile (reference)	**2nd quintile (IR night; 53.5-69.4%)**	**3rd quintile (IR night; 69.5-79.5%)**	**4th quintile (IR night; 79.6-87.3%)**	**5th quintile (IR night; 87.4-100%)**	**p-value for IR[a]**
CVD	1	1.001 (0.984-1.018)	**1.019 (1.002-1.037)**	**1.021 (1.003-1.038)**	0.994 (0.976-1.012)	0.003
BP	1	1.018 (0.992-1.045)	1.022 (0.995-1.049)	**1.033 (1.006-1.061)**	1.01 (0.982-1.039)	0.311
IHD	1	0.982 (0.94-1.025)	1.036 (0.993-1.082)	1.024 (0.98-1.069)	1.003 (0.958-1.05)	0.148
MI	1	1.019 (0.963-1.077)	1.045 (0.988-1.105)	1.018 (0.962-1.078)	0.987 (0.93-1.048)	0.290
HF	1	1.007 (0.949-1.068)	1.004 (0.947-1.065)	0.986 (0.929-1.047)	0.954 (0.896-1.015)	0.370
Stroke	1	0.939 (0.861-1.025)	1.041 (0.955-1.135)	1.066 (0.977-1.163)	0.968 (0.881-1.063)	0.096
HS	1	1.023 (0.912-1.149)	1.044 (0.929-1.173)	1.008 (0.895-1.135)	0.926 (0.815-1.053)	0.024
IS	1	1.029 (0.982-1.077)	1.017 (0.971-1.066)	1.046 (0.999-1.097)	1.006 (0.957-1.057)	0.366

[a] based on a likelihood ratio of models with and without IR. Models without IR presented in Table 2.

Multipollutant models including L_{den}(Road), L_{den}(Rail) and L_{den}(Air) adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO_2 exposure.

Supplementary material

Transportation noise exposure and cardiovascular mortality: a nationwide cohort study from Switzerland.

Harris Héritier[1,2]; Danielle Vienneau[1,2]; Maria Foraster[1,2], Ikenna C. Eze[1,2], Emmanuel Schaffner[1,2] Laurie Thiesse[3], Franziska Ruzdik[3], Manuel Habermacher[4], Micha Köpfli[4], Reto Pieren[5], Mark Brink[6], Christian Cajochen[3], Jean Marc Wunderli[5], Nicole Probst-Hensch[1,2], Martin Röösli [1,2] for the SNC study group

1 Swiss Tropical and Public Health Institute, Basel, Switzerland

2 University of Basel, Basel, Switzerland

3 Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Basel, Switzerland

4 N-sphere AG, Zürich, Switzerland

5 Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Dubendorf, Switzerland.

6 Federal Office for the Environment, Bern, Switzerland

Content

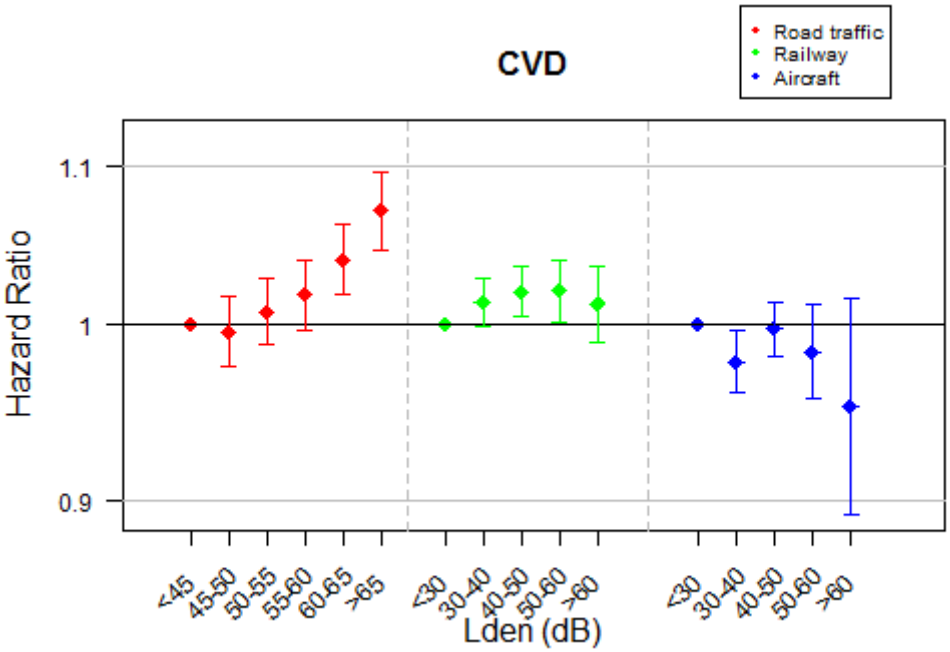

Fig 2: Categorical HRs for CVD for road traffic, railway, and aircraft noise. Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

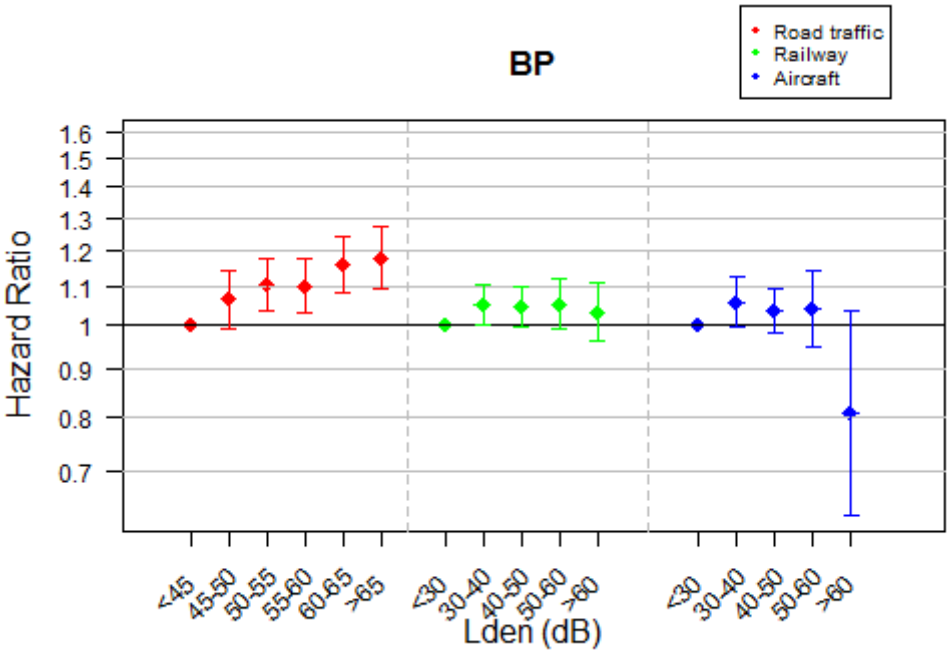

Fig 3: Categorical HRs for BP for road traffic, railway, and aircraft noise. Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

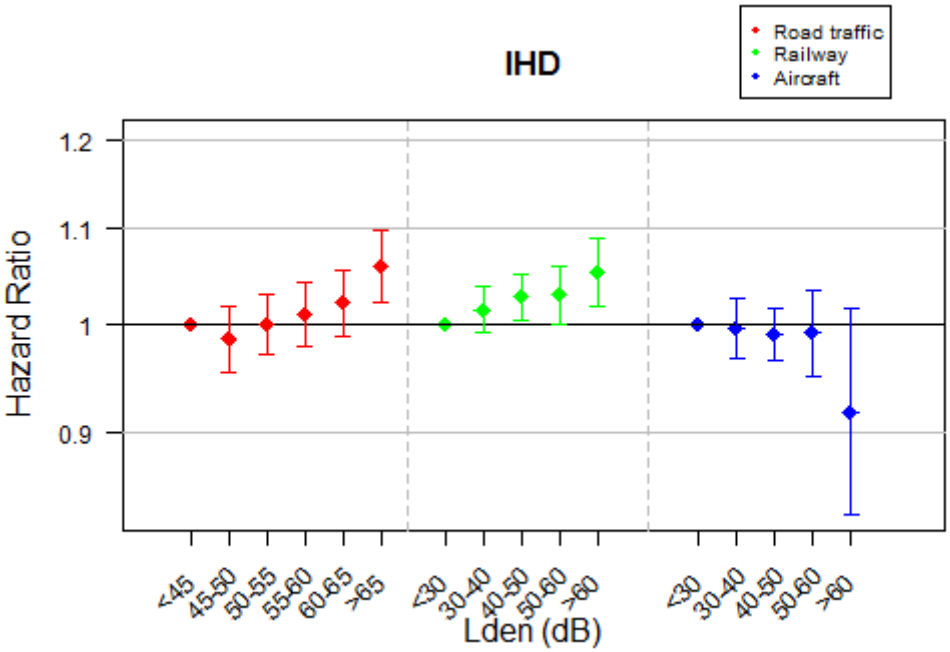

Fig 4: Categorical HRs for IHD for road traffic, railway, and aircraft noise. Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

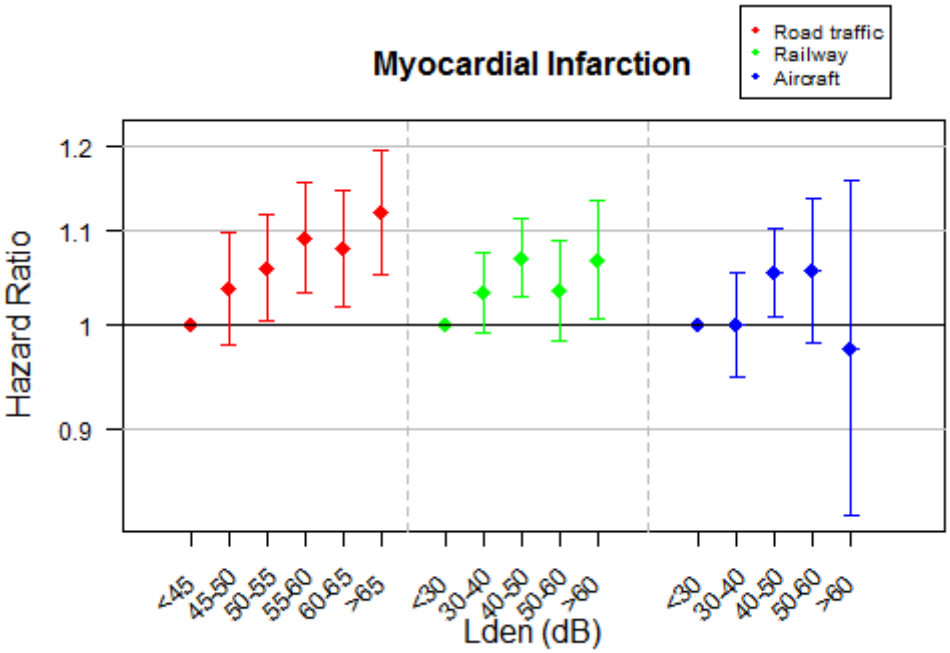

Fig 5: Categorical HRs for Myocardial Infarction for road traffic, railway, and aircraft noise. Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

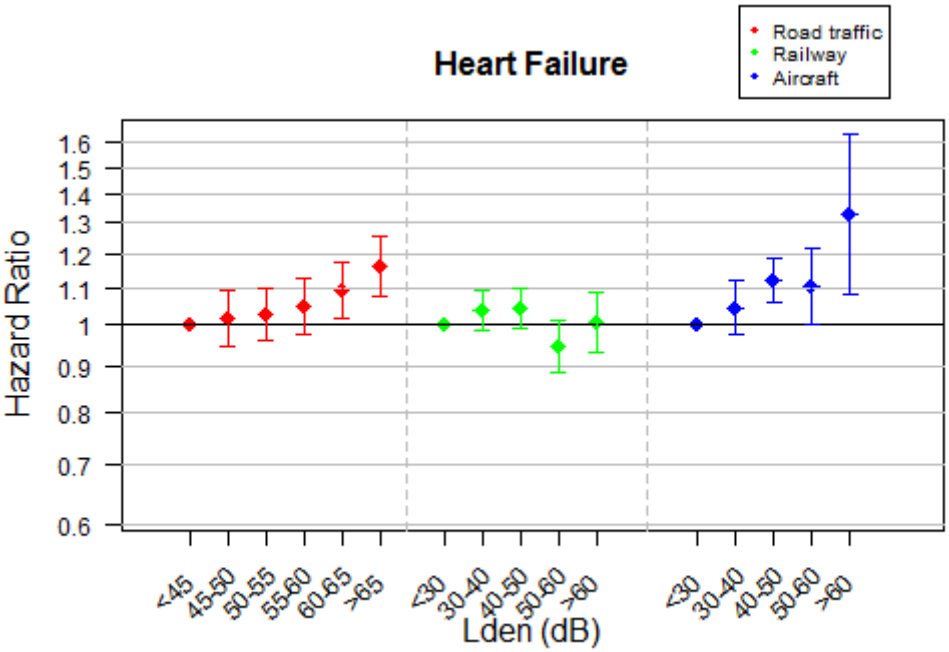

Fig 6: Categorical HRs for Heart failure for road traffic, railway, and aircraft noise. Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

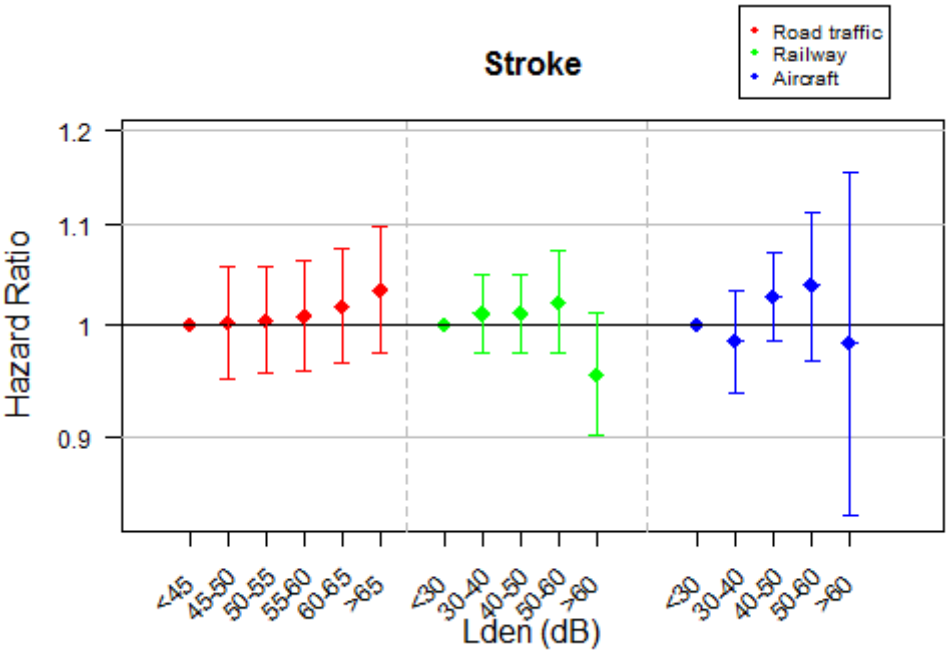

Fig 7: Categorical HRs for Stroke for road traffic, railway, and aircraft noise. Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

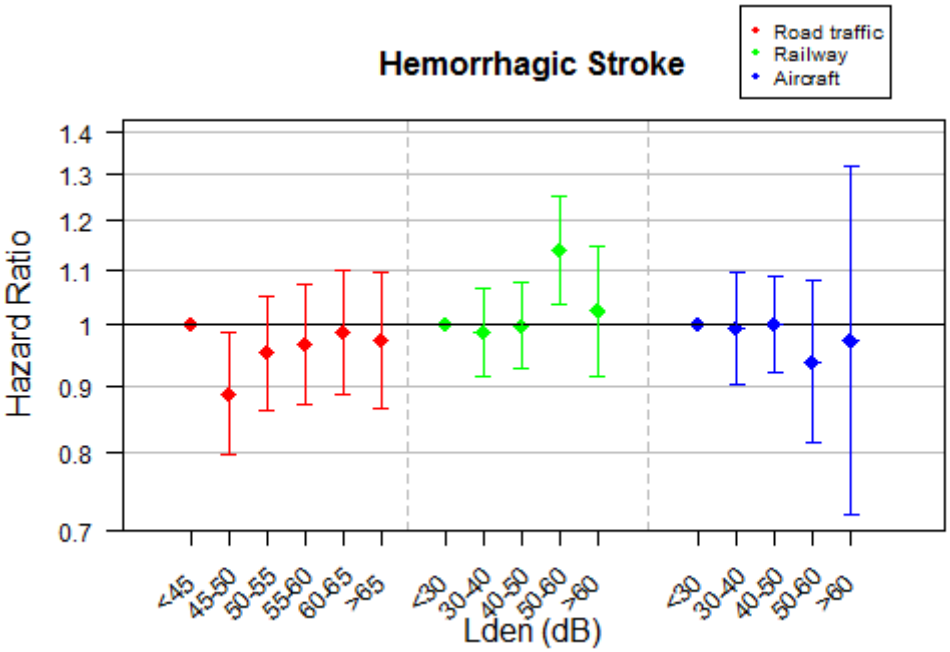

Fig 8: Categorical HRs for Hemorrhagic stroke for road traffic, railway, and aircraft noise. Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

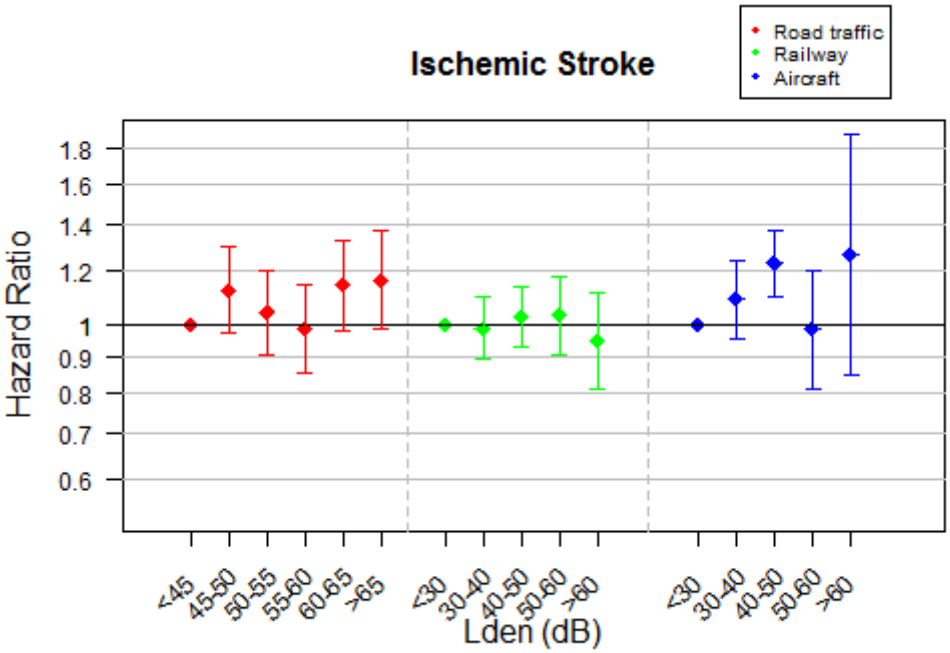

Fig 9: Categorical HRs for Ischemic stroke for road traffic, railway, and aircraft noise. Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

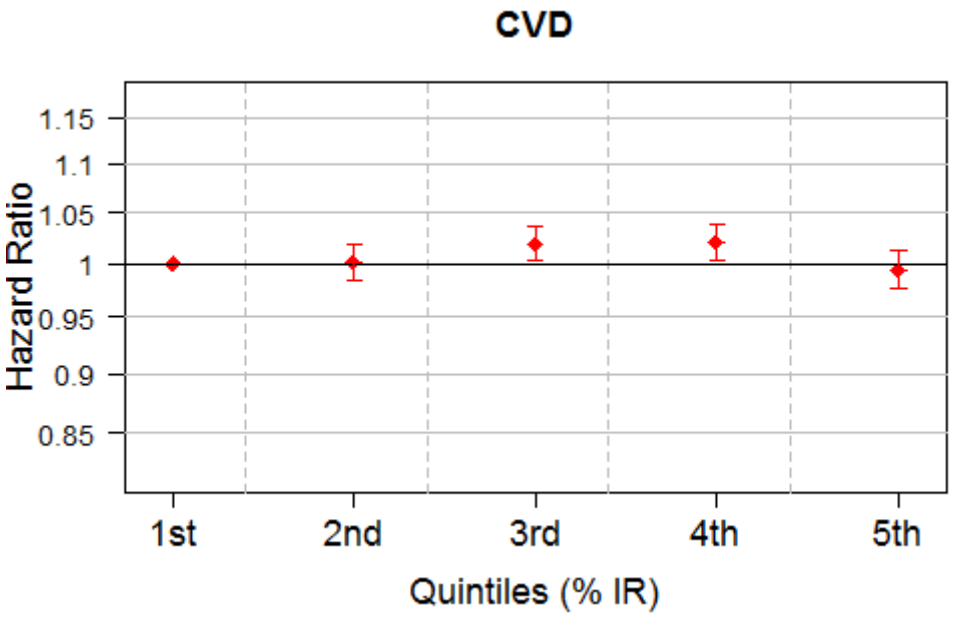

Fig 10: HRs by quintiles of IR night for CVD. Multipollutant models including L_{den}(Road), L_{den}(Rail) and L_{den}(Air) adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

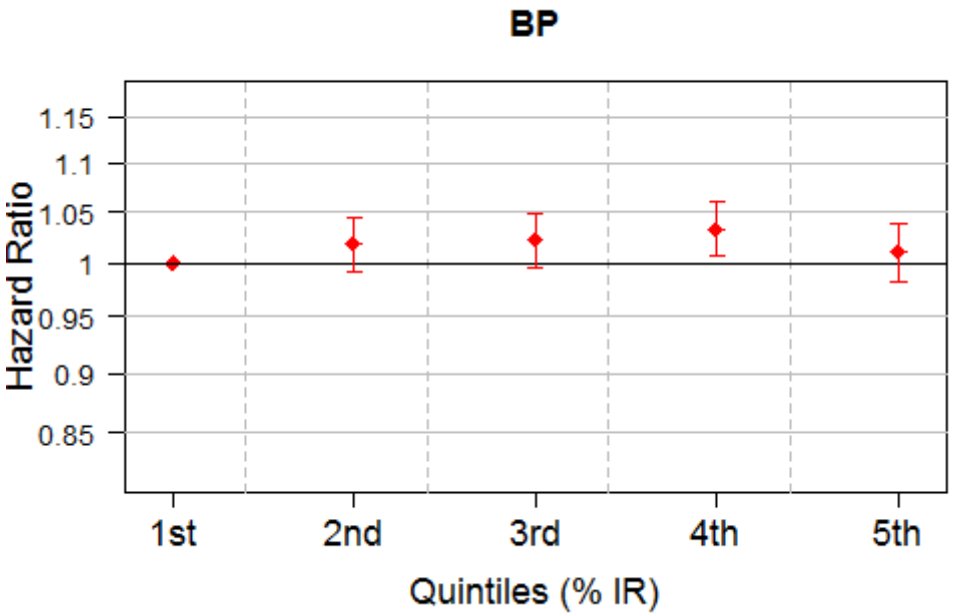

Fig 11: HRs by quintiles of IR night for BP. Multipollutant models including L_{den}(Road), L_{den}(Rail) and L_{den}(Air) adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

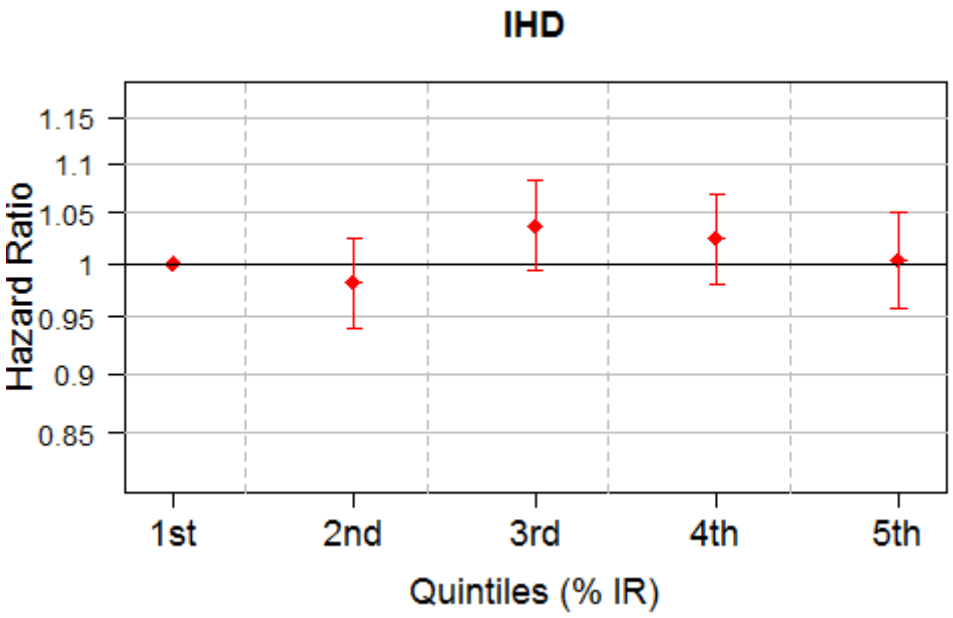

Fig 12: HRs by quintiles of IR night for IHD. Multipollutant models including L_{den}(Road), L_{den}(Rail) and L_{den}(Air) adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

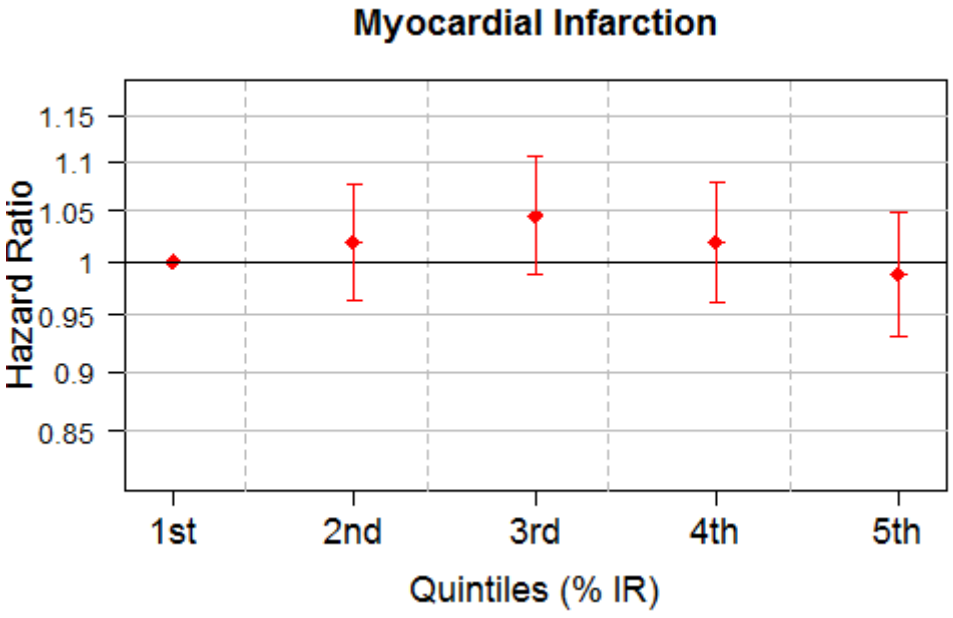

Fig 13: HRs by quintiles of IR night for Myocardial Infarction. Multipollutant models including L_{den}(Road), L_{den}(Rail) and L_{den}(Air) adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

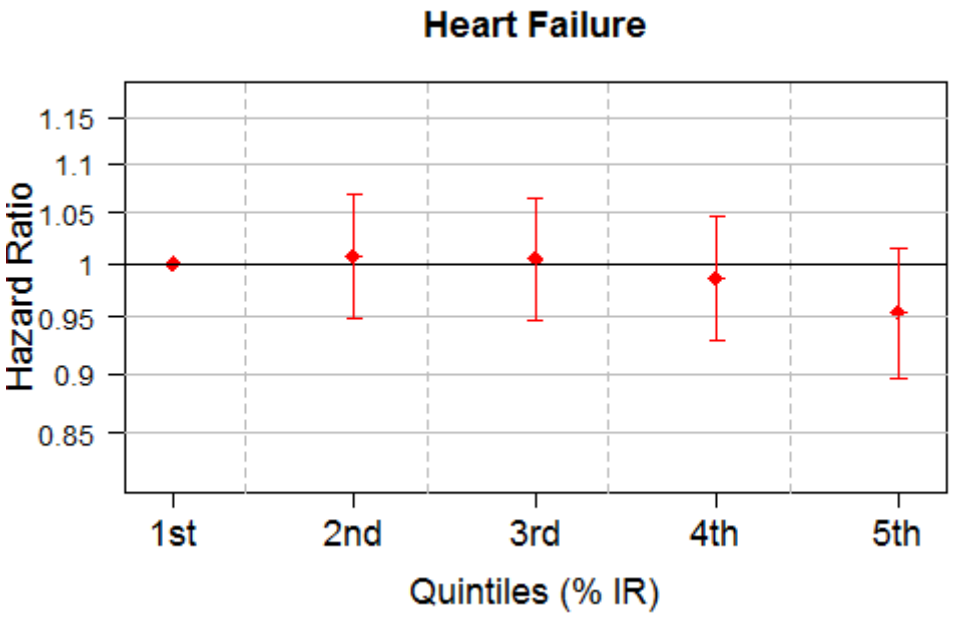

Fig 14: HRs by quintiles of IR night for Heart failure. Multipollutant models including L_{den}(Road), L_{den}(Rail) and L_{den}(Air) adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

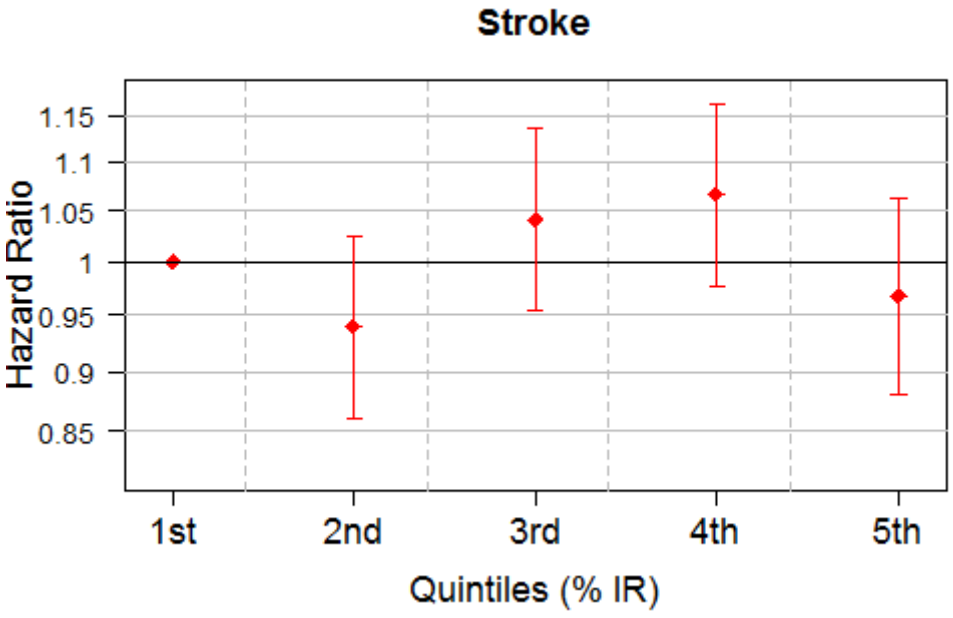

Fig 15: HRs by quintiles of IR night for Stroke. Multipollutant models including L_{den}(Road), L_{den}(Rail) and L_{den}(Air) adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

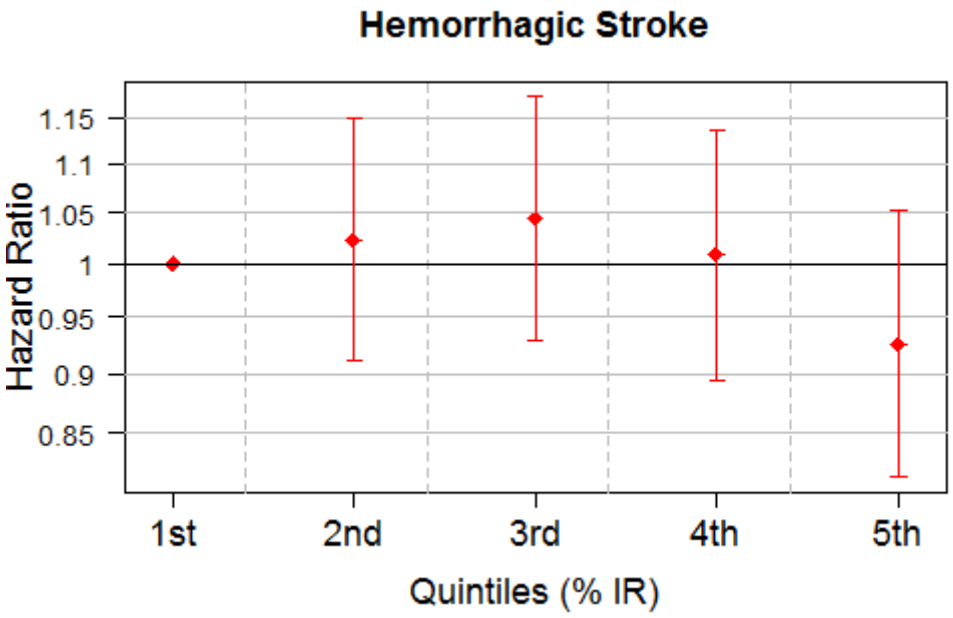

Fig 16: HRs by quintiles of IR night for Hemorrhagic stroke. Multipollutant models including L_{den}(Road), L_{den}(Rail) and L_{den}(Air) adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

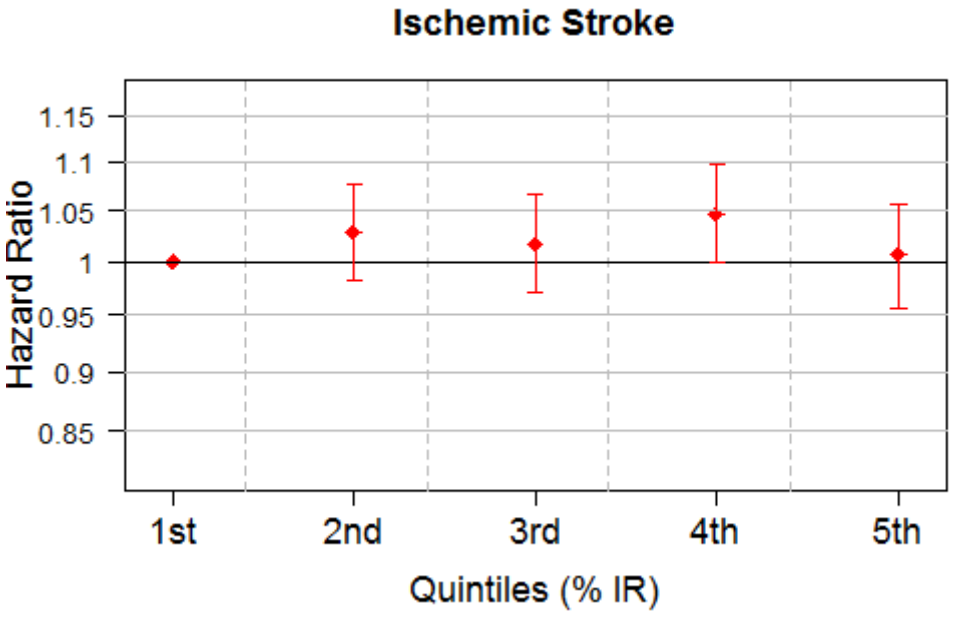

Fig 17: HRs by quintiles of IR night for Ischemic stroke. Multipollutant models including L_{den}(Road), L_{den}(Rail) and L_{den}(Air) adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure

Table 4: Stratified HRs for CVD for Road traffic, Railway and Aircraft noise.

Outcome	Strata	N cases	HR L_{den}(Road) (95% CI)	HR L_{den}(Rail) (95% CI)	HR L_{den}(Air) (95% CI)
CVD	<65y[a]	17431	1.067 (1.046-1.089)	1.02 (1.006-1.034)	0.997 (0.975-1.019)
	>=65y[a]	125524	1.019 (1.012-1.027)	1.003 (0.997-1.008)	0.993 (0.984-1.002)
	new building[b]	75025	1.028 (1.018-1.038)	1.004 (0.997-1.011)	0.999 (0.988-1.011)
	old building[b]	67930	1.021 (1.01-1.031)	1.004 (0.997-1.011)	0.989 (0.978-1.001)
	Movers[c]	65521	1.021 (1.011-1.032)	1.001 (0.993-1.008)	0.994 (0.982-1.007)
	non-movers[c]	73411	1.031 (1.021-1.041)	1.009 (1.002-1.016)	0.991 (0.979-1.002)
	low n events[d]	70330	1.009 (0.997-1.022)	1.003 (0.997-1.01)	0.989 (0.977-1.001)
	high n events[d]	72625	1.04 (1.029-1.051)	1.007 (0.999-1.015)	0.998 (0.986-1.009)
	Male	69915	1.043 (1.032-1.053)	1.013 (1.006-1.02)	0.987 (0.975-0.998)
	Female	73040	1.008 (0.998-1.018)	0.997 (0.991-1.004)	1 (0.989-1.012)
	low sep[e]	77763	1.015 (1.005-1.024)	1.003 (0.996-1.009)	0.984 (0.972-0.997)
	high sep[e]	65192	1.039 (1.028-1.05)	1.007 (0.999-1.014)	1.002 (0.991-1.013)
	Urban[f]	101283	1.03 (1.022-1.039)	1.008 (1.002-1.014)	0.992 (0.984-1.001)
	Rural[f]	40428	1.012 (1-1.025)	0.995 (0.985-1.005)	1.016 (0.991-1.042)

Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure. [a] Age split at 65 years, [b] building newer than 30y or renovated vs. older than 30y without renovation, [c] more than 5 years of residence vs. less than 5 years of residence, [d] number of events that exceed background level (median split), [e] socio-economic position (median split), [f] urbanity degree (urban vs. rural)

Table 5: Stratified HRs for BP for Road traffic, Railway and Aircraft noise.

Outcome	Strata	N cases	HR L_{den}(Road) (95% CI)	HR L_{den}(Rail) (95% CI)	HR L_{den}(Air) (95% CI)
BP	<65y[a]	1089	1.017 (0.939-1.101)	1.048 (0.992-1.106)	1.091 (1.003-1.187)
	>=65y[a]	12460	1.057 (1.033-1.083)	1.008 (0.991-1.025)	1.004 (0.976-1.033)
	new building[b]	6894	1.042 (1.01-1.076)	1.016 (0.993-1.039)	1.042 (1.003-1.084)
	old building[b]	6655	1.068 (1.034-1.104)	1.008 (0.985-1.031)	0.986 (0.949-1.025)
	Movers[c]	6171	1.036 (1.002-1.072)	1.008 (0.984-1.032)	1.015 (0.975-1.057)
	non-movers[c]	7002	1.07 (1.037-1.104)	1.014 (0.992-1.037)	1.002 (0.964-1.041)
	low n events[d]	6724	1.036 (0.996-1.078)	1.002 (0.982-1.023)	0.98 (0.941-1.02)
	high n events[d]	6825	1.094 (1.056-1.133)	1.018 (0.992-1.045)	1.038 (1-1.078)
	Male	5023	1.069 (1.03-1.109)	1.027 (1-1.054)	1.045 (1-1.091)
	Female	8526	1.045 (1.016-1.075)	1.002 (0.982-1.022)	0.993 (0.959-1.028)
	low sep[e]	7455	1.043 (1.013-1.075)	1 (0.979-1.021)	1.008 (0.967-1.051)
	high sep[e]	6094	1.069 (1.032-1.107)	1.026 (1.001-1.051)	1.016 (0.98-1.054)
	Urban[f]	9288	1.062 (1.032-1.093)	1.015 (0.996-1.034)	1.011 (0.982-1.041)
	Rural[f]	4125	1.028 (0.99-1.068)	0.987 (0.956-1.019)	1.032 (0.955-1.114)

Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure. [a] Age split at 65 years, [b] building newer than 30y or renovated vs. older than 30y without renovation, [c] more than 5 years of residence vs. less than 5 years of residence, [d] number of events that exceed background level (median split), [e] socio-economic position (median split), [f] urbanity degree (urban vs. rural)

Table 6: Stratified HRs for MI for Road traffic, Railway and Aircraft noise.

Outcome	Strata	N cases	HR L_{den}(Road) (95% CI)	HR L_{den}(Rail) (95% CI)	HR L_{den}(Air) (95% CI)
MI	<65y[a]	4217	1.081 (1.038-1.125)	1.027 (0.999-1.056)	1.039 (0.995-1.085)
	>=65y[a]	15096	1.029 (1.008-1.052)	1.018 (1.003-1.033)	1.025 (1-1.05)
	new building[b]	10398	1.046 (1.02-1.073)	1.018 (1-1.037)	1.037 (1.006-1.069)
	old building[b]	8915	1.034 (1.005-1.063)	1.018 (0.999-1.038)	1.016 (0.985-1.048)
	Movers[c]	8744	1.038 (1.009-1.067)	1.012 (0.992-1.032)	1.041 (1.008-1.075)
	non-movers[c]	9968	1.049 (1.022-1.077)	1.023 (1.004-1.042)	1.012 (0.982-1.044)
	low n events[d]	9510	1.04 (1.005-1.075)	1.023 (1.005-1.04)	1.005 (0.973-1.037)
	high n events[d]	9803	1.044 (1.014-1.075)	1.008 (0.987-1.031)	1.045 (1.015-1.077)
	Male	11451	1.049 (1.024-1.075)	1.017 (1-1.035)	1.026 (0.998-1.054)
	Female	7862	1.028 (0.998-1.059)	1.024 (1.003-1.045)	1.031 (0.996-1.066)
	low sep[e]	10737	1.041 (1.016-1.068)	1.019 (1.002-1.037)	1.016 (0.983-1.049)
	high sep[e]	8576	1.038 (1.008-1.069)	1.021 (1-1.042)	1.039 (1.009-1.069)
	Urban[f]	13684	1.043 (1.019-1.068)	1.017 (1.001-1.032)	1.03 (1.007-1.054)
	Rural[f]	5485	1.036 (1.002-1.07)	1.026 (0.999-1.054)	0.99 (0.926-1.058)

Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure. [a] Age split at 65 years, [b] building newer than 30y or renovated vs. older than 30y without renovation, [c] more than 5 years of residence vs. less than 5 years of residence, [d] number of events that exceed background level (median split), [e] socio-economic position (median split), [f] urbanity degree (urban vs. rural)

Table 7: Stratified HRs for heart failure for Road traffic, Railway and Aircraft noise.

Outcome	Strata	N cases	HR L_{den}(Road) (95% CI)	HR L_{den}(Rail) (95% CI)	HR L_{den}(Air) (95% CI)
HF	<65y[a]	478	1.206 (1.069-1.361)	0.911 (0.833-0.996)	1.199 (1.067-1.348)
	>=65y[a]	11867	1.046 (1.021-1.071)	1.001 (0.984-1.018)	1.05 (1.019-1.081)
	new building[b]	6146	1.058 (1.023-1.093)	0.998 (0.974-1.022)	1.04 (0.996-1.086)
	old building[b]	6199	1.043 (1.009-1.079)	0.992 (0.969-1.017)	1.061 (1.021-1.103)
	Movers[c]	5681	1.021 (0.986-1.057)	0.988 (0.963-1.013)	1.069 (1.025-1.115)
	non-movers[c]	6344	1.085 (1.05-1.121)	1.007 (0.983-1.031)	1.049 (1.007-1.092)
	low n events[d]	5977	1.021 (0.98-1.064)	1.008 (0.986-1.031)	1.052 (1.008-1.098)
	high n events[d]	6368	1.058 (1.02-1.097)	0.992 (0.964-1.019)	1.067 (1.026-1.109)
	Male	5088	1.103 (1.064-1.144)	0.997 (0.971-1.024)	1.062 (1.016-1.11)
	Female	7257	1.017 (0.986-1.048)	0.997 (0.975-1.019)	1.054 (1.015-1.094)
	low sep[e]	7247	1.025 (0.994-1.056)	0.981 (0.96-1.003)	1.091 (1.047-1.137)
	high sep[e]	5098	1.097 (1.056-1.139)	1.019 (0.992-1.047)	1.033 (0.993-1.075)
	Urban[f]	8302	1.079 (1.046-1.112)	1.003 (0.983-1.023)	1.048 (1.016-1.081)
	Rural[f]	3927	1.01 (0.972-1.049)	0.987 (0.954-1.02)	1.143 (1.058-1.234)

Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure. [a] Age split at 65 years, [b] building newer than 30y or renovated vs. older than 30y without renovation, [c] more than 5 years of residence vs. less than 5 years of residence, [d] number of events that exceed background level (median split), [e] socio-economic position (median split), [f] urbanity degree (urban vs. rural)

Table 8: Stratified HRs for ischemic stroke for Road traffic, Railway and Aircraft noise.

Outcome	Strata	N cases	HR L_{den}(Road) (95% CI)	HR L_{den}(Rail) (95% CI)	HR L_{den}(Air) (95% CI)
IS	<65y[a]	337	1.006 (0.872-1.16)	0.888 (0.796-0.991)	1.038 (0.892-1.208)
	>=65y[a]	2654	1.058 (1.005-1.114)	1.002 (0.967-1.039)	1.081 (1.022-1.145)
	new building[b]	1612	1.062 (0.995-1.134)	0.98 (0.935-1.027)	1.039 (0.962-1.121)
	old building[b]	1379	1.044 (0.972-1.123)	0.999 (0.951-1.05)	1.121 (1.039-1.208)
	Movers[c]	1368	1.086 (1.011-1.166)	0.98 (0.931-1.031)	1.11 (1.026-1.2)
	non-movers[c]	1564	1.024 (0.958-1.096)	0.996 (0.95-1.043)	1.047 (0.972-1.129)
	low n events[d]	1440	1.008 (0.924-1.099)	0.991 (0.947-1.037)	1.065 (0.984-1.152)
	high n events[d]	1551	1.073 (0.997-1.155)	0.99 (0.937-1.046)	1.089 (1.012-1.172)
	Male	1435	1.05 (0.98-1.126)	0.967 (0.92-1.017)	1.081 (1.001-1.167)
	Female	1556	1.052 (0.984-1.126)	1.009 (0.964-1.057)	1.071 (0.994-1.154)
	low sep[e]	1549	1.041 (0.975-1.112)	1.014 (0.97-1.061)	1.077 (0.991-1.17)
	high sep[e]	1442	1.07 (0.996-1.149)	0.956 (0.907-1.008)	1.071 (0.998-1.148)
	Urban[f]	2181	1.051 (0.992-1.115)	0.997 (0.959-1.037)	1.084 (1.024-1.147)
	Rural[f]	780	1.055 (0.967-1.151)	0.969 (0.9-1.043)	1.038 (0.874-1.232)

Multipollutants models adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and NO2 exposure. [a] Age split at 65 years, [b] building newer than 30y or renovated vs. older than 30y without renovation, [c] more than 5 years of residence vs. less than 5 years of residence, [d] number of events that exceed background level (median split), [e] socio-economic position (median split), [f] urbanity degree (urban vs. rural)

6. Paper 3

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Diurnal variability of transportation noise exposure and cardiovascular mortality: a nationwide cohort study from Switzerland

Harris Héritier[1,2]; Danielle Vienneau[1,2]; Maria Foraster[1,2], Ikenna C. Eze[1,2], Emmanuel Schaffner[1,2] Laurie Thiesse[3], Franziska Rudzik[3], Manuel Habermacher[4], Micha Köpfli[4], Reto Pieren[5], Arno Schmidt-Trucksäss[6], Mark Brink[6], Christian Cajochen[3], Jean Marc Wunderli[5], Nicole Probst-Hensch[1,2], Martin Röösli [1,2] for the SNC study group

[1] Swiss Tropical and Public Health Institute, Basel, Switzerland

[2] University of Basel, Basel, Switzerland

[3] Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Basel, Switzerland

[4] N-sphere AG, Zürich, Switzerland

[5] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Dubendorf, Switzerland.

[6] Department of Sport, Exercise and Health, Division of Sports and Exercise Medicine, University of Basel, Basel, Switzerland

[7] Federal Office for the Environment, Bern, Switzerland

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Diurnal variability of transportation noise exposure and cardiovascular mortality: A nationwide cohort study from Switzerland

Harris Héritier[a,b,1], Danielle Vienneau[a,b,1], Maria Foraster[a,b], Ikenna C. Eze[a,b], Emmanuel Schaffner[a,b], Laurie Thiesse[c], Franziska Ruzdik[c], Manuel Habermacher[d], Micha Köpfli[d], Reto Pieren[e], Arno Schmidt-Trucksäss[f], Mark Brink[g], Christian Cajochen[c], Jean Marc Wunderli[e], Nicole Probst-Hensch[a,b], Martin Röösli[a,b,*], for the SNC study group

[a] Swiss Tropical and Public Health Institute, Socinstrasse 57,4051 Basel, Switzerland
[b] University of Basel, Petersplatz 1, 4001 Basel, Switzerland
[c] Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Wilhelm Klein-Strasse 27, 4012 Basel, Switzerland
[d] N-sphere AG, Räffelstrasse 29, 8045 Zürich, Switzerland
[e] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Überland Str. 129, 8600 Dübendorf, Switzerland
[f] Department of Sport, Exercise and Health, Division of Sports and Exercise Medicine, University of Basel, Birsstrasse 320, 4052 Basel, Switzerland
[g] Federal Office for The Environment, Papiermühlestrasse 172, 3063 Ittigen, Switzerland

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ABSTRACT

Background: Most epidemiological noise studies consider 24 h average noise exposure levels. Our aim was to exploratively analyze the impact of noise exposure at different time windows during day and night on cardiovascular mortality.

Methods: We generated Switzerland-wide exposure models for road traffic, railway and aircraft noise for different time windows for the year 2001. Combined noise source equivalent continuous sound levels (L_{eq}) for different time windows at the most exposed façade were assigned to each of the 4.41 million Swiss National Cohort adult participants. Follow-up period was from 2000 to 2008. Hazard ratios (HR) of noise effects on various cardiovascular primary causes of death were computed by Cox regression models adjusted for potential confounders and NO_2 levels.

Results: For most cardiovascular causes of death we obtained indications for a diurnal pattern. For ischemic heart disease the highest HR was observed for the core night hours from 01 h to 05 h (HR per standard deviation of L_{eq}: 1.025, 95% CI: 1.016–1.034) and lower HR for the daytime 07 h to 19 h (1.018 [1.009–1.028]). Heart failure and daytime L_{eq} yielded the highest HR (1.047 [1.027–1.068]).

Conclusion: For acute cardiovascular diseases, nocturnal intermittent noise exposure tended to be more relevant than daytime exposure, whereas it was the opposite for chronic conditions such as heart failure most strongly associated with continuous daytime noise. This suggests that for acute diseases sleep is an important mediator for health consequences of transportation noise.

1. Introduction

Transportation noise has been shown to affect cardiovascular health (Münzel et al., 2014). With regard to road traffic noise, a meta-analysis reported a risk increase of 8% (95% CI: 4–13%) for coronary heart diseases per 10 dB increase in noise levels (Babisch 2014). Also with regard to myocardial infarction the majority of studies published to date have reported significant positive associations with road traffic noise (Babisch et al., 2005; Sørensen, 2012; Willich et al., 2006). Meta-analyses focusing on aircraft noise reported pooled relative risk increases of 1.63 (95% CI: 1.14, 2.33) and 1.06 (95% CI: 1.04, 1.08) per 10 dB increase in noise levels for hypertension (Huang et al., 2015) and ischemic heart disease (Vienneau et al., 2015), respectively. This evidence is based on studies that used 24 h average noise level metrics such as $L_{eq,24}$ (defined as the energetic average of continuous measurements over a 24 h period) or L_{den} (defined as the weighted energetic average of $L_{eq,day}$, $L_{eq,evening}$ and $L_{eq,night}$ with a penalty of 5 dB and 10 dB applied respectively to the latter).

* Corresponding author at: Swiss Tropical and Public Health Institute, Socinstrasse 57, P.O. Box CH − 4002, Basel.
E-mail address: martin.roosli@swisstph.ch (M. Röösli).
[1] Both authors contributed equally.

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Using the exposure metric L_{den}, we found associations between different cardiovascular causes of death and noise exposure from road, railway and aircraft noise in a previous study of the Swiss National Cohort (SNC) using data from 2000 to 2008 (Heritier et al., 2017). For instance, increased risk of death from myocardial infarction (MI) per 10 dB increase in L_{den} of road, railway and aircraft noise exposure were 4.0% (95% CI: 2.1–5.9%), 2.0% (95% CI: 0.7–3.3%) and 2.7% (95% CI: 0.6–4.9%), respectively.

However, it has been hypothesized that the impact of exposure to transportation noise on cardiovascular health is at least in part mediated via sleep alterations (Münzel et al., 2014), and thus noise exposure during sleep may be more critical than during the day. In sleeping subjects, transportation noise can induce sleep disturbances (Frei et al., 2014), and short sleep duration is a predictor of coronary heart disease and stroke (Cappuccio et al., 2011). Further, noise during sleep has a direct impact on the cardiovascular system by an acute increase in heart rate and blood pressure (Jarup et al., 2007) as well as neurocortical arousals (Basner et al., 2011).

Despite this evidence, little is known from epidemiological research about cardiovascular effects of noise at different times of the day and night, particularly during the nighttime hours; likely due to the high correlations between the exposures. So far only a few studies report the difference between nighttime and daytime exposure (Dratva et al., 2012; Hansell et al., 2013; Jarup et al., 2007) and there is a lack of knowledge and considerable uncertainty about the most sensitive nighttime windows and their detrimental impact on cardiovascular health. Such knowledge, however, is needed for policy makers to create an effective regulatory framework for noise.

The aim of this paper was to use data from our previous study (Heritier et al., 2017) to explore the effects of transportation noise in different time windows during day and night on cardiovascular mortality. In particular, we aimed at providing exposure-response relationships between transportation noise and cardiovascular outcomes for specific time windows: 07–19 h, 19–23 h, 23–01 h, 01–05 h, 05–06 h, and 06–07 h.

2. Material and methods

2.1. Study population

The SNC links national census data with mortality and emigration records (Spoerri et al., 2010). The census data contains personal, household and building information. Date of death, is included in the mortality records. The data used in our study is based on the 4 December 2000 census and on mortality and emigration data for the period 5 December 2000–31 December 2008 and contains 7.28 million people. We excluded subjects below 30 years of age (n = 2.59 million), observations for which residential coordinates were missing (n = 0.19 million), subjects living in an institution (n = 0.25 million), and observations for which the cause of death was imputed (0.03 million) leaving 4.41 million observations for the analyzes.

The outcomes under investigation were primary causes of death from all cardiovascular diseases (CVD) (ICD-10: I00-I99), ischemic heart diseases (IHD) (ICD-10: I20-I25), stroke (ICD-10: I60-I64), myocardial infarction (MI) (ICD-10: I21-I22), heart failure (ICD-10: I50) and hypertensive disease related death (ICD-10: I10-I15, comprising primary hypertension, hypertensive heart diseases, hypertensive renal disease, and hypertensive heart and renal disease). For 37% of all stroke deaths we had information about type of stroke (ICD-10: I60-I64), hemorrhagic (ICD 10: I60-I62) or ischemic (ICD 10: I63).

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), we built a Swiss-wide noise exposure database for the reference year 2001 which includes the three major transportation noise sources in Switzerland (road traffic, railway and aircraft noise) in different time windows (for details see (Karipidis et al., 2014)). For these analyzes we calculated combined noise exposure ($L_{eq,Comb}$) from all three sources (i.e. the energetic sum of road, railway, and aircraft noise) for the following time windows: 07–19 h ($L_{eq,Comb}$(07-19)), 19–23 h ($L_{eq,Comb}$(19–23)), 23–01 h ($L_{eq,Comb}$(23-01)), 01–05 h ($L_{eq,Comb}$(01-05)), 05–06 h ($L_{eq,Comb}$(05-06)), and 06–07 h ($L_{eq,Comb}$(06-07)). The time windows in the night were selected to roughly represent the different aspects of sleep at the population level, which is falling asleep (23:00–01:00), persistent sleep (01:00–05:00) and wake up phase (05:00–06:00 and 06:00–07:00) (Tinguely et al., 2014).

To consider the different characteristics of the three transportation noise sources, we additionally calculated the intermittency ratio (IR) quantifying the contribution of individual noise events above the background level to the combined noise exposure (Wunderli et al., 2016). IR values range from 0% to 100%, where 0% IR means that no single event occurs 3 dB above background (i.e. continuous noise) and 100% IR means that all noise energy is produced by “individual” noise events. Background corresponds to hourly L_{eq}. We estimated IR from all transportation sources combined in the same time periods as the L_{eq}.

For each building and each floor in Switzerland, noise exposure was estimated at pre-defined façade points. For each façade point, we calculated the $L_{den,Comb}$ from all transportation sources combined. $L_{eq,Comb}$ and IR_{Comb} for the specific time ranges were assigned on the basis of the façade point with the highest $L_{den,Comb}$ value. Using the available geocodes for each SNC participant and the information about floor of residence, we linked participants to their respective façade points. If information on floor of residence was not available (n = 1,290,042), we assigned the noise estimates of the middle floor of the building.

2.3. Statistical analysis

We analyzed the data using the Cox proportional hazards model with age as the underlying time variable. Participants were followed until emigration, death or end of follow-up in 2008, whatever occurred first. For all time windows $L_{eq,Comb}$ values below 35 dB were set to 35 dB for the analyzes. In order to calculate comparable hazard ratios (HR) for different times of the day and night with different noise exposure distributions, the L_{eqs} for each time window of interest were standardized and HRs per one standard deviation (SD) of exposure reported. To adjust for the noise peaking characteristics, we included a categorical (quintile) term for IR_{Comb}(Night) (19:00–7:00) in all models based on previous findings, where nighttime IR was significantly associated with CVD in a bell shaped relationship with highest HR for the 3rd and 4th quintile (Heritier et al., 2017). Additional to age (timescale), the following confounders were included in the model: sex (female/male), neighborhood index of socio-economic position(Panczak et al., 2012) (tertiles low, medium, high), civil status (single, married, widowed, divorced), education level (compulsory education or less, upper secondary level education, tertiary level education, not known), average NO_2 concentration (estimated outdoor residential levels using PolluMap, a 100 × 100 m dispersion model for Switzerland for the year 2010 (FOEN, 2013)), mother tongue (German and Rhaeto-Romansch, French, Italian, other language) and nationality (Swiss, rest of Europe, rest of the world/unknown).

In addition to exploring linear relationships with standardized $L_{eq,Comb}$ at during different time windows, we also conducted categorical noise analyzes using absolute values of $L_{eq,Comb}$ in 5 dB categories to explore the form of the exposure response at different times of the day and night.

In a second analysis we evaluated the impact of IR in different time windows during day and night adjusted for $L_{den,Comb}$ and the other adjustment variables mentioned above. IR_{Comb} at different time

Table 1
Study population characteristics.

Characteristics at baseline	Study population (n = 4,415,206)
Males (%)	47.9
Age: mean (SD)	52.4 (15.1)
Education level (%)	
Compulsory education or less	23.8
Upper secondary level education	51.6
Tertiary level education	22.2
Not known	2.2
Civil status (%)	
Single	13.9
Married	69.6
Widowed	7.9
Divorced	8.4
Socio-economic position (%) (Panczak et al., 2012)	
low	33.3
medium	33.3
high	33.3
Mother tongue (%)	
German and Rhaeto-Romansch	64.7
French	19.4
Italian	7.3
Other	8.4
Nationality	
Swiss	82.2
Rest of Europe (inclusive ex-USSR)	15.9
Other/unknown	1.9
NO_2 concentration in µg/m^3: mean (SD)	20.4 (6.2)
Outcomes	
All cardiovascular diseases (CVD) (ICD-10: I00-99)	142,955 (37.4%[a])
Hypertensive diseases (ICD-10: I10-I15)	13,549 (9.4%[b])
Ischemic heart diseases (IHD) (ICD-10: I20-I25)	60,261 (42.2%[b])
Myocardial infarction (MI) (ICD-10: I21-I22)	19,313 (13.5%[b])
Heart failure (ICD-10: I50)	12,345 (8.6%[b])
Stroke (ICD-10: I60-I64)	22,377 (15.7%[b])
Hemorrhagic stroke (ICD-10: I60-I62)	5354 (3.7%[b])
Ischemic stroke (ICD-10: I63)	2991 (2.1%[b])

[a] % of total causes of death.
[b] % of all CVD causes of death.

windows was split into quintiles to visualize non-linear relationships.

All analyzes were conducted with the statistical software R and the package "survival" (R Development Core Team 2008; Therneau, 2000).

3. Results

3.1. Study population

The study sample amounted to 4.41 million observations, with 33.85 million person-years for the period 5th December 2000–31 st December 2008. Median (mean) time of follow-up was 8.07 (7.64) years. Characteristics of the study sample are displayed in Table 1.

The cohort contained 142,955 deaths from CVD, of which 42.2%, 15.7%, 9.4%, 8.6% were deaths from IHD, stroke, hypertensive disease and heart failure. Increase in crude incidence rates by increasing daytime and nighttime noise were observed for all outcomes except for stroke (eTable 1*)*.

3.2. Noise exposure

The percentage of the study population exposed to levels > 45 dB and > 55 dB was higher during daytime than nighttime and consequently the percentage of observations below 35 dB was highest during the core night from 01 h to 05 h (Table 2).

The distribution of transportation noise exposure $L_{eq,Comb}$ in the different time windows was slightly skewed as displayed in Fig. 1 and summarized in eTable 2.

Table 2
Percentage of the study population exposed to noise < 35 dB, > 45 dB and > 55dB.

Exposure variable	% of population < 35 dB	% of population with > 45 dB	% of population with > 55 dB
$L_{eq,Comb}$(07–19 h)	1.2	89.6	47.7
$L_{eq,Comb}$(19–23 h)	2.7	77.9	32.4
$L_{eq,Comb}$(23–01 h)	7.8	54.2	15.9
$L_{eq,Comb}$(01–05 h)	18.2	37.7	7.3
$L_{eq,Comb}$(05–06 h)	4.9	64.9	22.8
$L_{eq,Comb}$(06–07 h)	1.8	84.1	38.9

Spearman's rank correlations between the $L_{eq,Comb}$ for different time windows were high (≥0.94, eTable 3 *Supplementary material*), whereas Spearman's rank correlations between the IR_{Comb} for different time windows were somewhat lower (eTable 4 *Supplementary material*). Road traffic is the most prevalent transportation noise source and thus correlation between $L_{eq,Road}$ with $L_{eq,Comb}$ at different time windows was higher than corresponding correlations of $L_{eq,Rail}$ and $L_{eq,Air}$ with $L_{eq,Comb}$ (eTable 5). Correlation between $L_{eq,Comb}$ and IR_{Comb} at different time periods was generally low and did not exceed 0.4.

3.3. Diurnal effect pattern for $L_{eq,Comb}$

For IHD, we observed the highest HR for standardized noise exposure during the core night ($L_{eq,Comb}$(01–05 h)) and the lowest HR for the time window 06–07 h and during the day, although differences were not statistical significant (Fig. 2). The HR per standard deviation of $L_{eq,Comb}$ was 1.025 (95% CI: 1.016–1.034) for the core night, 1.017 (95% CI: 1.008–1.026) for the time window 06–07 h and 1.018 (95% CI: 1.009–1.028) for daytime exposure (eTable 6). A similar pattern was seen for all CVD deaths combined (eFigure 1). The diurnal pattern was not pronounced for MI (eFigure 1*)*. For heart failure, an inverse pattern was observed, the risk being lowest for $L_{eq,Comb}$(01–05 h) and highest during the day (07–19 h, HR = 1.047 [1.027–1.068]) (Fig. 2). For hypertensive disease highest HR was observed in the early morning (05–06 h) and lowest HR in the early night (23–01 h) (Fig. 2). For all stroke deaths, no overall association and no overall diurnal pattern was seen (Fig. 2). However, for hemorrhagic stroke, HRs tended to be slightly increased during night but not during the day (eFigure 1*)*. For ischemic stroke, HRs were highest during daytime and only slightly above unity in the core night (*eFigure 1*).

Non-standardized HR for each time window in absolute 5 dB $L_{eq,Comb}$ categories demonstrated a fairly linear exposure-response pattern for hypertensive disease, IHD and heart failure (Fig. 3), CVD and MI (eFigure 2). During the core night (01–05 h), HRs tended to start increasing from a lower level than during daytime. This pattern also held for heart failure (Fig. 3) where a different diurnal pattern was seen for the standardized exposure analysis.

3.4. Day-night pattern for the intermittency ratio (IR)

For IR_{Comb} during night (23:00–6:00) the categorical HRs followed a bell shaped curve with highest risk in the 3rd and 4th quintiles for cardiovascular disease (eFigure 3). For IR_{comb} during daytime windows the HR curves were somewhat flatter and skewed toward the 2nd quintile (eTable 7), which corresponds to a more continuous noise level. A similar pattern was observed for ischemic heart disease but an opposite pattern for hypertensive diseases (Fig. 4): noise events were relevant during daytime as whereas during core night (01:00–05:00), continuous noise (1st quintile) tended to be more detrimental. For heart failure highly intermittent noise (5th quintile) produced somewhat reduced HR except during the core night. Highly intermittent noise implies relatively few noise events with longer quiet phases between the

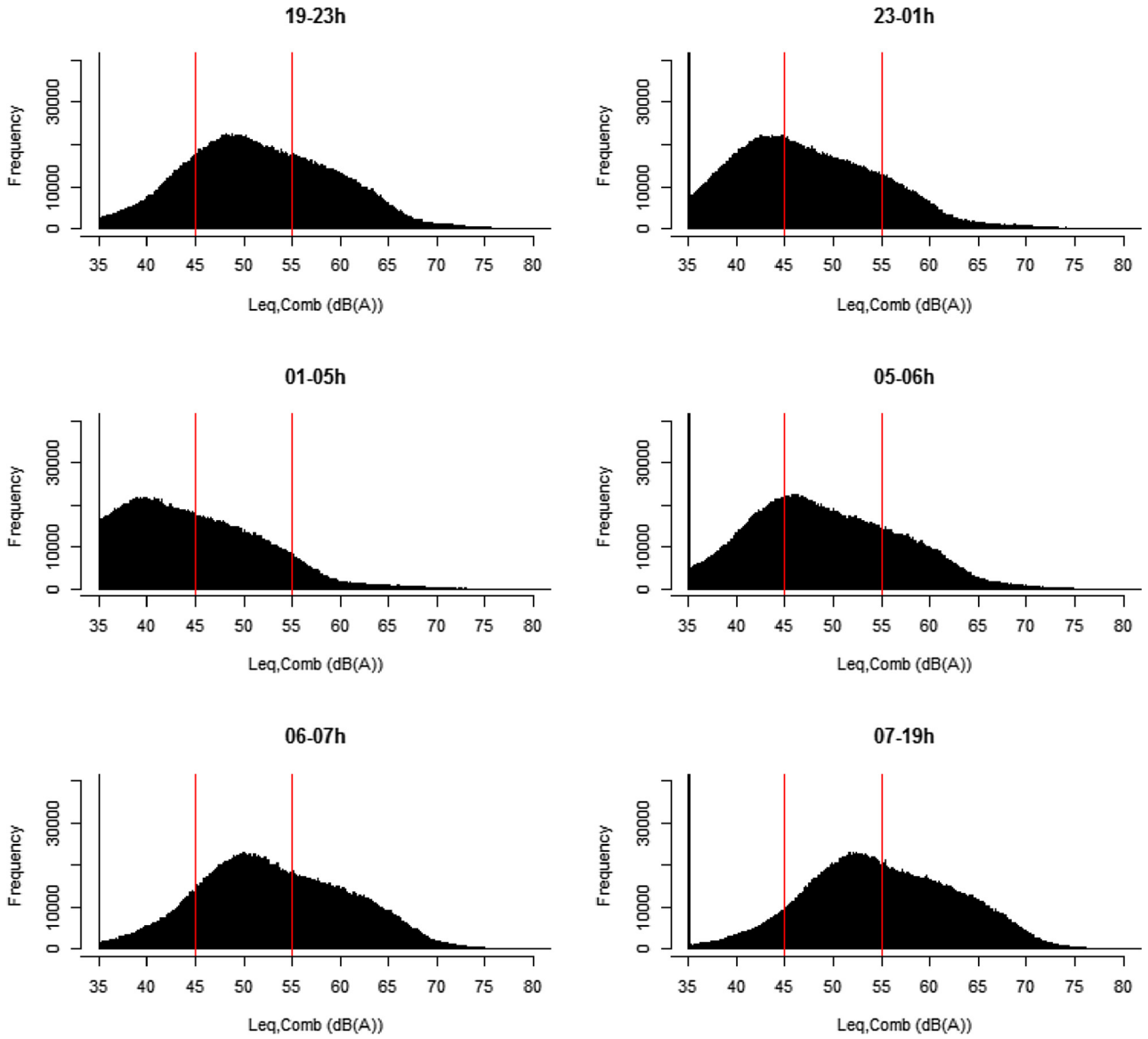

Fig. 1. Distribution of Leq,Comb(19–23 h), Leq,Comb(23–01 h), Leq,Comb(01–05 h), Leq,Comb(05–06 h), Leq,Comb(06–07 h), and Leq,Comb(07–19 h) in the study population. The vertical red lines show the 45 dB (left) and 55 dB (right) threshold. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

events.

4. Discussion

Similar to other studies, we previously observed in this SNC dataset associations between various cardiovascular causes and deaths and average noise exposure expressed as L_{den} (Heritier et al., 2017). In this follow-up paper we could investigate exposure effects from transportation noise in different time windows during day and night, which is useful information for an effective regulatory framework of noise. Our results point toward a stronger association between transportation noise exposure during the nighttime hours 23–06 h and acute heart diseases, (e.g. IHD) than for daytime. Thereby intermittent noise seems to be more critical than continuous noise exposure. This pattern tended to be reversed when considering chronic heart disease such as heart failure. For hypertensive disease, the diurnal pattern was less distinct with highest HRs in the early morning and during daytime. No association at all was found with stroke, although noise tended to be associated with ischemic stroke, mainly during daytime. During night, cardiovascular disease risk started to increase as low as 35 dB.

Different outcome related impacts of the timing of noise exposure may indicate that different mechanisms are at play in the aetiology of cardiovascular outcomes. With regard to IHD, our finding supports the hypothesis that noise associated sleep disturbances impair cardiovascular health in the long run. The core night contains more deep sleep than the early morning hours and thus disturbances from noise events during this time interval seem to be considerably critical (Webb and Agnew 1971). Sleep disturbances have been linked with higher risk for obesity and diabetes (Eze et al., 2017; Knutson and Van Cauter, 2008) which are known risk factor for IHD (Lavie et al., 2009; Wilson et al., 1998) and also recently found to be related to transportation noise exposure (Eze et al., 2017; Sörensen et al., 2013); thus further supporting the link between nighttime noise and this particular outcome. The onset of acute coronary events shows a strong morning peak during the time of sympathetic nervous system activation (Muller 1999) going along with endothelial dysfunction, arterial stiffening, platelet aggregation and thrombus formation (Chen and Yang 2015). Thus, noise exposure in the previous night may act as an additional stressor (Münzel et al., 2014) which triggers the onset of fatal events.

Interestingly, for heart failure an opposite diurnal pattern was seen

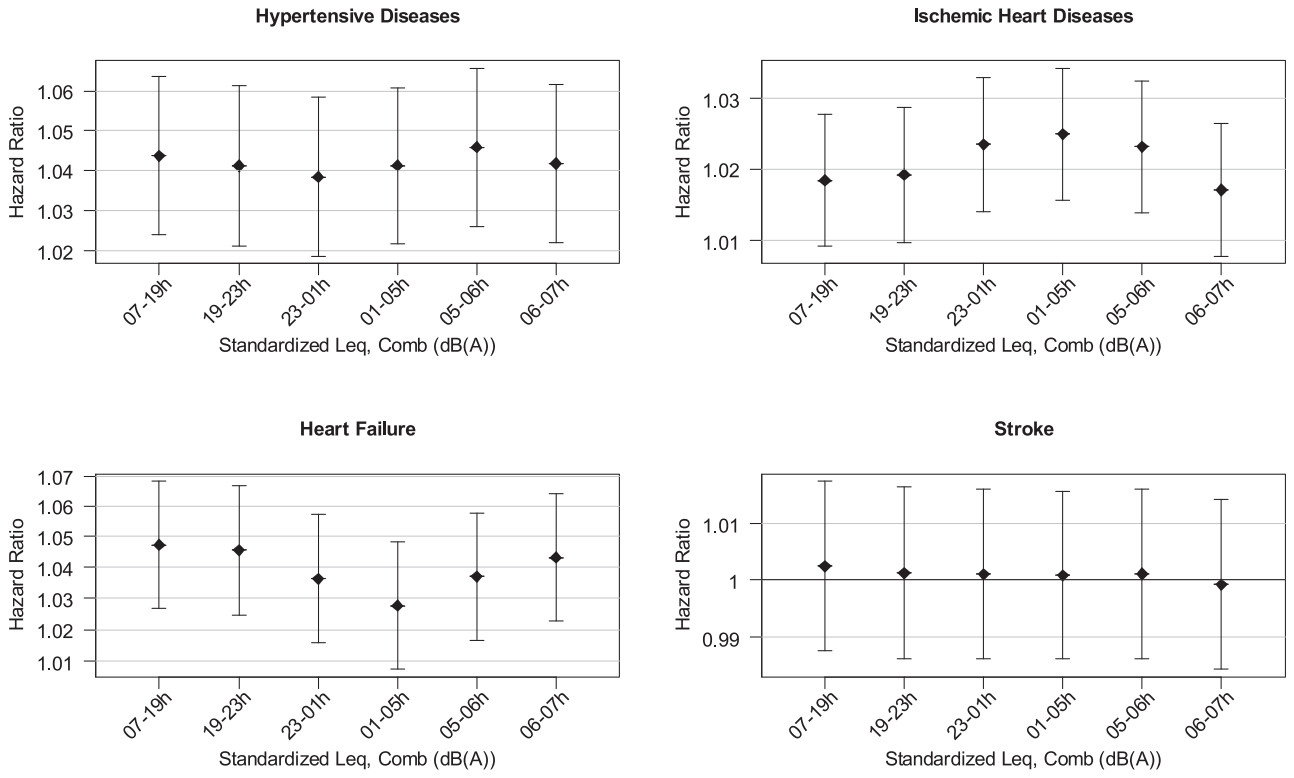

Fig. 2. Adjusted[a)] hazard ratio and 95% confidence intervals per increase of 1 standard deviation[b)] of standardized $L_{eq,Comb}$ in the different time windows for the following causes of death: Hypertensive diseases, IHD, heart failure and stroke. Note different y-axis for all the figures.
[a)] adjusted for age (timescale), sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality, NO_2 exposure and IR_{comb}(Night), [b)] SD is between 7.35 and 8.01 dB (see eTable 6)

compared to death due to IHD. Daytime noise and continuous noise exposure was most critical. Heart failure has only recently been linked to noise exposure (Correia et al., 2013; Seidler et al., 2016). Death by heart failure is caused in approximately 50% of cases by sudden cardiac death and in 35% by acute worsening of ischemia (Ørn and Dickstein, 2002). Deaths from sudden cardiac death occur most often during 6 am and 12 noon (Moser et al., 1994) and thus it is plausible that noise associated stress during daytime is a relevant trigger.

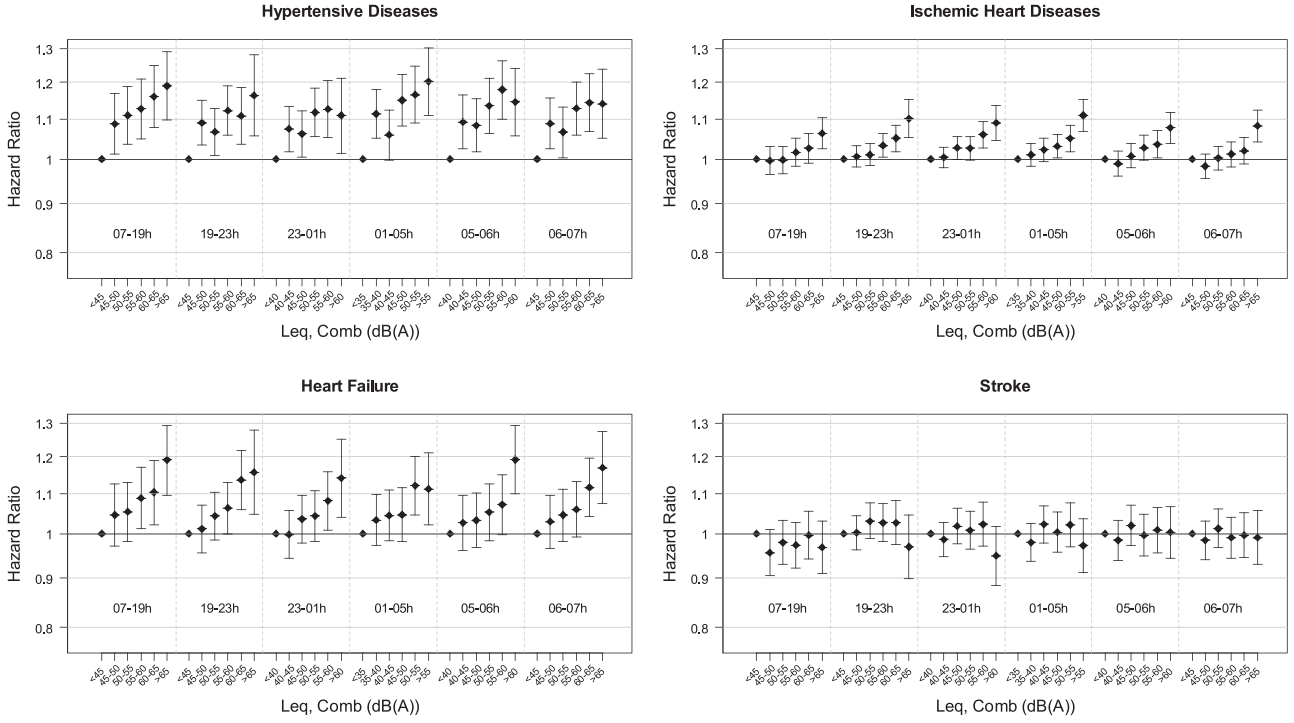

Fig. 3. Adjusted[a)] categorical HRs for specific time windows. Models for the following causes of death: Hypertensive diseases, IHD, heart failure and stroke.
[a)] adjusted for age (timescale), sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality, NO_2 exposure and IR_{comb}(Night)

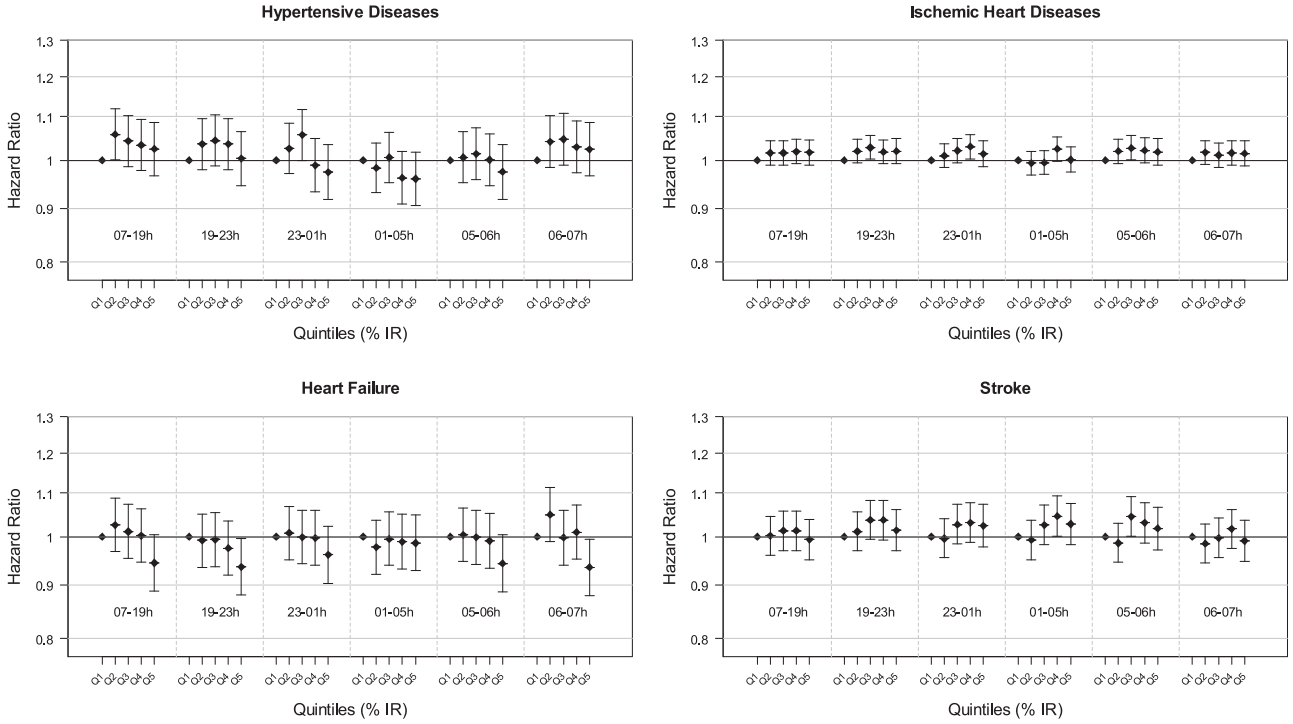

Fig. 4. Adjusted[a)] categorical Hazard ratio and 95% confidence intervals per quintiles of IR at different time windows for the following causes of death: Hypertensive diseases, IHD, heart failure and stroke.

[a)] adjusted for age (timescale), sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality, NO_2 exposure and L_{den}.

For hypertensive disease related mortality the diurnal pattern was most similar to heart failure but less distinct. Early morning hours and daytime exposure may be most critical. The start of activity phase in the early morning hours coincides with the diurnal peak levels of glucocorticoids (Dickmeis, 2009). Additional external noise stressors may thus affect body homeostasis with higher risk for a fatal event. Daytime noise exposure is expected to lead to vegetative arousals with increases in blood pressure and stress hormone levels, which may result arterial hypertension (Münzel et al., 2014). Our results suggest that for such arousals during the day noise events are more critical than continuous noise exposure levels. Conversely, during night continuous noise exposure might be most critical for hypertensive diseases.

Only a few epidemiological studies systematically addressed the different impacts of daytime and nighttime noise on cardiovascular outcomes. Halonen et al. (2015) found higher risk of cardiovascular hospital admissions in the elderly for road traffic noise during daytime. Since cardiovascular admissions comprised heart failure, this observation may be in line with our findings for heart failure related deaths during the day. Others observed stronger effects at night for hypertension and aircraft noise (Jarup et al., 2007) as well as for systolic blood pressure and railway noise (Dratva et al., 2012). However, the difference in exposure assessment of those studies limits comparison with ours. It is well possible that the exact timing of a fatal event also depends on the typical sleeping attitudes in a population. In Switzerland people wake up on average at 06:30 (IQR:06:00–07:00) (Tinguely et al., 2014). Critical time windows may be different for other populations with a different sleeping pattern.

A particular challenge for such an evaluation in observational research is the inherent high correlation between exposure levels at different time windows which limits causal inference and produces largely overlapping confidence intervals as shown in Fig. 2. Thus, chance cannot be completely ruled out as a potential explanation for the observed findings. However, HR of L_{eq} for IHD, hypertensive diseases and heart failure showed relative smooth cosine-shaped diurnal patterns, which speak against random fluctuations of the risk estimates.

By standardizing the noise exposure during different time windows we were able to directly compare the HRs. We also combined exposure from the three traffic sources road, railway and aircraft. This is expected to reduce within day correlation since the diurnal pattern is different for the three traffic sources (e.g. night curfews for aircraft, heavy nighttime railway traffic from freight trains). On the other hand, this may create confounding if one assumes different health effects for different noise characteristics emitted by different sources. To prevent, or at least minimize, such confounding we have additionally included in all our models the IR and could indeed demonstrate that – independent on the noise levels – IR is related to cardiovascular mortality. For IHD we found in tendency a pattern of larger effects for nighttime IR compared to daytime IR and a shift of the risk toward higher IR quintiles. This indicates that noise events may be more problematic for nighttime; whereas during daytime continuous noise with absence of quiet phases may be most detrimental.

We did not find an association between stroke and noise exposure. In line with a Danish cohort study (Sorensen et al., 2014) noise was not associated with hemorrhagic stroke, but some indication for an association with ischemic stroke was observed, in particular for daytime noise (HR:1.029, 95% CI: 0.988–1.072, eTable 6). However, information about type of stroke was only available for 37% of the 22,377 stroke deaths, which explains why the subgroup results for hemorrhagic and ischemic stroke do not add up to the results for all types of stroke, which also includes non-specified strokes (ICD-10: I64). In addition, stroke does not share a common etiology with the other CVD if no information about cardiac embolism is available (Arboix and Alió, 2012; Ferro, 2003). Therefore including stroke in the broader CVD category, as most often done in cardiovascular noise research, will have an impact on the strength of association for CVD in general. Indeed, in a sensitivity analysis excluding stroke from all CVD yielded a 0.2% increase of the HR for CVD per 10dB of $L_{eq,Comb}$(01–05 h).

4.1. Exposure misclassification

One might argue that the diurnal pattern of HRs throughout the different time windows may reflect exposure misclassification. During

the core night, the exposure assigned to the study population may be closer to the true exposure since people are at home, while during the day noise exposure is not relevant for the true exposure since people may be elsewhere for example at work. However, if there was only exposure misclassification at play, this would affect all estimates for all outcomes. This is not the case for hypertensive disease, ischemic heart disease and heart failure for which the estimates of the core night (01–05 h) and the day (07–19 h) have either the same or even a lower magnitude. This suggests that the observed pattern reflects a true underlying risk pattern and is not just caused by a common underlying bias.

4.2. Strength and limitations

The strengths of this study are the large study population and the long follow-up time providing a high number of cases. We developed an extensive and detailed noise exposure model, which allowed an individual exposure assessment at the address and floor level, which has rarely been done in previous large population studies. Detailed input data of the noise models such as traffic flow statistics could be used to calculate L_{eq} and IR with high temporal resolution. Potential selection bias is minimal in this nationwide study based on census data.

Though our models are adjusted for socioeconomic status and other demographic variables, we could not adjust for lifestyle and smoking as this information is not available in the SNC. We therefore cannot rule out that residual confounding of lifestyle may play a role for our analyses, although no indications for this were seen in a previous SNC noise study (Huss et al., 2010). Further, as in every study on long term noise exposure, exposure misclassification is unavoidable due to uncertainty in the input data (e.g., traffic information). A validation of the road traffic model using 123 weekly noise measurements demonstrated a good agreement with an average difference between modeling and measurements of +0.4 dB with an interquartile range (IQR) of −1.7 to 3.1 dB (Schlatter, 2017). Most relevant, even during night (23:00–7:00) when other sources of noise may have a larger impact on the measurements due to low traffic density, a good model performance was observed in the exposure range between 30 and 70 dB (mean deviance: 0.2 dB, IQR: −1.8 to 2.7 dB. Note that noise exposure was assigned based on estimates for the loudest outdoor façade point while no information was available regarding indoor noise levels. We did not have information on sleeping room orientation nor on factors relevant for the noise propagation into the residency such as sound proof window installation and window closing behavior. However, such misclassification is more likely to dilute the association than introduce a bias.

5. Conclusion

This study indicates diurnal patterns for the association between transportation noise and various cardiovascular causes of mortality. This suggests that different mechanisms are at play in the aetiology of various noise associated cardiovascular causes of deaths. Our results provide more support for protecting the population against nighttime noise as risk factor for CVD.

Conflict of interests

None to declare

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.ijheh.2018.02.005.

References

Ørn, S., Dickstein, K., 2002. How do heart failure patients die? Eur. Heart J. Suppl. D59–D65. http://dx.doi.org/10.1093/ehjsupp/4.
Arboix, A., Alió, J., 2012. Acute cardioembolic cerebral infarction: answers to clinical questions. Curr. Cardiol. Rev. 8, 54–67. http://dx.doi.org/10.2174/157340312801215791.
Babisch, W., Beule, B., Schust, M., Kersten, N., Ising, H., 2005. Traffic noise and risk of myocardial infarction. Epidemiology 16, 33–40.
Babisch, W., 2014. Updated exposure-response relationship between road traffic noise and coronary heart diseases: a meta-analysis. Noise Health 16, 1–9. http://dx.doi.org/10.4103/1463-1741.127847.
Basner, M., Müller, U., Elmenhorst, E.-M., 2011. Single and combined effects of air, road and rail traffic noise on sleep and recuperation. Sleep 34, 11–23.
Cappuccio, F.P., Cooper, D., D'Elia, L., Strazzullo, P., Miller, M.A., 2011. Sleep duration predicts cardiovascular outcomes: a systematic review and meta-analysis of prospective studies. Eur. Heart J. 32, 1484–1492.
Chen, L., Yang, G., 2015. Recent advances in circadian rhythms in cardiovascular system. Front. Pharmacol. 6, 71. http://dx.doi.org/10.3389/fphar.2015.00071.
Correia, A.W., Peters, J.L., Levy, J.I., Melly, S., Dominici, F., 2013. Residential Exposure to Aircraft Noise and Hospital Admissions for Cardiovascular Diseases: Multi-airport Retrospective Study.
Dickmeis, T., 2009. Glucocorticoids and the circadian clock. J. Endocrinol. 200, 3–22. http://dx.doi.org/10.1677/JOE-08-0415.
Dratva, J., et al., 2012. Transportation noise and blood pressure in a population-based sample of adults. Environ. Health Perspect. 120, 50.
Eze, I.C., et al., 2017. Long-term exposure to transportation noise and air pollution in relation to incident diabetes in the SAPALDIA study. Int. J. Epidemiol. 46, 115–1125. http://dx.doi.org/10.1093/ije/dyx020.
FOEN, 2013. PM10 and PM2.5 Ambient Concentrations in Switzerland. Modeling Results for 2005, 2010 and 2013. Federal Office for the Environment, Berne.
Ferro, J.M., 2003. Cardioembolic stroke: an update. Lancet Neurol. 2, 177–188.
Frei, P., Mohler, E., Röösli, M., 2014. Effect of nocturnal road traffic noise exposure and annoyance on objective and subjective sleep quality. Int. J. Hyg. Environ. Health 217, 188–195.
Halonen, J.I., et al., 2015. Road traffic noise is associated with increased cardiovascular morbidity and mortality and all-cause mortality in London. Eur. Heart J. 36, 2653–2661.
Hansell, A.L., et al., 2013. Aircraft noise and cardiovascular disease near Heathrow airport in London: small area study. BMJ Br. Med. J. 347. http://dx.doi.org/10.1136/bmj.f5432.
Heritier, H., et al., 2017. Transportation noise exposure and cardiovascular mortality: a nationwide cohort study from Switzerland. Eur. J. Epidemiol. 32, 307–315. http://dx.doi.org/10.1007/s10654-017-0234-2.
Huang, D., Song, X., Cui, Q., Tian, J., Wang, Q., Yang, K., 2015. Is there an association between aircraft noise exposure and the incidence of hypertension? A meta-analysis of 16784 participants. Noise Health 17, 93.
Huss, A., Spoerri, A., Egger, M., Röösli, M., 2010. Aircraft noise, air pollution, and mortality from myocardial infarction. Epidemiology 21, 829–836.
Jarup, L., Dudley, M., Babisch, W., Houthuijs, D., Swart, W., Pershagen, G., 2007. Hypertension and exposure to noise near airports—the HYENA study. Epidemiology 18, S137.
Karipidis, I., et al., 2014. Reconstruction of historical noise exposure data for environmental epidemiology in Switzerland within the SiRENE project. Noise Mapp. 1.
Knutson, K.L., Van Cauter, E., 2008. Associations between sleep loss and increased risk of obesity and diabetes. Ann. N. Y. Acad. Sci. 1129, 287–304. http://dx.doi.org/10.1196/annals.1417.033.
Lavie, C.J., Milani, R.V., Ventura, H.O., 2009. Obesity and cardiovascular disease. J. Am. Coll. Cardiol. 53, 1925.
Münzel, T., Gori, T., Babisch, W., Basner, M., 2014. Cardiovascular effects of environmental noise exposure. Eur. Heart J. 35, 829–836.
Moser, D.K., Stevenson, W.G., Woo, M.A., Stevenson, L.W., 1994. Timing of sudden death in patients with heart failure. J. Am. Coll. Cardiol. 24, 963–967.

Muller, J.E., 1999. Circadian variation and triggering of acute coronary events. Am. Heart J. 137, S1–S8.
Panczak, R., Galobardes, B., Voorpostel, M., Spoerri, A., Zwahlen, M., Egger, M., 2012. A Swiss neighbourhood index of socioeconomic position: development and association with mortality. J. Epidemiol. Commun. Health (JECH 2011-200699).
R Development Core Team, 2008. R: A Language and Environment for Statistical Computing. R Development Core Team, Vienna, Austria.
Sörensen, M., Andersen, Z.J., Nordsborg, R.B., Becker, T., Tjonneland, A., Overvad, K., Raaschou-Nielsen, O., 2013. Long-term exposure to road traffic noise and incident diabetes: a cohort study. Environ. Health Perspect. 121, 217–222. http://dx.doi.org/10.1289/ehp.1205503.
Sørensen, M., et al., 2012. Road traffic noise and incident myocardial infarction: a prospective cohort study. PLoS One 7, e39283.
Schlatter, F., et al., 2017. Validation of large scale noise exposure modeling by long-term measurements. Noise Mapping 4, 75–86.
Seidler, A., et al., 2016. Aircraft, road and railway traffic noise as risk factors for heart failure and hypertensive heart disease—a case-control study based on secondary data International. J. Hyg. Environ. Health 219, 749–758. https://doi.org/10.1016/j.ijheh.2016.09.012.
Sorensen, M., Luhdorf, P., Ketzel, M., Andersen, Z.J., Tjonneland, A., Overvad, K., Raaschou-Nielsen, O., 2014. Combined effects of road traffic noise and ambient air pollution in relation to risk for stroke? Environ. Res. 133, 49–55. http://dx.doi.org/10.1016/j.envres.2014.05.011.
Spoerri, A., Zwahlen, M., Egger, M., Bopp, M., 2010. The Swiss National Cohort: a unique database for national and international researchers. Int. J. Public Health 55, 239–242.
Therneau Terry, Terry M., 2000. Modeling Survival Data: Extending the Cox Model. Springer, New York.
Tinguely, G., Landolt, H.-P., Cajochen, C., 2014. Schlafgewohnheiten, Schlafqualität und Schlafmittelkonsum der Schweizer Bevölkerung Ergebnisse aus einer Umfrage bei einer repräsentativen. Stichprobe Therapeutische Umschau 71, 637–646.
Vienneau, D., Schindler, C., Perez, L., Probst-Hensch, N., Röösli, M., 2015. The relationship between transportation noise exposure and ischemic heart disease: a meta-analysis. Environ. Res. 138, 372–380.
Webb, W., Agnew, H., 1971. Stage 4 sleep: influence of time course variables. Science 174, 1354–1356.
Willich, S.N., Wegscheider, K., Stallmann, M., Keil, T., 2006. Noise burden and the risk of myocardial infarction. Eur. Heart J. 27, 276–282.
Wilson, P.W.F., D'Agostino, R.B., Levy, D., Belanger, A.M., Silbershatz, H., Kannel, W.B., 1998. Prediction of Coronary heart disease using risk factor categories. Circulation 97, 1837.
Wunderli, J.M., et al., 2016. Intermittency ratio: a metric reflecting short-term temporal variations of transportation noise exposure. J. Expo. Sci. Environ. Epidemiol. 26, 575–585.

Supplemental material

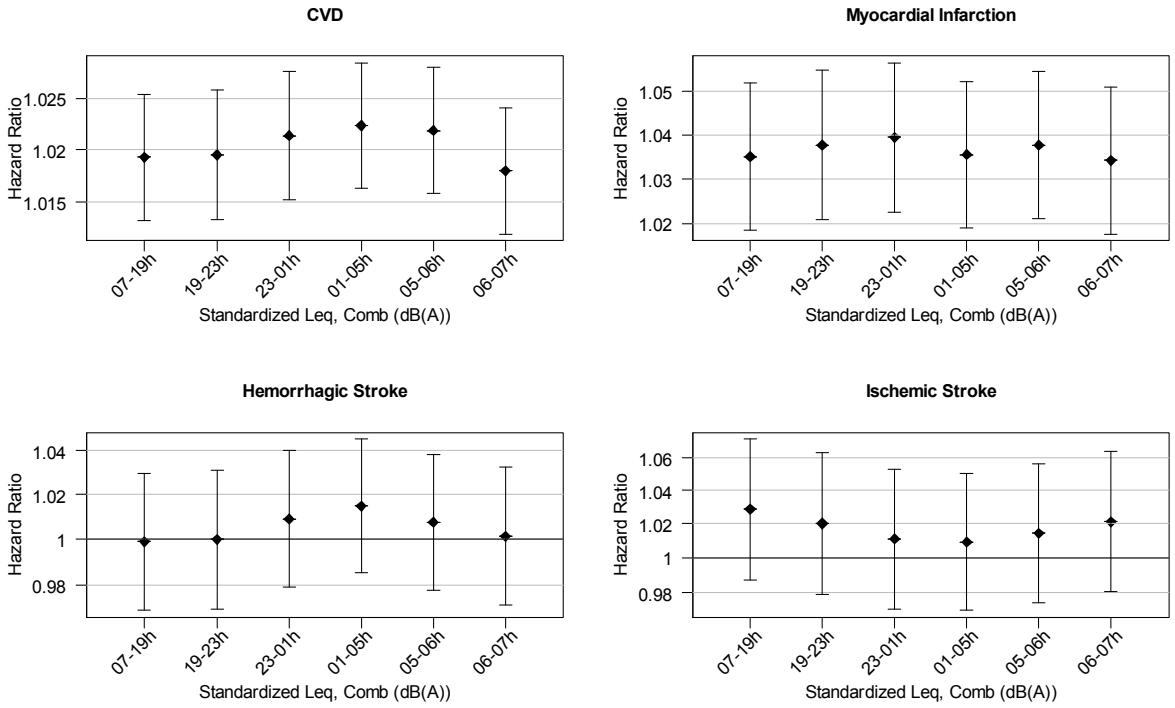
Diurnal variability of transportation noise exposure and cardiovascular mortality: a nationwide cohort study from Switzerland

Harris Héritier[a,b]; Danielle Vienneau[a,b]; Maria Foraster[a,b], Ikenna C. Eze[a,b], Emmanuel Schaffner[a,b] Laurie Thiesse[c], Franziska Ruzdik[c], Manuel Habermacher[d], Micha Köpfli[d], Reto Pieren[e], Arno Schmidt-Trucksäss[f], Mark Brink[g], Christian Cajochen[c], Jean Marc Wunderli[e], Nicole Probst-Hensch[a,b], Martin Röösli[a,b] for the SNC study group

[a] Swiss Tropical and Public Health Institute, Socinstrasse 57,4051 Basel, Switzerland
[b] University of Basel, Petersplatz 1, 4001 Basel, Switzerland
[c] Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Wilhelm Klein-Strasse 27, 4012 Basel, Switzerland
[d] N-sphere AG, Räffelstrasse 29, 8045 Zürich, Switzerland
[e] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Überland Str. 129, 8600 Dübendorf, Switzerland.
[f] Department of Sport, Exercise and Health, Division of Sports and Exercise Medicine, University of Basel, Birsstrasse 320, 4052 Basel, Switzerland
[g] Federal Office for the Environment, Papiermühlestrasse 172, 3063 Ittigen, Switzerland

***Corresponding author:**
Martin Röösli
Swiss Tropical and Public Health Institute
Socinstrasse 57
P.O. Box
CH-4002 Basel
E-Mail martin.roosli@swisstph.ch
Tel. +41 (0)61 284 83 83
Fax +41 (0)61 284 85 01

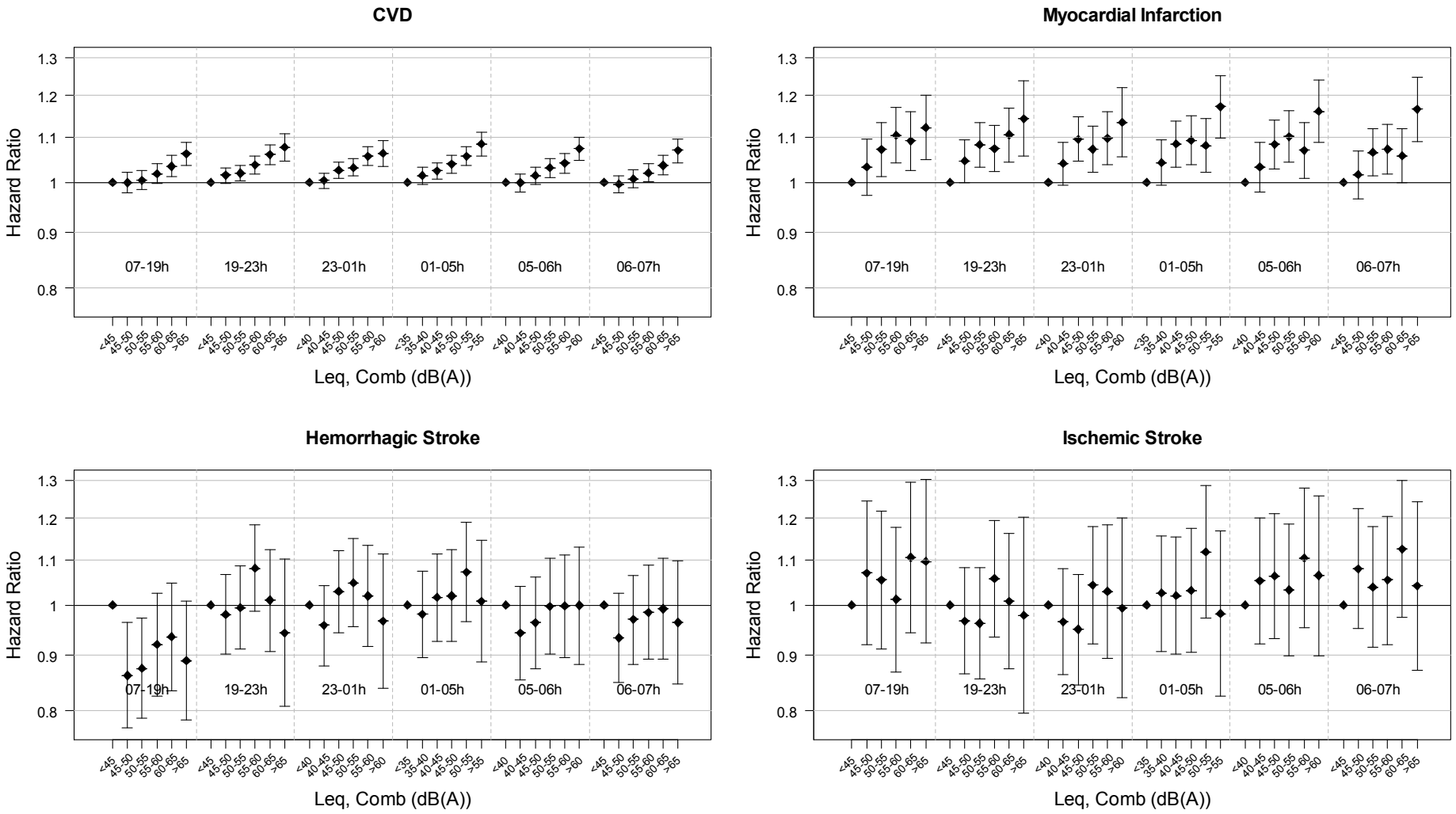
eFigure 1: Adjusted[a)] Hazard Ratios and 95% confidence intervals per increase of 1 standard deviation[b)] of standardized $L_{eq,comb}$ in the different time windows for the following causes of death: all CVD, myocardial infarction, hemorrhagic stroke and ischemic stroke.



[a)] adjusted for age (timescale), sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality, NO_2 exposure and IR_{comb}(Night).

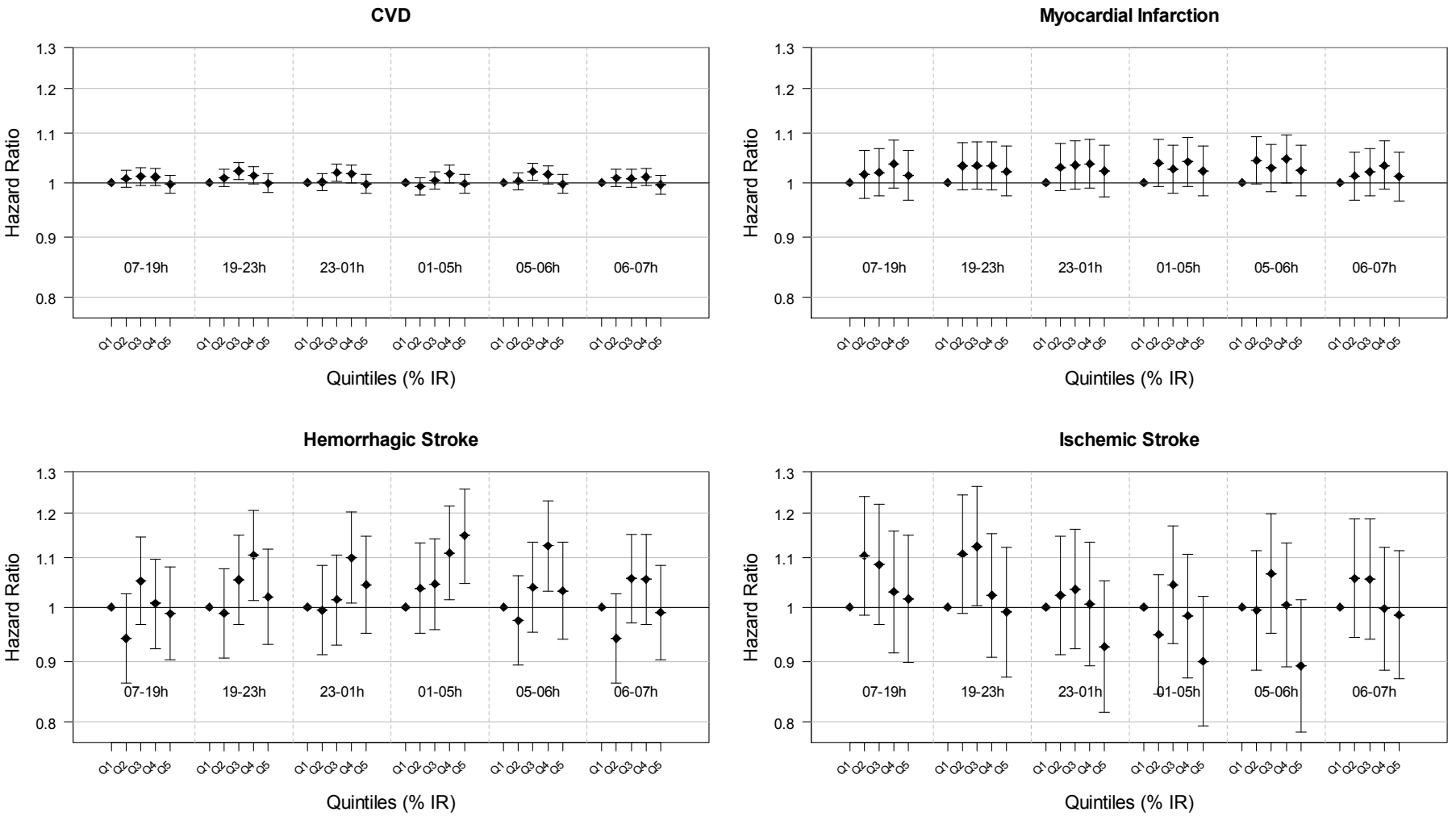
[b)] SD is between 7.35 and 8.01 (see eTable 6).

eFigure 2: Adjusted[a)] categorical HRs for specific time windows for the following causes of death: CVD, myocardial infarction, hemorrhagic and ischemic stroke.



[a)] adjusted for age (timescale), sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality, NO_2 exposure and IR_{comb}(Night).

eFigure 3: Adjusted[a)] categorical hazard ratio and 95% confidence intervals per quintiles of IR at different time windows for the following causes of death: CVD, myocardial infarction, hemorrhagic and ischemic stroke.



a) adjusted for age (timescale), sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality, NO_2 exposure and L_{den}.

eTable 1: Crude rates for all outcomes in relation to daytime and nighttime noise exposure categories (per 100,000 person years).

	Daytime noise (7:00-23:00)				Nighttime noise (23:00-7:00)			
	<45 dB	45-55 B	55-65 B	>65 dB	<45 dB	45-55 B	55-65 B	>65 dB
All cardiovascular diseases	2990	3126	3386	3487	3023	3329	3520	3225
Hypertensive diseases	265	297	327	326	284	316	339	303
Ischemic heart diseases	1262	1322	1421	1474	1280	1399	1477	1394
Myocardial infarction	388	423	461	474	402	457	471	471
Heart failure	260	262	298	313	256	288	315	272
Stroke	469	493	529	529	473	526	544	467
Hemorrhagic stroke	115	115	129	128	112	126	132	126
Ischemic stroke	58	67	70	72	64	69	75	58

eTable 2: Summary statistics of the transportation noise exposure $L_{eq,Comb}$ in the different time windows (all values in dB).

Exposure measure	Time window	Min	25th percentile	Median	Arithm. mean	75th percentile	95th percentile	Max
$L_{eq,Comb}$	19-23h	35	45.7	50.9	51.32	57	64.4	89.2
$L_{eq,Comb}$	23-01h	35	40.7	45.9	46.71	52.2	60.1	87.8
$L_{eq,Comb}$	01-05h	35	36.8	42.1	43.41	48.5	56.5	86.4
$L_{eq,Comb}$	05-06h	35	42.9	48.1	48.71	54.4	62.2	87.6
$L_{eq,Comb}$	06-07h	35	47.4	52.5	52.93	58.7	66.2	88.6
$L_{eq,Comb}$	07-19h	35	49.4	54.6	54.87	60.6	67.9	88.9
IR	19-23h	0	50.7	71.6	65.54	84	93.5	100
IR	23-01h	0	65.4	80	74.18	89.1	96.1	100
IR	01-05h	0	68.8	84.2	77.79	91.8	97.6	100
IR	05-06h	0	61.9	78.3	72.09	88.1	95.4	100
IR	06-07h	0	45.5	68.2	62.68	83.1	93.2	100
IR	07-19h	0	39.4	62.6	58.52	79.9	92.2	100

eTable 3: Spearman's rank correlations between $L_{den,Comb}$ and $L_{eq,Comb}$ in the different time windows.

	$L_{den,Comb}$	$L_{eq,Comb}$ (Day)[a]	$L_{eq,Comb}$ (Night)[b]	$L_{eq,Comb}$ (19-23)	$L_{eq,Comb}$ (23-01)	$L_{eq,Comb}$ (01-05)	$L_{eq,Comb}$ (05-06)	$L_{eq,Comb}$ (06-07)	$L_{eq,Comb}$ (07-19)	NO_2
$L_{den,Comb}$	1									
$L_{eq,Comb}$(Day)[a]	0.99	1								
$L_{eq,Comb}$ (Night)[b]	0.99	0.98	1							
$L_{eq,Comb}$(19-23)	1	0.99	0.98	1						
$L_{eq,Comb}$ (23-01)	0.99	0.97	0.99	0.98	1					
$L_{eq,Comb}$ (01-05)	0.96	0.94	0.98	0.95	0.98	1				
$L_{eq,Comb}$ (05-06)	0.98	0.97	0.99	0.97	0.98	0.99	1			
$L_{eq,Comb}$ (06-07)	0.99	0.99	0.99	0.98	0.98	0.96	0.98	1		
$L_{eq,Comb}$ (07-19)	0.99	1	0.97	0.99	0.96	0.94	0.97	0.98	1	
NO_2	0.34	0.35	0.34	0.35	0.33	0.34	0.33	0.33	0.34	1

[a]Daytime defined from 07 to 23h, [b] Nighttime defined from 23 to 07h

eTable 4: Spearman's rank Correlation for $L_{den,Comb}$ and IR_{Comb} in the different time windows.

	$L_{den,Comb}$	IR_{Comb} (Day)[a]	IR_{Comb} (Night)[b]	IR_{Comb} (19-23)	IR_{Comb} (23-01)	IR_{Comb} (01-05)	IR_{Comb} (05-06)	IR_{Comb} (06-07)	IR_{Comb} (07-19)
$L_{den,Comb}$	1								
IR_{Comb} (Day)[a]	0.1	1							
IR_{Comb} (Night)[b]	0.21	0.9	1						
IR_{Comb} (19-23)	0.15	0.98	0.95	1					
IR_{Comb} (23-01)	0.26	0.78	0.96	0.87	1				
IR_{Comb} (01-05)	0.37	0.45	0.73	0.56	0.85	1			
IR_{Comb} (05-06)	0.24	0.82	0.96	0.89	0.96	0.8	1		
IR_{Comb} (06-07)	0.12	0.97	0.94	0.97	0.83	0.52	0.88	1	
IR_{Comb} (07-19)	0.09	1	0.89	0.97	0.77	0.43	0.82	0.97	1

[a] Daytime defined from 07 to 23h, [b] Nighttime defined from 23 to 07h

eTable 5: Spearman's rank correlations between $L_{eq,Comb}$ including $L_{den,Comb}$ and source-specific L_{eqs} including L_{dens} in the different time windows.

	$L_{den,Comb}$	$L_{eq,Comb}$ (Day)[a]	$L_{eq,Comb}$ (Night)[b]	$L_{eq,Comb}$ (19-23)	$L_{eq,Comb}$ (23-01)	$L_{eq,Comb}$ (01-05)	$L_{eq,Comb}$ (05-06)	$L_{eq,Comb}$ (06-07)	$L_{eq,Comb}$ (07-19)	NO_2
L_{den}(Road)	0.88	0.91	0.86	0.86	0.81	0.79	0.87	0.90	0.91	0.33
$L_{eq,Road}$(Day)[a]	0.88	0.90	0.86	0.86	0.81	0.79	0.87	0.90	0.91	0.33
$L_{eq,Road}$ (Night)[b]	0.87	0.89	0.86	0.85	0.82	0.81	0.88	0.89	0.90	0.33
$L_{eq,Road}$(19-23)	0.88	0.90	0.86	0.86	0.82	0.80	0.88	0.90	0.91	0.33
$L_{eq,Road}$ (23-01)	0.86	0.89	0.86	0.85	0.82	0.82	0.87	0.89	0.89	0.33
$L_{eq,Road}$ (01-05)	0.84	0.86	0.85	0.83	0.81	0.82	0.86	0.87	0.87	0.34
$L_{eq,Road}$ (05-06)	0.87	0.90	0.86	0.85	0.82	0.81	0.88	0.90	0.90	0.33
$L_{eq,Road}$ (06-07)	0.88	0.90	0.86	0.86	0.82	0.80	0.88	0.90	0.91	0.33
$L_{eq,Road}$ (07-19)	0.88	0.90	0.86	0.86	0.81	0.79	0.87	0.90	0.91	0.33
L_{den}(Rail)	0.43	0.36	0.47	0.44	0.53	0.53	0.47	0.39	0.35	0.30
$L_{eq,Rail}$(Day)[a]	0.44	0.38	0.49	0.45	0.55	0.54	0.48	0.41	0.36	0.30
$L_{eq,Rail}$ (Night)[b]	0.42	0.36	0.46	0.43	0.52	0.52	0.46	0.38	0.34	0.30
$L_{eq,Rail}$(19-23)	0.44	0.38	0.49	0.45	0.55	0.55	0.48	0.41	0.36	0.30
$L_{eq,Rail}$ (23-01)	0.42	0.36	0.46	0.43	0.52	0.51	0.46	0.38	0.34	0.30
$L_{eq,Rail}$ (01-05)	0.42	0.36	0.47	0.44	0.53	0.54	0.46	0.38	0.34	0.30
$L_{eq,Rail}$ (05-06)	0.42	0.36	0.46	0.43	0.51	0.51	0.45	0.38	0.34	0.30
$L_{eq,Rail}$ (06-07)	0.41	0.35	0.45	0.42	0.50	0.50	0.45	0.38	0.34	0.30
$L_{eq,Rail}$ (07-19)	0.44	0.38	0.49	0.45	0.54	0.54	0.48	0.41	0.36	0.30
L_{den}(Air)	0.18	0.21	0.11	0.23	0.12	0.04	0.07	0.14	0.20	0.40
$L_{eq,Air}$(Day)[a]	0.18	0.21	0.11	0.23	0.12	0.05	0.07	0.14	0.21	0.40
$L_{eq,Air}$ (Night)[b]	0.17	0.19	0.12	0.21	0.12	0.04	0.06	0.15	0.18	0.40
$L_{eq,Air}$(19-23)	0.18	0.21	0.11	0.23	0.12	0.04	0.06	0.14	0.20	0.40
$L_{eq,Air}$ (23-01)	0.17	0.19	0.11	0.22	0.12	0.04	0.05	0.13	0.18	0.40
$L_{eq,Air}$ (01-05)	0.09	0.09	0.08	0.12	0.08	0.00	0.01	0.09	0.08	0.41
$L_{eq,Air}$ (05-06)	0.08	0.07	0.05	0.10	0.05	0.00	0.02	0.07	0.07	0.40
$L_{eq,Air}$ (06-07)	0.17	0.19	0.12	0.21	0.12	0.04	0.07	0.15	0.18	0.40
$L_{eq,Air}$ (07-19)	0.18	0.21	0.11	0.23	0.12	0.05	0.07	0.14	0.21	0.40

[a] Daytime defined from 07 to 23h, [b] Nighttime defined from 23 to 07h

eTable 6: Adjusted HRs and 95% CI for standardized L_{eq} ad different time window with corresponding standard deviation (SD) for all outcomes under investigation.

Outcome	Time window	HR (95% CI)	SD (in dB)
Cardiovascular diseases	07-19h	1.019 (1.013-1.025)	7.98
Cardiovascular diseases	19-23h	1.020 (1.013-1.026)	7.97
Cardiovascular diseases	23-01h	1.021 (1.015-1.028)	7.81
Cardiovascular diseases	01-05h	1.022 (1.016-1.028)	7.35
Cardiovascular diseases	05-06h	1.022 (1.016-1.028)	7.95
Cardiovascular diseases	06-07h	1.018 (1.012-1.024)	8.01
Hypertensive disease	07-19h	1.044 (1.024-1.064)	7.98
Hypertensive disease	19-23h	1.041 (1.021-1.061)	7.97
Hypertensive disease	23-01h	1.038 (1.019-1.058)	7.81
Hypertensive disease	01-05h	1.041 (1.022-1.061)	7.35
Hypertensive disease	05-06h	1.046 (1.026-1.066)	7.95
Hypertensive disease	06-07h	1.042 (1.022-1.062)	8.01
Ischemic heart diseases	07-19h	1.018 (1.009-1.028)	7.98
Ischemic heart diseases	19-23h	1.019 (1.010-1.029)	7.97
Ischemic heart diseases	23-01h	1.023 (1.014-1.033)	7.81
Ischemic heart diseases	01-05h	1.025 (1.016-1.034)	7.35
Ischemic heart diseases	05-06h	1.023 (1.014-1.033)	7.95
Ischemic heart diseases	06-07h	1.017 (1.008-1.026)	8.01
Myocardial infarction	07-19h	1.035 (1.019-1.052)	7.98
Myocardial infarction	19-23h	1.038 (1.021-1.055)	7.97
Myocardial infarction	23-01h	1.039 (1.023-1.057)	7.81
Myocardial infarction	01-05h	1.035 (1.019-1.052)	7.35
Myocardial infarction	05-06h	1.038 (1.021-1.054)	7.95
Myocardial infarction	06-07h	1.034 (1.018-1.051)	8.01
Heart failure	07-19h	1.047 (1.027-1.068)	7.98
Heart failure	19-23h	1.046 (1.025-1.067)	7.97
Heart failure	23-01h	1.036 (1.016-1.057)	7.81
Heart failure	01-05h	1.028 (1.007-1.048)	7.35
Heart failure	05-06h	1.037 (1.016-1.058)	7.95
Heart failure	06-07h	1.043 (1.023-1.064)	8.01
Stroke	07-19h	1.002 (0.988-1.018)	7.98
Stroke	19-23h	1.001 (0.986-1.016)	7.97
Stroke	23-01h	1.001 (0.986-1.016)	7.81
Stroke	01-05h	1.001 (0.986-1.016)	7.35
Stroke	05-06h	1.001 (0.986-1.016)	7.95
Stroke	06-07h	0.999 (0.984-1.014)	8.01
Hemorrhagic stroke	07-19h	0.999 (0.969-1.030)	7.98
Hemorrhagic stroke	19-23h	1.000 (0.970-1.031)	7.97
Hemorrhagic stroke	23-01h	1.009 (0.979-1.040)	7.81
Hemorrhagic stroke	01-05h	1.015 (0.985-1.045)	7.35
Hemorrhagic stroke	05-06h	1.007 (0.978-1.038)	7.95
Hemorrhagic stroke	06-07h	1.001 (0.971-1.032)	8.01
Ischemic stroke	07-19h	1.029 (0.988-1.072)	7.98
Ischemic stroke	19-23h	1.020 (0.979-1.063)	7.97
Ischemic stroke	23-01h	1.011 (0.971-1.053)	7.81
Ischemic stroke	01-05h	1.009 (0.970-1.051)	7.35
Ischemic stroke	05-06h	1.014 (0.974-1.056)	7.95
Ischemic stroke	06-07h	1.021 (0.980-1.064)	8.01

eTable 7: Adjusted categorical HRs and 95% confidence intervals (CI) for cardiovascular causes of death and categorical IR_{Comb} for various time windows by quintiles.

	Time window	1st quintile (reference)	2nd quintile	3rd quintile	4th quintile	5th quintile
Cardiovascular diseases	**IR_{Comb} (07-19h)**	1	1.008 (0.991-1.025)	1.012 (0.995-1.029)	1.011 (0.994-1.028)	0.997 (0.980-1.015)
	IR_{Comb} (19-23h)	1	1.009 (0.992-1.026)	1.023 (1.006-1.040)	1.014 (0.997-1.032)	0.999 (0.981-1.017)
	IR_{Comb} (23-01h)	1	1.001 (0.984-1.018)	1.020 (1.002-1.037)	1.017 (0.999-1.035)	0.997 (0.979-1.016)
	IR_{Comb} (01-05h)	1	0.993 (0.976-1.009)	1.004 (0.987-1.021)	1.017 (1.000-1.035)	0.998 (0.980-1.016)
	IR_{Comb} (05-06h)	1	1.003 (0.986-1.020)	1.022 (1.004-1.039)	1.016 (0.998-1.034)	0.997 (0.979-1.016)
	IR_{Comb} (06-07h)	1	1.009 (0.992-1.026)	1.008 (0.991-1.026)	1.011 (0.994-1.028)	0.996 (0.978-1.014)
Hypertensive disease	**IR_{Comb} (07-19h)**	1	1.058 (1.001-1.119)	1.043 (0.986-1.103)	1.034 (0.978-1.094)	1.025 (0.967-1.086)
	IR_{Comb} (19-23h)	1	1.036 (0.979-1.095)	1.044 (0.987-1.104)	1.036 (0.979-1.096)	1.004 (0.946-1.064)
	IR_{Comb} (23-01h)	1	1.026 (0.971-1.085)	1.057 (0.999-1.117)	0.989 (0.933-1.048)	0.974 (0.917-1.035)
	IR_{Comb} (01-05h)	1	0.983 (0.931-1.039)	1.006 (0.952-1.063)	0.962 (0.908-1.019)	0.960 (0.905-1.017)
	IR_{Comb} (05-06h)	1	1.006 (0.952-1.064)	1.014 (0.959-1.073)	1.001 (0.945-1.060)	0.975 (0.918-1.035)
	IR_{Comb} (06-07h)	1	1.041 (0.985-1.102)	1.047 (0.990-1.108)	1.029 (0.973-1.089)	1.024 (0.966-1.086)

Ischemic heart diseases	**IR_{Comb} (07-19h)**	1	1.016 (0.990-1.043)	1.016 (0.990-1.043)	1.019 (0.993-1.047)	1.017 (0.990-1.045)
	IR_{Comb} (19-23h)	1	1.020 (0.994-1.047)	1.028 (1.002-1.055)	1.018 (0.992-1.046)	1.020 (0.992-1.049)
	IR_{Comb} (23-01h)	1	1.010 (0.985-1.037)	1.021 (0.994-1.048)	1.030 (1.002-1.058)	1.014 (0.986-1.043)
	IR_{Comb} (01-05h)	1	0.994 (0.968-1.020)	0.995 (0.970-1.022)	1.025 (0.998-1.053)	1.001 (0.974-1.029)
	IR_{Comb} (05-06h)	1	1.020 (0.993-1.047)	1.027 (1.001-1.055)	1.022 (0.995-1.050)	1.018 (0.990-1.048)
	IR_{Comb} (06-07h)	1	1.017 (0.991-1.044)	1.011 (0.984-1.038)	1.016 (0.989-1.043)	1.015 (0.988-1.044)
Myocardial infarction	**IR_{Comb} (07-19h)**	1	1.016 (0.970-1.064)	1.020 (0.974-1.068)	1.037 (0.990-1.086)	1.014 (0.966-1.064)
	IR_{Comb} (19-23h)	1	1.033 (0.986-1.081)	1.034 (0.987-1.083)	1.034 (0.986-1.083)	1.022 (0.974-1.073)
	IR_{Comb} (23-01h)	1	1.030 (0.984-1.079)	1.035 (0.988-1.084)	1.037 (0.989-1.088)	1.023 (0.973-1.075)
	IR_{Comb} (01-05h)	1	1.039 (0.992-1.088)	1.027 (0.980-1.075)	1.041 (0.993-1.092)	1.023 (0.974-1.074)
	IR_{Comb} (05-06h)	1	1.044 (0.997-1.093)	1.029 (0.982-1.078)	1.047 (0.999-1.098)	1.024 (0.974-1.076)
	IR_{Comb} (06-07h)	1	1.013 (0.967-1.061)	1.021 (0.974-1.069)	1.034 (0.987-1.084)	1.012 (0.964-1.062)
Heart failure	**IR_{Comb} (07-19h)**	1	1.026 (0.968-1.088)	1.011 (0.953-1.073)	1.002 (0.945-1.063)	0.944 (0.888-1.004)
	IR_{Comb} (19-23h)	1	0.992 (0.935-1.051)	0.994 (0.937-1.054)	0.975 (0.919-1.035)	0.936 (0.880-0.996)
	IR_{Comb} (23-01h)	1	1.008 (0.951-1.069)	0.999 (0.942-1.060)	0.997 (0.939-1.059)	0.961 (0.902-1.023)
	IR_{Comb} (01-05h)	1	0.977 (0.921-1.036)	0.995 (0.939-1.055)	0.989 (0.932-1.050)	0.986 (0.928-1.048)

	IR_{Comb} (05-06h)	1	1.004 (0.947-1.065)	0.999 (0.941-1.060)	0.991 (0.933-1.052)	0.943 (0.886-1.004)
	IR_{Comb} (06-07h)	1	1.049 (0.990-1.113)	0.998 (0.940-1.059)	1.010 (0.952-1.072)	0.935 (0.879-0.995)
Stroke	**IR_{Comb} (07-19h)**	1	1.002 (0.960-1.045)	1.013 (0.970-1.057)	1.013 (0.970-1.057)	0.994 (0.951-1.039)
	IR_{Comb} (19-23h)	1	1.011 (0.969-1.055)	1.037 (0.994-1.082)	1.037 (0.993-1.083)	1.014 (0.969-1.061)
	IR_{Comb} (23-01h)	1	0.996 (0.955-1.040)	1.027 (0.984-1.073)	1.031 (0.987-1.078)	1.024 (0.978-1.073)
	IR_{Comb} (01-05h)	1	0.993 (0.951-1.036)	1.026 (0.983-1.071)	1.046 (1.001-1.094)	1.028 (0.982-1.076)
	IR_{Comb} (05-06h)	1	0.986 (0.945-1.030)	1.045 (1.001-1.091)	1.031 (0.986-1.077)	1.018 (0.972-1.066)
	IR_{Comb} (06-07h)	1	0.985 (0.944-1.028)	0.997 (0.955-1.041)	1.017 (0.974-1.062)	0.991 (0.947-1.036)
Hemorrhagic stroke	**IR_{Comb} (07-19h)**	1	0.941 (0.863-1.026)	1.052 (0.966-1.146)	1.007 (0.923-1.098)	0.987 (0.902-1.081)
	IR_{Comb} (19-23h)	1	0.988 (0.906-1.078)	1.054 (0.967-1.150)	1.106 (1.013-1.206)	1.020 (0.930-1.119)
	IR_{Comb} (23-01h)	1	0.994 (0.912-1.084)	1.015 (0.929-1.107)	1.100 (1.007-1.202)	1.044 (0.950-1.147)
	IR_{Comb} (01-05h)	1	1.037 (0.951-1.132)	1.046 (0.957-1.142)	1.111 (1.015-1.216)	1.149 (1.047-1.259)
	IR_{Comb} (05-06h)	1	0.974 (0.893-1.063)	1.039 (0.952-1.134)	1.126 (1.031-1.230)	1.032 (0.939-1.134)
	IR_{Comb} (06-07h)	1	0.941 (0.862-1.027)	1.057 (0.970-1.152)	1.055 (0.967-1.151)	0.990 (0.903-1.085)
Ischemic stroke	**IR_{Comb} (07-19h)**	1	1.105 (0.984-1.240)	1.086 (0.966-1.221)	1.030 (0.914-1.160)	1.016 (0.898-1.150)
	IR_{Comb} (19-23h)	1	1.109 (0.988-1.244)	1.125 (1.002-1.264)	1.023 (0.907-1.154)	0.991 (0.873-1.124)

	IR_{Comb} (23-01h)	1	1.023 (0.912-1.148)	1.035 (0.922-1.163)	1.006 (0.892-1.135)	0.926 (0.815-1.053)
	IR_{Comb} (01-05h)	1	0.948 (0.844-1.064)	1.044 (0.931-1.171)	0.983 (0.872-1.108)	0.900 (0.794-1.021)
	IR_{Comb} (05-06h)	1	0.994 (0.885-1.116)	1.067 (0.950-1.198)	1.004 (0.891-1.132)	0.892 (0.784-1.015)
	IR_{Comb} (06-07h)	1	1.057 (0.942-1.187)	1.055 (0.939-1.186)	0.997 (0.885-1.124)	0.985 (0.870-1.115)

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7. Paper 4

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Modelling daily $PM_{2.5}$ concentrations at high spatio-temporal resolution across Switzerland

Kees de Hoogh[1,2], Harris Héritier[1,2], Massimo Stafoggia[3], Nino Künzli[1,2], Itai Kloog[4]

[1]Swiss Tropical and Public Health Institute, Basel, Switzerland

[2]University of Basel, Basel, Switzerland

[3]Department of Epidemiology, Lazio Regional Health Service, Rome, Italy

[4]Department of Geography and Environmental Development, Ben-Gurion University of the Negev, P.O.B. 653, Beer Sheva, Israel

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Modelling daily $PM_{2.5}$ concentrations at high spatio-temporal resolution across Switzerland☆

Kees de Hoogh [a, b, *], Harris Héritier [a, b], Massimo Stafoggia [c], Nino Künzli [a, b], Itai Kloog [d]

[a] Swiss Tropical and Public Health Institute, Basel, Switzerland
[b] University of Basel, Basel, Switzerland
[c] Department of Epidemiology, Lazio Regional Health Service, Rome, Italy
[d] Department of Geography and Environmental Development, Ben-Gurion University of the Negev, P.O.B. 653, Beer Sheva, Israel

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ABSTRACT

Spatiotemporal resolved models were developed predicting daily fine particulate matter ($PM_{2.5}$) concentrations across Switzerland from 2003 to 2013. Relatively sparse $PM_{2.5}$ monitoring data was supplemented by imputing $PM_{2.5}$ concentrations at PM_{10} sites, using $PM_{2.5}/PM_{10}$ ratios at co-located sites. Daily $PM_{2.5}$ concentrations were first estimated at a 1 × 1km resolution across Switzerland, using Multiangle Implementation of Atmospheric Correction (MAIAC) spectral aerosol optical depth (AOD) data in combination with spatiotemporal predictor data in a four stage approach. Mixed effect models (1) were used to predict $PM_{2.5}$ in cells with AOD but without $PM_{2.5}$ measurements (2). A generalized additive mixed model with spatial smoothing was applied to generate grid cell predictions for those grid cells where AOD was missing (3). Finally, local $PM_{2.5}$ predictions were estimated at each monitoring site by regressing the residuals from the 1 × 1km estimate against local spatial and temporal variables using machine learning techniques (4) and adding them to the stage 3 global estimates. The global (1 km) and local (100 m) models explained on average 73% of the total,71% of the spatial and 75% of the temporal variation (all cross validated) globally and on average 89% (total) 95% (spatial) and 88% (temporal) of the variation locally in measured $PM_{2.5}$ concentrations.

1. Introduction

Fine particulate matter ($PM_{2.5}$) has been linked to a number of adverse health effects including lung cancer, natural cause mortality, low birth weight, lung function in both adults and children and acute coronary events (Raaschou-Nielsen et al., 2013; Beelen et al., 2014; Pedersen et al., 2013; Gehring et al., 2013; Cesaroni et al., 2014; Adam et al., 2015). These studies also show that effects are significant at $PM_{2.5}$ levels below 15–20 μg/m^3 (Brunekreef et al., 2015).

Estimating historic levels of $PM_{2.5}$ at a small spatial and temporal scale, necessary for epidemiological studies, however, is an ongoing challenge. $PM_{2.5}$ monitoring sites are sparse, especially going back in time, and therefore do not provide good spatial nor temporal coverage. Current research often relies on so-called back-extrapolation to estimate historical air pollution exposures (Pedersen et al., 2013; Tétreault et al., 2016; Panasevich et al., 2016). Routine monitoring data is thereby used to convert estimates from a recent land use regression (LUR) model to the time of the health event or baseline of cohort study. The assumption in back extrapolation is that the spatial structure of the air pollution surface is stable over a sufficiently long period of time. There is evidence to justify back-extrapolation for annual air pollution surfaces (Gulliver et al., 2016), the spatial structure for shorter time periods, however, is likely to change, for example due to changing meteorological conditions, leading to fluctuations in air pollution levels which are difficult to be picked up by back-extrapolation. In Switzerland there are existing long-term (annual and bi-annual) air pollution models for $PM_{2.5}$ developed using LUR for specific years (Aguilera et al., 2015; Eeftens et al., 2015) and regularly updated dispersion models (FOEN, 2013). Models to estimate historic $PM_{2.5}$ concentrations at a fine spatial temporal scale do not exist, something this paper will attempt to address.

Since the early 2000s, satellites have been measuring the light

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* Corresponding author. Swiss Tropical and Public Health Institute, Socinstrasse 57, CH-4051, Basel, Switzerland.
E-mail address: c.dehoogh@swisstph.ch (K. de Hoogh).

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extinction by aerosol scattering and its absorption in the atmospheric column, also called aerosol optical depth (AOD). Recent studies in North America (US, Mexico), Europe (Italy) and China have utilised this data by applying sophisticated statistical methods to predict $PM_{2.5}$ concentrations (Kloog et al., 2012, 2014; Stafoggia et al., 2017; Li et al., 2017). These studies all demonstrate that satellite-derived AOD measurements have the potential to help explain spatial and temporal variations in $PM_{2.5}$ concentrations across large spatial domains (Kloog et al., 2012, 2014; Stafoggia et al., 2017). While this method has been successfully applied in Northern America, Mexico and Italy, more testing, in other challenging areas, is still needed.

Switzerland poses some unique challenges in terms of air pollution modelling, both topographically and meteorologically. The country has a diversity of landscapes and climates, from high mountains to plateaus to valleys, which influences the meteorological phenomena including complex wind fields, inversions and temperature patterns due to the topography. Compared to other parts of the world, Switzerland has very low levels of $PM_{2.5}$. The range of annual $PM_{2.5}$ concentrations measured at the National Air Pollution Monitoring Network (NABEL), has decreased from 11 to 28 $\mu g/m^3$ in 2003 to 6–18 $\mu g/m^3$ in 2013. With the ongoing debate as to whether there is a threshold for health effects (Brunekreef et al., 2015), it is important to be able to model exposures for low concentration levels. This makes Switzerland an interesting study area to further develop, test and adapt the method originally applied in Northern America (Kloog et al., 2014) and test its validity in low pollution areas. Another challenge is the, for our purposes, small number of $PM_{2.5}$ monitoring sites in Switzerland, with 10 sites operating between 2003 and 2013. In this paper we used PM_{10} monitoring data to supplement the sparse $PM_{2.5}$ data.

This paper describes the development of spatiotemporal models predicting daily $PM_{2.5}$ concentrations for Switzerland from 2003 to 2013. The modelling further develops a previously presented methodology (Kloog et al., 2014; Stafoggia et al., 2017) applied in North-America and Italy. In the first three stages of the 4 stage approach, daily $PM_{2.5}$ concentrations are estimated at a 1×1km resolution across Switzerland, taking advantage of the recently available MAIAC (Multiangle Implementation of Atmospheric Correction) spectral AOD data and spatiotemporal predictor data. In stage 4, we attempt to explain the local variability of air pollution at a 100×100 m scale by applying support vector machine algorithms.

2. Materials and methods

2.1. Study area

Switzerland, situated in Central Europe and with a surface area of 41,285 km^2, is land-locked by Germany, Austria, Italy and France and at the end of 2015 had a population of 8.3 million (FSO, 2016). Switzerland is dominated by two mountain ranges; the Alps in the South and the Jura in the Northwest. In between lies the Swiss Mittelland, which is densely populated and intensely farmed. Due to its topography, the climate across the country varies locally, but in general is temperate, with warm and humid summers and more dryer and stable conditions in the winter. The most important sectors of the Swiss economy are industry, tourism and services.

2.2. Monitoring data

The regulatory monitoring network for Switzerland, NABEL, comprises of 16 measurement sites distributed across the country, of which 10 measure $PM_{2.5}$ (see Figure A.1). Daily (Jan 2003–April 2010) and four-daily (April 2010–Dec 2013) $PM_{2.5}$ measurement data, conducted at these 10 sites, were obtained from the Immissionsdatenbank Luft (IDB) (Federal Office of the Environment, Bern, Switzerland), with not all sites measuring all years (see Table A.1). $PM_{2.5}$ measurements were carried out using gravimetry. Given the insufficient number of $PM_{2.5}$ monitoring sites to inform our models, we further obtained daily PM_{10} data from IDB for the same 10 stations measuring $PM_{2.5}$ (i.e. co-located) plus at an additional 89 stations including more NABEL stations and cantonal sites (see Figure A.1). PM_{10} measurements were carried out by gravimetry, betameter or TEOM. With the view to impute $PM_{2.5}$ concentrations at PM_{10} sites, we calculated the correlation of determination between daily $PM_{2.5}$ and PM_{10} concentrations at co-located sites for each year. We evaluated the performance of this approach at each of the 10 co-located sites, by applying the $PM_{2.5}/PM_{10}$ relationship from the nearest co-located site and comparing imputed $PM_{2.5}$ with measured $PM_{2.5}$. Subsequently, the extracted equations were applied to the 89 PM_{10} sites to impute $PM_{2.5}$ concentrations, again, by using the nearest co-located site. The 99 monitoring sites with both measured and imputed $PM_{2.5}$ concentrations will be referred to as $PM_{2.5}$ monitoring sites from now on.

2.3. AOD data

Daily MAIAC spectral AOD was derived from MODIS (Moderate Resolution Imaging Spectroradiometer) Aqua Collection 6 L1B data for the period of 2003–2013. The MAIAC algorithm provides the high-resolution (1 km) AOD product that was used in our analysis for Switzerland. The MAIAC data from MODIS Aqua represent a mid-afternoon measurement [mean daily acquisition time of 2:57 p.m. Greenwich Mean Time (GMT) −5 h; range 2:10–3:45 p.m. local overpass time]. MAIAC uses a time series analysis and processing of groups of pixels to derive surface bidirectional reflectance distribution function (BRDF) and aerosol parameters over both dark vegetated surfaces and bright surfaces, without assumptions typical of current MODIS operational processing algorithms. The spatio-temporal analysis also helps MAIAC's cloud mask augmenting traditional pixel-level cloud detection techniques. MAIAC's high resolution, which is important in many applications such as air pollution studies, brings new information about aerosol sources and, potentially, their strength. An in-depth description of the MAIAC product and algorithm details can be found elsewhere (Lyapustin et al., 2011a, 2011b). Because AOD values may be spurious at cloud edges, AOD data were filtered to exclude values with adjacent cloud or high uncertainty flags and with a moving window variance in the top 2.5th percentile. The resulting MAIAC data set included 19.8% of all possible observations, which is consistent with previous work (Kloog et al., 2014).

2.4. Spatial and temporal predictors

Spatial and temporal predictor data were extracted at different scales. Global predictors (to inform stage 1–3 models) were extracted for each grid cell (1×1km, based on the MAIAC AOD geography). Local predictors (to inform stage 4 models) were extracted at the PM monitoring site locations. Lastly, local predictor variables were extracted at 100×100 m grid cell centroids (nested in the 1×1km grid) to facilitate the mapping process at the local scale by combining stage 3 1×1km and stage 4 100×100 m estimates. Predictors, detailed information of which can be found in Appendix A, included: emission data obtained from Meteotest (FOEN, 2013; FOEN, 2011); road data originating from VECTOR25 and TeleAtlas MultiNet TM; elevation data from the Swiss Federal Office of Topography; land use from European Corine Landcover; meteorology obtained from the European Centre for Medium-Range Weather Forecasts (Dee et al., 2011); and NDVI from the

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LANDSAT 8 satellite.

2.5. Pre-processing $PM_{2.5}$ and AOD data

Before use in the modelling framework, the $PM_{2.5}$ monitoring and the AOD retrieval data were cleaned. The rationale for this lies in the objective of the mixed model framework's first stage, namely the optimisation of the calibration between daily $PM_{2.5}$ measurements and co-located AOD retrievals, i.e. to predict $PM_{2.5}$ based on AOD data and additional spatial and spatiotemporal parameters. Depending on the conditions (i.e. meteorological, geographical etc.), AOD and $PM_{2.5}$ are poorly to weakly correlated, but by accounting for spatial and temporal heterogeneity in the AOD-$PM_{2.5}$ relationship we assume that AOD becomes an important factor to explain $PM_{2.5}$ variability. To be able to do this, however, we needed to make sure that only good quality AOD observations were retained and that unreliable AOD data due to, for example, cloud contamination, proximity to water surfaces or snow-covered areas, malfunctioning of the sensor on board the satellite, etc., were removed. This cleaning process, prior to the modelling, therefore gave us a well formulated calibration model which could be reasonably applied to areas and days when no $PM_{2.5}$ data exist (Stafoggia et al., 2017). More details about the pre-processing, including of numbers of record removed, can be found in Appendix A.

2.6. Statistical methods

A 4-staged modelling approach (see Figure A.2), described in detail by Kloog et al. (2014). was adapted and further developed to estimate $PM_{2.5}$ concentrations at both 1 km a 100 m grid cells across Switzerland from 2003 to 2013.

2.6.1. Stage 1

Due to the complex topography and climatology of Switzerland we decided to perform the stage 1 modelling, i.e. the day-specific random intercepts and slopes, in five nested climatic regions; Upper Rhine plains, Jura, Swiss Mittelland, Prealpine zones and Swiss Alps.

For each year we fitted the following model (slight deviations for each year), calibrating the nearest (within 1500 m) AOD observation to the $PM_{2.5}$ monitoring data using all daily $PM_{2.5}$ measurements, whilst adjusting for temporal and spatial covariates:

$$PM2.5_{ijk} = \left(\alpha + u_j + u_{jk}\right) + \left(\beta_1 + v_j + v_{jk}\right)AOD_{ij} + \left(\beta_2 + w_j + w_{jk}\right)PBL_{ij} + \beta_3 Wind\,Sp_{ijk} + \beta_4 Wind\,Dir_{ijk} + \beta_5 Temp_{ijk} + \beta_6 Prec_{ijk} + \beta_7 NDVI_{ijk} + \beta_8 Spatial_{ik} + \varepsilon_{ijk}$$

- Where $PM25_{ijk}$ denotes the measured $PM_{2.5}$ concentration at a site i on a day j in climatic region k;
- α is the fixed and u_j and u_{jk} are the random daily intercepts on a day j in the climatic region k;
- AOD_{ij} is the AOD value in the grid cell corresponding to site i on day j, β_1 is the fixed slope, and v_j and v_{jk} are the random slopes on a day j and on a day j in the climatic region k;
- PBL_{ij} is the planet boundary layer (PBL) value (site i on day j), β_2 is the fixed slope, and w_j and w_{jk} are the random slopes on a day j and on a day j in the climatic region k;
- $Wind\,Sp_{ijk}$, $Wind\,Dir_{ijk}$, $Temp_{ijk}$ and $Prec_{ijk}$ are the assigned values of respectively wind speed, wind direction, temperature and precipitation in the grid cells corresponding to site i on a day j in the climatic region k
- $NDVI_{ijk}$ is the mean NDVI value in the grid cell containing site i on a day j in the climatic region k;
- $Spatial_{ik}$ are the spatial predictors (including land use, road, emissions, elevation) in the grid cell containing site i in the climatic region k;
- ε_{ijk} is the error term in the grid cell containing site i on a day j in the climatic region k.

2.6.2. Stage 2

Using the stage 1 models fits, $PM_{2.5}$ was estimated in grid cells with AOD, but without $PM_{2.5}$ monitoring data.

2.6.3. Stage 3

In grid cells without AOD available a generalized additive mixed model with spatial smoothing (thin plate spline) was applied to generate grid cell predictions of $PM_{2.5}$.

$$PredPM2.5_{cj} = (\alpha + u_c) + (\beta + v_c)MeanPM2.5_{cj} + S(X_c, Y_c)_{k(j)} + \varepsilon_{cj}$$

- Where, $PredPM2.5_{cj}$ is the predicted $PM_{2.5}$ concentration from Stage 2 (cell c, day j);
- α and u_c are the fixed and random (cell specific; day and climatic region levels) intercepts;
- $MeanPM2.5_{cj}$ is the average $PM_{2.5}$ concentration in grid cell c and day j, computed from all stations within 60 km from the cell centroid;
- β_1 and v_c are fixed and cell-specific random slopes for $Mean\mathrm{PM}_{2.5}$;
- $S(X_c, Y_c)_{k(j)}$ is a smooth function (thin plate spline) of the location (computed as a thin-plate spline of latitude (X) and longitude (Y) of the cell centroid c) specific to the bi-monthly period $k(j)$ in which day j falls (a separate spatial smooth was fitted for each bi-monthly period).

2.6.4. Stage 4

In the fourth stage $PM_{2.5}$ predictions are estimated at the local level. First the residuals from the stage 1 model at each monitoring site were regressed against the local spatial and temporal variables at each monitoring site using machine learning techniques (Mitchell, 1997). The modelled residuals were then added to $PM_{2.5}$ predictions from the global model (stage 3) resulting in stage 4 local $PM_{2.5}$ estimates. For stage 4 we defined the following support vector machine (SVM) algorithm:

$$ResPM2.5_{ij} = \sum_m \alpha_m K\left(X_{m_{ij}}, x\right) + \varepsilon$$

- Where, $ResPM2.5_{ij}$ is the residual of the observed minus the predicted $PM_{2.5}$ concentration (site i, day j) from the stage 1 cross validation (CV) model;
- Each x is an instance from X_{mij} representing the spatial and temporal predictors (m) around the monitoring station i on day j;
- K is a kernel function (Gaussian) capturing nonlinearities and interactions among the predictors in predicting $ResPM2.5_{ij}$, and its parameters are chosen using 10-fold cross validation.

The 5 temporal predictors included are planetary boundary layer height, ambient temperature, wind speed, wind direction and

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precipitation. Spatial predictors included in this step are emissions (agriculture, household, industry, traffic and wood smoke), major and all road density, land use (residential, industry, urban green, total build up, agriculture and natural) within a 100 m buffer around the monitoring site, NDVI, elevation, and distance to nearest major road.

Support Vector Machines (SVMs), first introduced in the nineties, are a classification algorithm aimed at categorizing new objects into two separate groups, based on their properties and a set of known examples (observed data), which are already classified (Vapnik, 1995). SVM trains a model to assign new unseen objects into a certain category by creating a linear partition of the feature space into two categories. It places an object "above" or "below" the separation plane, based on features in the new unseen objects, leading to a categorization. SVMs has been extended to more complex situations, such as nonlinear dependencies and continuous outcomes, by introducing various types of non-linear decision boundaries, and training the SVM to identify the best linear separating hyperplanes in high-dimensional vector spaces (Vapnik et al., 1997).

Statistical analyses have been performed with the R statistical software, version 3 (R Development Core Team; http://R-project.org) and SPSS version 24. Geospatial analysis was performed with ESRI ArcGIS 10.

2.7. Statistical performance

We assessed the performance of the models (stage 1, 3 and 4) by reporting statistics of the regression between observed and predicted $PM_{2.5}$ (n = 99), for the full model and for the spatial and temporal components of the models. To avoid possible overfitting of the mixed modelling framework we checked the suitability of the spatial and temporal predictors/covariates by screening, prior to entering the model, on their partial correlations of determination with $PM_{2.5}$. We also performed a ten-fold cross validation (CV) by building models on 90% of the $PM_{2.5}$ monitoring sites, and evaluating them on the other 10%. We repeated this 10 times. In summary we report the following statistics by year:

- Total R^2, by regressing the predicted versus observed $PM_{2.5}$;
- Total CV R^2, a 10-fold cross validation of predicted versus observed $PM_{2.5}$;
- Total root mean square error (RMSE) in $\mu g/m^3$, representing a measure of prediction error;
- Spatial R^2, by regressing the annual average observed and predicted $PM_{2.5}$ concentrations, representing the explained fraction of spatial variation (annual average concentrations were calculated by averaging the daily predicted and observed $PM_{2.5}$ concentrations in each cell);
- Temporal R^2, by subtracting the annual average from both the observed and predicted daily $PM_{2.5}$ concentrations and regressing the daily observed versus predicted $PM_{2.5}$ deviations, representing the fraction of temporal variation in daily $PM_{2.5}$ concentration for all days at all monitoring stations;
- Spatial and temporal RMSE, intercepts and slopes, reporting the prediction errors, intercepts and slopes for the spatial and temporal components of $PM_{2.5}$.

3. Results and discussion

3.1. Imputation of $PM_{2.5}$ measurement data

We found high coefficients of determination between PM_{10} and $PM_{2.5}$ at the 10 co-located sites (range R^2's between 0.69 and 0.98; Table A.2). A high coefficient of determination ($R^2 = 0.89$) with a relationship close to the 1:1 line ($y = 1.36 + 0.88 \times x$) was found when applying, at each of the 10 co-located sites, the $PM_{2.5}/PM_{10}$ relationship from the nearest co-located site and comparing imputed $PM_{2.5}$ with measured $PM_{2.5}$ (see Figure A.3). This robust relationship gave us confidence of using imputed $PM_{2.5}$ estimates at the 89 PM_{10} monitoring sites. Restricting the comparison to $PM_{2.5}$ concentrations up to 40 $\mu g/m^3$ (for both imputed and observed) made no real difference to the performance ($R^2 = 0.87$). We applied the regression equation from the nearest co-located site to impute daily $PM_{2.5}$ concentrations at the 89 PM_{10}-only locations (Table A.2). From here on, the $PM_{2.5}$ data include the $PM_{2.5}$ imputed concentrations, for a total of 99 sites.

3.2. Global (1 × 1km) models

Table 1 presents the global model results from 2003 to 2013 for stage 1, showing performance statistics of the daily predicted $PM_{2.5}$ concentrations against the daily observed $PM_{2.5}$ concentrations at the monitoring sites including the 10-fold cross validation. Table A.3 shows, by year, the number of PM2.5 sites used and the minimum, maximum and mean number of daily observations. Predicted $PM_{2.5}$ concentrations at a 1 × 1km resolution were strongly correlated with observed $PM_{2.5}$ reflected in total R^2 (0.784–0.912), spatial R^2 (0.604–0.793) and temporal R^2 (0.819–0.925). The 10-fold cross validation also showed strong coefficients of determination (total CV R^2; 0.642 to 0.816, spatial CV R^2; 0.592 to 0.802, temporal CV R^2; 0.639 to 0.831) giving confidence in the robustness of the model fits. Total and temporal RMSE were very similar ranging from 2.4 to 4.5 $\mu g/m^3$, whereas the spatial RMSE was lower ranging from 1.5 to 2.1 $\mu g/m^3$. The slope was close to 1 for all years indicating an almost 1:1 relationship between predicted and observed $PM_{2.5}$ (slopes between 1.02 and 1.04). Table A.4 shows the models for each year reporting the intercept and the beta for predictor variables. In line with the

Table 1
Global model performance statistics over Switzerland (1 × 1km) stage 1.

Year	R^2	Intercept	Slope	RMSE	CV R^2	CV RMSE	Spatial R^2	Spatial RMSE	CV Spatial R^2	Temporal R^2	Temporal RMSE	CV Temporal R^2
2003	0.850	19.098	1.028	4.470	0.763	5.783	0.662	2.006	0.707	0.868	4.001	0.778
2004	0.843	16.852	1.034	3.460	0.698	4.927	0.724	1.581	0.761	0.856	3.105	0.700
2005	0.851	17.440	1.040	3.302	0.642	5.119	0.772	1.486	0.773	0.862	2.961	0.639
2006	0.912	14.736	1.017	3.429	0.816	4.804	0.793	1.639	0.802	0.925	3.015	0.831
2007	0.845	14.320	1.031	3.721	0.747	4.723	0.749	1.511	0.754	0.859	3.364	0.762
2008	0.865	12.801	1.019	3.557	0.815	4.080	0.716	1.532	0.751	0.881	3.210	0.830
2009	0.784	13.150	1.032	3.211	0.698	3.710	0.604	1.736	0.592	0.819	2.709	0.734
2010	0.801	13.622	1.033	3.125	0.707	3.797	0.646	1.698	0.653	0.834	2.624	0.746
2011	0.814	15.423	1.028	4.165	0.732	4.957	0.665	2.125	0.670	0.842	3.564	0.761
2012	0.825	13.897	1.031	3.785	0.735	4.724	0.687	1.859	0.660	0.845	3.314	0.760
2013	0.832	13.145	1.034	2.874	0.713	3.844	0.655	1.497	0.676	0.860	2.449	0.740

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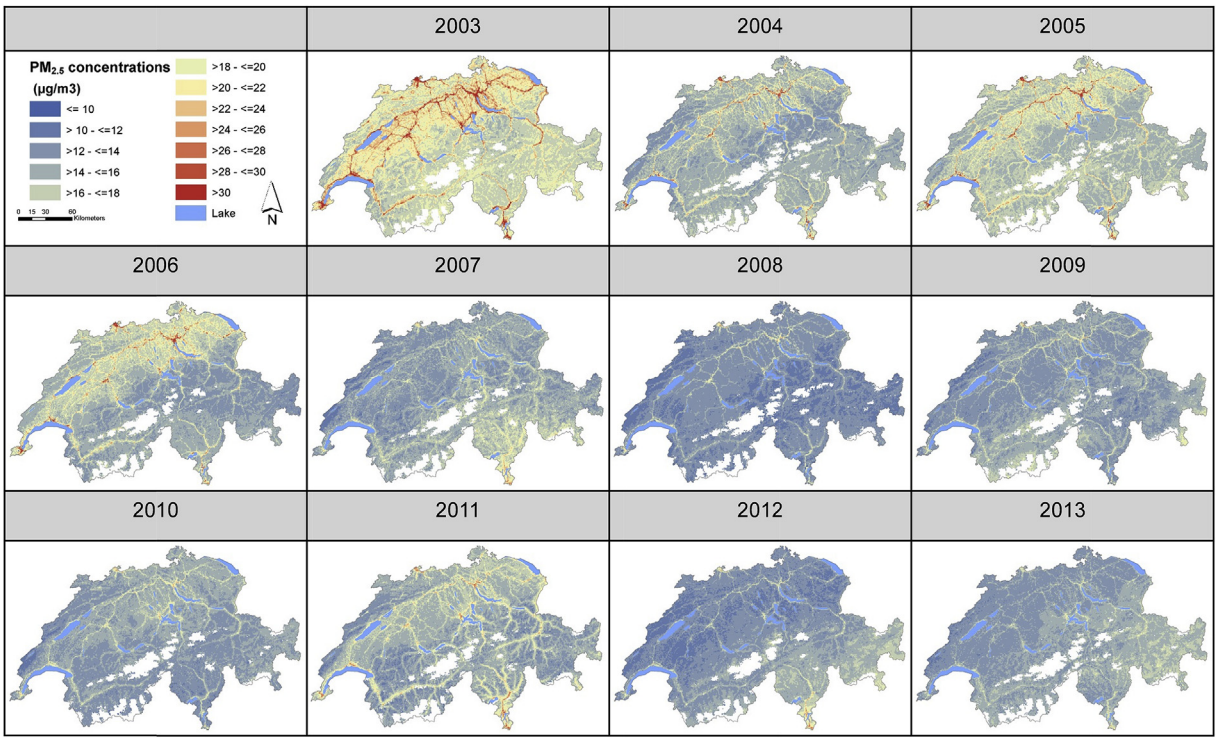

Fig. 1. Annual mean $PM_{2.5}$ concentrations ($\mu g/m^3$) for 2003 to 2013 at appr 1 × 1 km grid cells aggregated from daily estimates.

decline of $PM_{2.5}$ concentrations over time, the intercept of the models reduced from 19.1 in 2003 to 13.1 $\mu g/m^3$ in 2013. Most predictor variables had the same sign in the models, except for temperature and agricultural land use which exhibited − and + signs.

Fig. 1 shows the annual mean $PM_{2.5}$ concentrations estimated by the global models at a 1 × 1km resolution (AOD cells) (stage 3). The annual mean was calculated by averaging the estimated daily $PM_{2.5}$ concentrations by year. The maps show the gradual decline of annual mean $PM_{2.5}$ concentrations from 2003 to 2008 after which the concentrations stabilised. The declining trend is also observed in Figure A.4 where the national average of the measured annual mean $PM_{2.5}$ concentrations (at 10 stations), the national average measured including the imputed $PM_{2.5}$ concentrations (at 99 stations) and the average of all $PM_{2.5}$ estimates at 1 × 1km cells ≤ 1000 m altitude across Switzerland (model stage 3) are plotted. The 3 lines show similar trends over time with a decline in $PM_{2.5}$ concentrations (both measured and estimated) until 2008, after which it stabilised to just under 15 $\mu g/m^3$. The estimated $PM_{2.5}$ concentrations below 1000 m altitude mostly follow the measured $PM_{2.5}$ concentrations, both temporally and quantitatively, for example following the dip in concentrations in 2008. Performance statistics for stage 3 are presented in Table A.5 showing robust relationships between daily predicted 1 × 1km $PM_{2.5}$ and measured $PM_{2.5}$ concentrations by year (R^2 ranging between 0.730 and 0.872 and slopes close to 1).

To illustrate the daily variation in both the observed and predicted $PM_{2.5}$ concentrations, Figure A.5 shows for all years, the daily averages of observed $PM_{2.5}$ at all monitoring sites and the daily averages of predicted $PM_{2.5}$ over all 1 × 1km cells (Stage 3). The graph shows that predicted $PM_{2.5}$ followed the daily observed trend in $PM_{2.5}$ concentrations well, staying below the maximum and above the minimum measured $PM_{2.5}$ concentrations.

3.3. Local (100 × 100 m) models

Table 2 shows the statistical performance of the local models by year. Compared with the performance statistics for stage 1 (Table 1) we found an improvement in the total, spatial and temporal model R^2's together with a reduction of the associated RMSE. The improvement in the R^2 of the spatial models is most striking, changing from 0.604 to 0.793 in stage 1 (Table 1) to 0.917 to 0.973 in stage 4 (Table 2). Like in stage 1 slopes are very close to 1 (mean slope 1.01). Figure A.6 shows the improvement of the model fit after we applied the local model. For 2003 we found that the R^2 improves from 0.763 (CV R^2) to 0.899 (R^2) when compared to the measured $PM_{2.5}$ concentrations. An example of stage 4 daily estimated $PM_{2.5}$ concentrations is shown in Fig. 2, where 4 consecutive days (30 July − 2 August 2003) are mapped at a 100 × 100 m resolution. Temporal variations in estimated $PM_{2.5}$ concentrations are observed in line with observed $PM_{2.5}$ (see inset graph in Fig. 2) with 30 and 31 July generally being low pollution days with pollution increasing during 1 and 2 August, possible due to the Swiss national day fireworks. Also spatial patterns vary during the 4 days with estimated $PM_{2.5}$ levels increasing in the north-east of Switzerland during the 4 days.

The presented model results for Switzerland are comparable with results of studies conducted in northeastern USA (Kloog et al., 2014) and Mexico (Just et al., 2015) using a similar methodology. Kloog et al. (2014) developed $PM_{2.5}$ daily models for northeastern USA for 2003 to 2011 with an average CV R^2 of 0.88 and a mean spatial and temporal R^2s both of 0.87 at the 1 × 1km scale, which are slightly higher than model performance in Switzerland (mean CV Total $R^2 = 0.73$, Spatial $R^2 = 0.71$, temporal $R^2 = 0.86$). In Mexico, a $PM_{2.5}$ model for 2004 to 2014 yielded a similar mean CV R^2 of 0.72 (Just et al., 2015). A recent study conducted in China estimated daily $PM_{2.5}$ concentrations at the national scale using a generalized regression neural network (R = 0.81) between February 2013 and December 2014 (Li et al., 2017).

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Table 2
Local model performance statistics (at 100 m level) stage 4.

Year	Total R^2	Slope	RMSE	Spatial R^2	Spatial RMSE	Temporal R^2	Temporal RMSE
2003	0.899	1.017	3.512	0.973	0.608	0.891	3.459
2004	0.899	1.011	2.774	0.960	0.639	0.889	2.701
2005	0.893	1.015	2.794	0.960	0.665	0.882	2.716
2006	0.941	0.992	2.771	0.961	0.697	0.939	2.684
2007	0.886	1.019	3.175	0.970	0.550	0.877	3.128
2008	0.910	1.014	2.933	0.950	0.668	0.907	2.859
2009	0.858	1.007	2.592	0.917	0.812	0.847	2.472
2010	0.876	1.025	2.468	0.922	0.786	0.868	2.346
2011	0.894	1.014	3.107	0.971	0.620	0.882	3.042
2012	0.881	1.012	3.018	0.928	0.872	0.872	2.913
2013	0.893	1.004	2.274	0.955	0.578	0.884	2.197

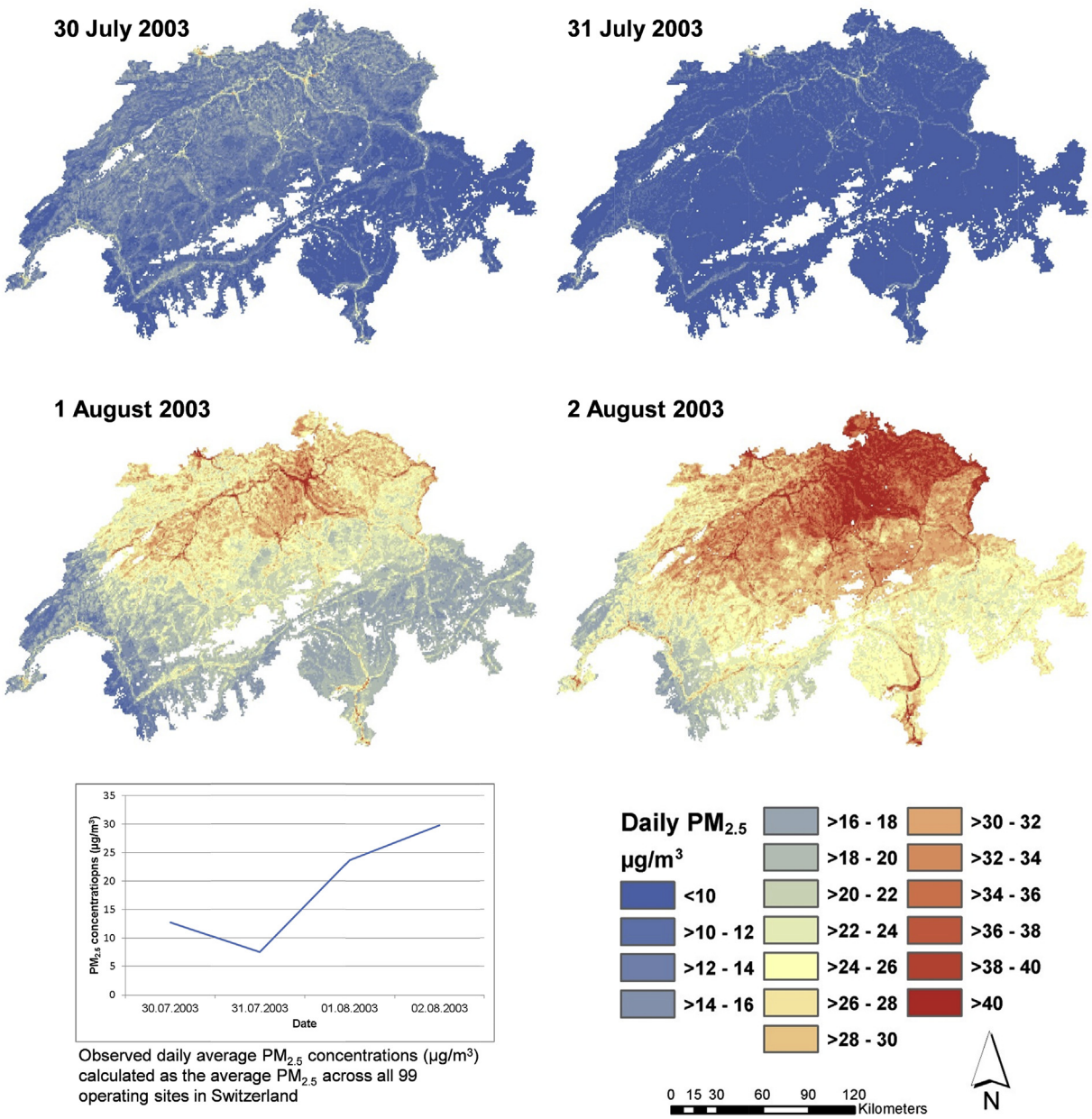

Fig. 2. Estimated daily average $PM_{2.5}$ concentrations after stage 4 (bottom) at the local scale (100 × 100 m) for 4 consecutive days in 2003 across Switzerland.

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The method presented here is a good alternative in estimating historical exposures (here from 2003 to 2013) when compared to for example backextrapolation. The $PM_{2.5}$ estimates are flexible, both temporally and spatially. The daily predictions can be aggregated to any required exposure time window to fit with the specific epidemiological research question (i.e. pregnancy trimesters for birth cohorts, multiple days prior to date of lung function reading in respiratory research, long term periods for lung cancer, etc.). The method also allows for predictions to be calculated at different spatial scales; at address locations from cohort participants to for example at census areas in large population studies. Fig. 1 shows annual $PM_{2.5}$ estimates at a 1 × 1km level, but similar maps can be produced at finer temporal and spatial scales.

Most predictor variables exhibited consistent signs across the study period (See Table A.4), except for temperature and agricultural land use having different signs across the years modelled. It can be argued, however, that these two variables can both have negative and positive associations with $PM_{2.5}$ concentrations. $PM_{2.5}$ concentrations generally increases with higher temperatures in the summer, as these are often associated with high pressure areas, with calm wind conditions, resulting in stagnant air mass and thus higher built up concentrations. In the winter, Switzerland has meteorological conditions that trigger inversion also under high pressure systems, but with low temperatures, showing the opposite tendency compared to the earlier described summer conditions. Moreover, heating is an important source of air pollution during cold winter periods. Agricultural land use can be seen as an air pollution source with emissions arising from agricultural activity (e.g. machinery, ploughing etc.), but this does not apply in all seasons, nor to areas under permanent crop (similar to natural areas).

3.4. Using PM_{10} measurements to impute $PM_{2.5}$ estimates used for modelling

Ten $PM_{2.5}$ monitoring sites were operating, at some point, during the eleven years study period. This was not deemed sufficient for our modelling purposes and we therefore gap-filled using information from the 89 PM_{10} monitoring sites also operating during that time period. At co-located sites (all of the 10 $PM_{2.5}$ stations were co-located with PM_{10}) we evaluated the relationship between $PM_{2.5}$ and PM_{10} concentrations and found robust temporal relationships. This gave us confidence to use these linear regression relationships (by year) to impute $PM_{2.5}$ concentrations based on measured PM_{10} using the nearest co-located site regression equation. Many studies have examined the $PM_{2.5}/PM_{10}$ relationship, but few have used this information to impute $PM_{2.5}$ concentrations in subsequent modelling. Yoval and Broday (Yuval and Broday, 2014) tested two imputation methods in Israel and found that the $PM_{10}/PM_{2.5}$ ratio method, similar as what was used here, performed the best, yielding a cross-validated R^2 of 0.76 when comparing observed daily $PM_{2.5}$ concentrations with imputed daily $PM_{2.5}$ concentrations. Gehrig and Buchmann (2003) further found mean daily $PM_{2.5}/PM_{10}$ ratios at 7 sites across Switzerland (1998–2001) ranging from 0.55 to 0.79 with R^2's for all years ranging from 0.85 to 0.98. These ratios are similar to the ratios we report: yearly ratios between 0.53 and 0.93, and yearly R^2's of daily $PM_{2.5}/PM_{10}$ ranging between 0.70 and 0.98 (2003–2013).

SVM is in principle a much more flexible method than conventional regression methods (Vapnik, 1995) SVM, a relatively new method in the air pollution exposure modelling scene, has recently been applied in Italy, where the implemented SVM model greatly improved the model fitting, explaining 71% of the total PM_{10} variation (Stafoggia et al., 2017).

Despite the high performance of our models we realise that there are limitations. For the majority of predictor data we used one data set to represent the whole duration of the study period. The exceptions where meteorological data (daily) and emission data (2 time periods), but for land use, road data and NDVI we used data representing respectively 2006, 2008 and 2014. We made the assumption that the three environmental and source features these data represent did not change much over the duration of the study period and in terms of land use and road data were in fact representing the middle point of the study period. In addition, NDVI was collected for the year 2014 as it was the only year with cloud free conditions on the days the LANDSAT 8 satellite orbited Switzerland.

During certain meteorological conditions (inversion) occurring during the winter, parts of Switzerland, and in particular the Swiss Platteau (or Schweizer Mittelland), suffer from foggy or low cloud days, sometimes lasting for days at a time. Days when these conditions occur will not have an AOD measurement due to the cloud cover. There is thus the chance that $PM_{2.5}$ concentrations estimated during those particular conditions are under- or over estimated. We accounted for this by applying a generalized additive mixed model with spatial smoothing to generate grid cell predictions for those grid cells where AOD was missing.

Despite the unique nature of Switzerland's challenges (e.g. elevation, climate) the global models we developed are robust explaining on average 73% of the total,71% of the spatial and 75% of the temporal variation (all cross validated) at the 1 km level. Models at the local level (100 m) explained on average 89% (total) 95% (spatial) and 89% (temporal) of the variation in $PM_{2.5}$ concentrations. The presented modelling approach may be particularly appealing for exposure assessment in large cohort studies on long-term effects of air pollution (Beelen et al., 2014; Pedersen et al., 2013; Gehring et al., 2013; Adam et al., 2015). It offers the opportunity to derive annual means over decades for any grid cell, using coherent methods and routine data fully capturing the annual and even seasonal patterns of pollution. Typically, backextrapolations relied on fewer data (Pedersen et al., 2013; Gehring et al., 2013; Adam et al., 2015) with a certain loss of specific annual and seasonal patterns of exposure. Indeed, the derivation of daily values (step 1) offers even the opportunity to retrospectively investigate short-term effects of air pollution on a national level, or to derive individually assigned windows of exposure such as during pregnancy.

The reliance on good quality monitoring network data is also a major advantage compared to the traditional approaches where extensive local measurements were needed to model pollution at only one or a few cities or regions (Aguilera et al., 2015, 2016), thus study participants moving out of those modelled regions are lost to follow up. Moreover, updates of such models need repeated investments in measurements.

4. Conclusions

We successfully estimated daily levels of $PM_{2.5}$ in Switzerland from 2003 to 2013 at a fine resolution, the predictions of which can facilitate environmental epidemiological research at fine temporal and spatial scales. We tested and succeeded in applying a novel approach to borrow information from an extensive PM_{10} monitoring network to inform the relative sparse $PM_{2.5}$ data set. This specifically addresses a major obstacle for other countries aiming to model $PM_{2.5}$, as these sites are often underrepresented, especially going back in time.

Conflict of interest

All authors declare no actual or potential conflict of financial or other interests.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.envpol.2017.10.025.

References

Adam, M., Schikowski, T., Carsin, A.E., Cai, Y., Jacquemin, B., Sanchez, M., Vierkötter, A., Marcon, A., Keidel, D., Sugiri, D., Al Kanani, Z., Nadif, R., Siroux, V., Hardy, R., Kuh, D., Rochat, T., Bridevaux, P.-O., Eeftens, M., Tsai, M.-Y., Villani, S., Phuleria, H.C., Birk, M., Cyrys, J., Cirach, M., de Nazelle, A., Nieuwenhuijsen, M.J., Forsberg, B., de Hoogh, K., Declerq, C., Bono, R., Piccioni, P., Quass, U., Heinrich, J., Jarvis, D., Pin, I., Beelen, R., Hoek, G., Brunekreef, B., Schindler, C., Sunyer, J., Krämer, U., Kauffmann, F., Hansell, A.L., Künzli, N., Probst-Hensch, N., 2015. Adult lung function and long-term air pollution exposure. ESCAPE: a multicentre cohort study and meta-analysis. Eur. Respir. J. 45 (1), 38–50.

Aguilera, I., Eeftens, M., Meier, R., Ducret-Stich, R.E., Schindler, C., Ineichen, A., Phuleria, H.C., Probst-Hensch, N., Tsai, M.-Y., Künzli, N., 2015. Land use regression models for crustal and traffic-related PM2.5 constituents in four areas of the SAPALDIA study. Environ. Res. 140, 377–384.

Aguilera, I., Dratva, J., Caviezel, S., Burdet, L., de Groot, E., Ducret-Stich, R.E., Eeftens, M., Keidel, D., Meier, R., Perez, L., Rothe, T., Schaffner, E., Schmit-Trucksäss, A., Tsai, M.-Y., Schindler, C., Künzli, N., Probst-Hensch, N., 2016. Particulate matter and subclinical atherosclerosis: associations between different particle sizes and sources with carotid intima-media thickness in the SAPALDIA study. Environ. Health Perspect. 124 (11), 1700–1706.

Beelen, R., Raaschou-Nielsen, O., Stafoggia, M., Andersen, Z.J., Weinmayr, G., Hoffmann, B., Wolf, K., Samoli, E., Fischer, P., Nieuwenhuijsen, M., 2014. Effects of long-term exposure to air pollution on natural-cause mortality: an analysis of 22 European cohorts within the multicentre ESCAPE project. Lancet 383 (9919), 785–795.

Brunekreef, B., Künzli, N., Pekkanen, J., Annesi-Maesano, I., Forsberg, B., Sigsgaard, T., Keuken, M., Forastiere, F., Barry, M., Querol, X., 2015. Clean air in Europe: beyond the horizon? Eur. Respir. J. 45 (1), 7–10.

Cesaroni, G., Forastiere, F., Stafoggia, M., Andersen, Z.J., Badaloni, C., Beelen, R., Caracciolo, B., de Faire, U., Erbel, R., Eriksen, K.T., 2014. Long term exposure to ambient air pollution and incidence of acute coronary events: prospective cohort study and meta-analysis in 11 European cohorts from the ESCAPE Project. BMJ 348, f7412.

Dee, D., Uppala, S., Simmons, A., Berrisford, P., Poli, P., Kobayashi, S., Andrae, U., Balmaseda, M., Balsamo, G., Bauer, P., 2011. The ERA-Interim reanalysis: configuration and performance of the data assimilation system. Q. J. R. Meteorological Soc. 137 (656), 553–597.

Eeftens, M., Phuleria, H.C., Meier, R., Aguilera, I., Corradi, E., Davey, M., Ducret-Stich, R., Fierz, M., Gehrig, R., Ineichen, A., Keidel, D., Probst-Hensch, N., Ragettli, M.S., Schindler, C., Künzli, N., Tsai, M.-Y., 2015. Spatial and temporal variability of ultrafine particles, NO2, PM2.5, PM2.5 absorbance, PM10 and PMcoarse in Swiss study areas. Atmos. Environ. 111, 60–70.

FOEN, 2011. NO2 Ambient Concentrations in Switzerland Modelling Results for 2005, 2010, 2015. Federal Office for the Environment, Bern, p. 68.

FOEN, 2013. PM10 and PM2.5 Ambient Concentrations InSwitzerland, Modelling Results for 2005, 2010, 2020. Federal Office for the Environment, Bern, p. 83.

FSO, 2016. Switzerland's Population 2015. Neuchatel.

Gehrig, R., Buchmann, B., 2003. Characterising seasonal variations and spatial distribution of ambient PM10 and PM2.5 concentrations based on long-term Swiss monitoring data. Atmos. Environ. 37 (19), 2571–2580.

Gehring, U., Gruzieva, O., Agius, R.M., Beelen, R., Custovic, A., Cyrys, J., Eeftens, M., Flexeder, C., Fuertes, E., Heinrich, J., 2013. Air pollution exposure and lung function in children: the ESCAPE project. Environ. Health Perspect. (Online) 121 (11–12), 1357.

Gulliver, J., de Hoogh, K., Hoek, G., Vienneau, D., Fecht, D., Hansell, A., 2016. Back-extrapolated and year-specific NO2 land use regression models for Great Britain - do they yield different exposure assessment? Environ. Int. 92–93, 202–209.

Just, A.C., Wright, R.O., Schwartz, J., Coull, B.A., Baccarelli, A.A., Tellez-Rojo, M.M., Moody, E., Wang, Y., Lyapustin, A., Kloog, I., 2015. Using high-resolution satellite aerosol optical depth to estimate daily PM2.5 geographical distribution in Mexico city. Environ. Sci. Technol. 49 (14), 8576–8584.

Kloog, I., Nordio, F., Coull, B.A., Schwartz, J., 2012. Incorporating local land use regression and satellite aerosol optical depth in a hybrid model of spatiotemporal PM2.5 exposures in the mid-atlantic states. Environ. Sci. Technol. 46 (21), 11913–11921.

Kloog, I., Chudnovsky, A.A., Just, A.C., Nordio, F., Koutrakis, P., Coull, B.A., Lyapustin, A., Wang, Y., Schwartz, J., 2014. A new hybrid spatio-temporal model for estimating daily multi-year PM 2.5 concentrations across northeastern USA using high resolution aerosol optical depth data. Atmos. Environ. 95, 581–590.

Li, T., Shen, H., Zeng, C., Yuan, Q., Zhang, L., 2017. Point-surface fusion of station measurements and satellite observations for mapping PM2.5 distribution in China: methods and assessment. Atmos. Environ. 152, 477–489.

Lyapustin, A., Martonchik, J., Wang, Y., Laszlo, I., Korkin, S., 2011. Multiangle implementation of atmospheric correction (MAIAC): 1. Radiative transfer basis and look-up tables. J. Geophys. Res. Atmos. 116 (D3).

Lyapustin, A., Wang, Y., Laszlo, I., Kahn, R., Korkin, S., Remer, L., Levy, R., Reid, J., 2011. Multiangle implementation of atmospheric correction (MAIAC): 2. Aerosol algorithm. J. Geophys. Res. Atmos. 116 (D3).

Mitchell, T., 1997. Machine Learning. McGraw Hill.

Panasevich, S., Håberg, S.E., Aamodt, G., London, S.J., Stigum, H., Nystad, W., Nafstad, P., 2016. Association between pregnancy exposure to air pollution and birth weight in selected areas of Norway. Archives Public Health 74 (1), 26.

Pedersen, M., Giorgis-Allemand, L., Bernard, C., Aguilera, I., Andersen, A.-M.N., Ballester, F., Beelen, R.M.J., Chatzi, L., Cirach, M., Danileviciute, A., Dedele, A., Eijsden, M. v., Estarlich, M., Fernández-Somoano, A., Fernández, M.F., Forastiere, F., Gehring, U., Grazuleviciene, R., Gruzieva, O., Heude, B., Hoek, G., Hoogh, K. d., van den Hooven, E.H., Håberg, S.E., Jaddoe, V.W.V., Klümper, C., Korek, M., Krämer, U., Lerchundi, A., Lepeule, J., Nafstad, P., Nystad, W., Patelarou, E., Porta, D., Postma, D., Raaschou-Nielsen, O., Rudnai, P., Sunyer, J., Stephanou, E., Sørensen, M., Thiering, E., Tuffnell, D., Varró, M.J., Vrijkotte, T.G.M., Wijga, A., Wilhelm, M., Wright, J., Nieuwenhuijsen, M.J., Pershagen, G., Brunekreef, B., Kogevinas, M., Slama, R., 2013. Ambient air pollution and low birthweight: a European cohort study (ESCAPE). Lancet Respir. Med. 1 (9), 695–704.

Raaschou-Nielsen, O., Andersen, Z.J., Beelen, R., Samoli, E., Stafoggia, M., Weinmayr, G., Hoffmann, B., Fischer, P., Nieuwenhuijsen, M.J., Brunekreef, B., 2013. Air pollution and lung cancer incidence in 17 European cohorts: prospective analyses from the European study of cohorts for air pollution effects (ESCAPE). Lancet Oncol. 14 (9), 813–822.

Stafoggia, M., Schwartz, J., Badaloni, C., Bellander, T., Alessandrini, E., Cattani, G., de' Donato, F., Gaeta, A., Leone, G., Lyapustin, A., Sorek-Hamer, M., de Hoogh, K., Di, Q., Forastiere, F., Kloog, I., 2017. Estimation of daily PM10 concentrations in Italy (2006–2012) using finely resolved satellite data, land use variables and meteorology. Environ. Int. 99, 234–244.

Tétreault, L.-F., Doucet, M., Gamache, P., Fournier, M., Brand, A., Kosatsky, T., Smargiassi, A., 2016. Childhood exposure to ambient air pollutants and the onset of asthma: an administrative cohort study in Québec. Environ. Health Perspect. 124 (8), 1276–1282.

Vapnik, V., 1995. The Nature of Statistical Learning Theory. Springer, N.Y., 0-387-94559-8.

Vapnik, V., Golowich, S.E., Smola, A., 1997. Support vector method for function approximation, regression estimation, and signal processing. Adv. Neural Inf. Process. Syst. 281–287.

Yuval, Broday, D.M., 2014. Enhancement of PM2.5 exposure estimation using PM10 observations. Environ. Sci. Process. Impacts 16 (5), 1094–1102.

Appendix A

Modelling daily $PM_{2.5}$ concentrations at high spatio-temporal resolution across Switzerland

Kees de Hoogh, Harris Héritier, Massimo Stafoggia, Nino Künzli, Itai Kloog

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Material and Methods

1. Global and local predictor variables

Emissions; $PM_{2.5}$ emissions for the years 2005 and 2010 were obtained from Meteotest at a 200x200m grid, covering agriculture, household, industry, traffic and wood smoke emissions [11, 18]. For traffic, emission data was modelled using information about the Swiss road network, traffic intensity from the Swiss national traffic model and national emission factors. Industrial emissions were calculated by summing the stationary sources from NFR categories 1A1 "Energy Industries", 1A2 "Manufacturing Industries and Construction", 2 "Industrial Processes" plus emissions from crematories. Agricultural, household and wood smoke emissions were obtained from the Swiss Federal Office of Environment (FOEN) and were distributed across the relevant land uses.

Distance to nearest main road; Main roads were extracted from the sonBASE traffic database, linked to the VECTOR25 road network, by selecting roads with an Annual Average Daily Traffic (AADT) greater than 5000. The NEAR command in ArcGIS10 was used to calculate the distance to the nearest main road from the centroid of the 1x1km grid cell (global); and from the point location of the monitoring sites and the centroid of the 100x100 grid cell (both local).

Roads; road data originated from the 1:10,000 EuroStreets digital road network (version 3.1, based on TeleAtlas MultiNet TM for year-2008). The road data was aggregated into the sum of length of all and major roads (using the classification available in EuroStreets) in a 100x100m grid cell for the local predictor and to the 1x1km grid cell for the global predictor.

Elevation; we use the digital height model DHM25 at a 200m grid, based on the Swiss National Map 1:25 000 (Source: Swiss Federal Office of Topography). Average errors, when comparing model heights with measurements, are between 1.5 and 8m depending on the region. In addition to elevation, two more variables were calculated. The gradient or slope (in degrees) within the

200x200m cell was extracted using the SLOPE function in ArcGIS. The standard deviation of elevation within a 1x1km cell, depicting flat (low standard deviation) versus hilly or mountainous (high standard deviation) terrain was determined.

Land Use; the 100x100m European Corine Land Cover (CLC2006) data set was obtained. From the 44 land classes available in Corine, six main groups were extracted: residential (Corine class = 1 + 2; RES), industry or commercial (3; IND), urban green (10; URBGR), total built up (1-9; BUILT), agriculture (12 - 22; AGR) and semi-natural and forest (23 - 41; NAT). The percentage of each land use variable within each grid cell was calculated.

Meteorology; daily modelled planetary boundary layer data, daily temperature, wind speed, wind direction and precipitation at a ~10x10km resolution from 1 January 2003 till 31 December 2013 were obtained from the European Centre for Medium-Range Weather Forecasts (ECMWF). The meteorological variables are modelled through the ERA-Interim, the global atmospheric re-analysis [19]. The temperature variable was converted from Kelvin to degrees Celsius. Wind direction at 10 metres was calculated from the U_{10} and v_{10} components using the formula:

$$atan2\ (-U_{10}, -v_{10})\ x\ \frac{\pi}{180}$$

NDVI; A 30x30m raster depicting the Normalised Difference Vegetation Index (NDVI) was derived by combining 6 tiles, covering the extent of Switzerland, downloaded from the USGS EarthExplorer website (http://earthexplorer.usgs.gov). The tiles were taken with the LANDSAT 8 satellite between 8 June and 19 July 2014 without any cloud cover

2. Pre-processing of $PM_{2.5}$ and AOD data

During the cleaning process we removed 14% of records (11082 of 76762) with: 1) conflicting AOD and $PM_{2.5}$ values (AOD < 50th percentile and $PM_{2.5}$ > 90th percentile, or $PM_{2.5}$ < 50th percentile and AOD > 90th percentile) (6422 or 8%); 2) monitors with less than 30 daily observations ; and 3) with

AOD values >1.2 and those with uncertainty parameter (from MAIAC algorithm) outside the range 0-0.04 (3021 or 4%) and 4) where the grid cell falls in a lake or in snow covered mountain tops (NDVI <=0) (1642 or 2%).

Table A.1: Monitoring stations measuring daily $PM_{2.5}$ concentrations in Switzerland from 2003 to 2013

Name	Acronym	Years	Site Type
Bern-Bollwerk	beBER	2003-2013	Urban, traffic
Basel-Binningen	blBAS	2003-2013	Suburban
Chaumont	neCHA	2003-2006	Rural, < 1000m
Härkingen-A1	soHAE	2011-2013	Rural, autoroute
Rigi-Seebodenapl	szRIG	2007-2013	Rural, > 1000m
Lugano-Università	tiLUG	2003-2013	Urban
Magadino-Cadenazzo	tiMAG	2011-2013	Rural, > 1000m
Payerne	vdPAY	2003-2013	Rural, > 1000m
Dübendorf-Empa	zhDUE	2011-2013	Suburban
Zürich-Kaserne	zhZUE	2008-2013	Urban

Table A.2: Results of regression (correlation, slope, intercept and RMSE) between daily concentrations of $PM_{2.5}$ and PM_{10} per year at co-located monitoring sites in Switzerland between 2003 and 2013

Site Name	Year	Number of days	R^2	RMSE	Slope	Intercept
beBER	2003	362	0.907	3.773	0.579	2.102
beBER	2004	359	0.860	3.722	0.571	1.463
beBER	2005	349	0.844	3.799	0.530	2.708
beBER	2006	364	0.940	3.640	0.596	-0.674
beBER	2007	364	0.888	3.396	0.634	-0.957
beBER	2008	365	0.864	3.584	0.617	-0.109
beBER	2009	343	0.817	4.067	0.682	-0.504
beBER	2010	167	0.884	4.102	0.639	0.878
beBER	2011	91	0.912	3.591	0.890	-2.563
beBER	2012	82	0.875	3.194	0.733	-0.919
beBER	2013	90	0.888	4.005	0.711	-1.081
blBAS	2003	365	0.941	3.495	0.897	-3.147
blBAS	2004	366	0.908	3.487	0.814	-0.882
blBAS	2005	365	0.928	3.060	0.916	-2.611
blBAS	2006	363	0.944	3.659	0.843	-2.032
blBAS	2007	360	0.900	3.386	0.790	-1.387
blBAS	2008	362	0.914	2.814	0.848	-1.925
blBAS	2009	365	0.958	2.149	0.908	-2.506
blBAS	2010	169	0.936	3.259	0.886	-1.586
blBAS	2011	89	0.980	1.516	0.892	-2.388
blBAS	2012	89	0.972	1.664	0.868	-1.772
blBAS	2013	91	0.978	1.650	0.906	-2.164
neCHA	2003	356	0.890	2.380	0.733	0.141
neCHA	2004	360	0.802	2.304	0.657	0.681
neCHA	2005	344	0.864	2.128	0.796	-0.251
neCHA	2006	352	0.732	3.212	0.676	0.788
soHAE	2011	88	0.971	1.827	0.874	-2.610
soHAE	2012	92	0.706	5.144	0.591	1.296
soHAE	2013	90	0.926	3.021	0.831	-2.216
szRIG	2007	348	0.880	2.256	0.766	-0.345
szRIG	2008	351	0.686	2.943	0.532	1.669
szRIG	2009	352	0.833	2.222	0.696	0.057
szRIG	2010	144	0.950	1.909	0.892	-0.523
szRIG	2011	82	0.912	1.769	0.816	-0.384
szRIG	2012	88	0.894	1.805	0.780	-0.420
szRIG	2013	84	0.948	1.532	0.894	-0.693
tiLUG	2003	337	0.923	4.977	0.800	-0.779
tiLUG	2004	356	0.919	4.967	0.876	-3.344
tiLUG	2005	364	0.908	5.208	0.796	-1.949
tiLUG	2006	357	0.922	4.600	0.774	-1.680
tiLUG	2007	331	0.901	4.567	0.817	-2.253

tiLUG	2011	89	0.972	2.407	0.881	-2.676
tiLUG	2012	89	0.973	2.340	0.881	-2.613
tiLUG	2013	87	0.963	2.131	0.861	-1.411
tiMAG	2011	89	0.952	2.661	0.805	-1.704
tiMAG	2012	88	0.959	2.582	0.844	-2.387
tiMAG	2013	87	0.956	2.205	0.821	-1.361
vdPAY	2003	346	0.951	3.436	0.896	-3.066
vdPAY	2004	363	0.902	3.156	0.847	-2.126
vdPAY	2005	361	0.913	2.762	0.882	-2.476
vdPAY	2006	351	0.943	3.865	0.772	-0.995
vdPAY	2007	364	0.880	3.362	0.745	-1.898
vdPAY	2008	365	0.900	3.226	0.741	-1.935
vdPAY	2009	355	0.934	2.333	0.857	-2.860
vdPAY	2010	175	0.964	2.380	0.932	-3.427
vdPAY	2011	84	0.972	1.719	0.837	-2.045
vdPAY	2012	90	0.930	2.389	0.834	-1.827
vdPAY	2013	88	0.972	1.947	0.903	-1.682
zhDUE	2011	89	0.928	2.565	0.815	-1.210
zhDUE	2012	92	0.958	1.837	0.836	-1.728
zhDUE	2013	91	0.979	1.589	0.864	-1.584
zhZUE	2008	365	0.917	2.940	0.860	-2.182
zhZUE	2009	364	0.940	2.472	0.859	-2.581
zhZUE	2010	169	0.977	2.069	0.912	-2.610
zhZUE	2011	91	0.951	2.233	0.836	-1.830
zhZUE	2012	87	0.962	1.898	0.828	-1.303
zhZUE	2013	88	0.968	1.981	0.847	-1.116

Table A.3: Summary statistics for number of observations used in the stage 1 modelling

Year	Number of $PM_{2.5}$ sites	Number of daily observations		
		Minimum	Maximum	Mean
2003	43	67	159	121
2004	44	39	115	87
2005	42	33	131	93
2006	47	28	130	92
2007	66	30	187	116
2008	64	45	130	92
2009	63	44	126	86
2010	67	36	105	78
2011	78	58	172	127
2012	80	46	142	95
2013	77	36	133	87

Table A.4: Beta's for predictor variables (fixed effects per year)

Category	Variable	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
(Intercept)		19.098	16.852	17.440	14.736	14.320	12.801	13.150	13.622	15.423	13.897	13.145
Satellite	AOD	1.660	1.577	1.929	1.860	1.623	1.057	1.218	1.095	1.339	1.320	1.396
Meteorology	Boundary Layer Height	-3.658	-2.417	-3.638	-4.762	-2.399	-2.438	-1.621				-0.733
	Ambient Temperature	3.065	3.255	3.477	3.641	1.448	1.814	0.937		-2.395	0.243	
	Wind Speed	-0.884	-1.094	-0.655	-0.880	-0.674	-0.692	-0.218	-0.485	-0.707	-0.708	-0.506
	Precipitation	-0.509		-0.571	-0.477			-0.647	-0.477	-0.484	-0.347	-0.167
Greenness	NDVI	-0.658	-0.454	-0.365	-0.438	-0.922	-0.395	-0.687	-0.817	-0.574	-0.668	-0.970
Land use (perc)	Total Build up	1.087										
	Agriculture	1.406	0.325	0.386				-0.583			-0.453	-0.398
	Urban Green		-0.892	-1.069	-0.664		-0.379					
	Industry									0.235		
Emissions	Industry				0.212							
	Household	0.205	0.238	0.197	0.415	0.462	0.253		0.213	0.394		
	Traffic	0.945	1.071	1.076	0.498		0.738	0.385	0.508	0.489	0.505	
	Agricultural					0.447		0.475	0.306			
	Wood smoke						0.215					
Road/Traffic	Sum of all roads length	0.264	0.922	0.947	0.862							
	Sum of major roads length	0.596	0.159	0.261	0.261	0.574	0.405	0.701	0.376	0.371	0.428	0.538
	Distance to nearest main road		0.273					0.561		0.359	0.224	
Elevation	Elevation	-0.383	-0.771	-0.846	-0.716	-1.542	-1.050	-1.340	-1.122	-2.285	-1.218	-0.953
	Slope									0.482		
	Stand Dev elevation			0.143		1.492	0.560	0.747	0.215			

Table A.5: Performance statistics over Switzerland (1x1km) stage 3

Year	R^2	RMSE	Slope
2003	0.785	5.329	0.937
2004	0.735	4.483	0.918
2005	0.730	4.427	0.947
2006	0.872	4.127	0.957
2007	0.798	4.236	1.000
2008	0.843	3.828	0.970
2009	0.751	3.439	1.032
2010	0.778	3.298	1.024
2011	0.768	4.659	1.019
2012	0.800	4.035	1.023
2013	0.770	3.360	1.186

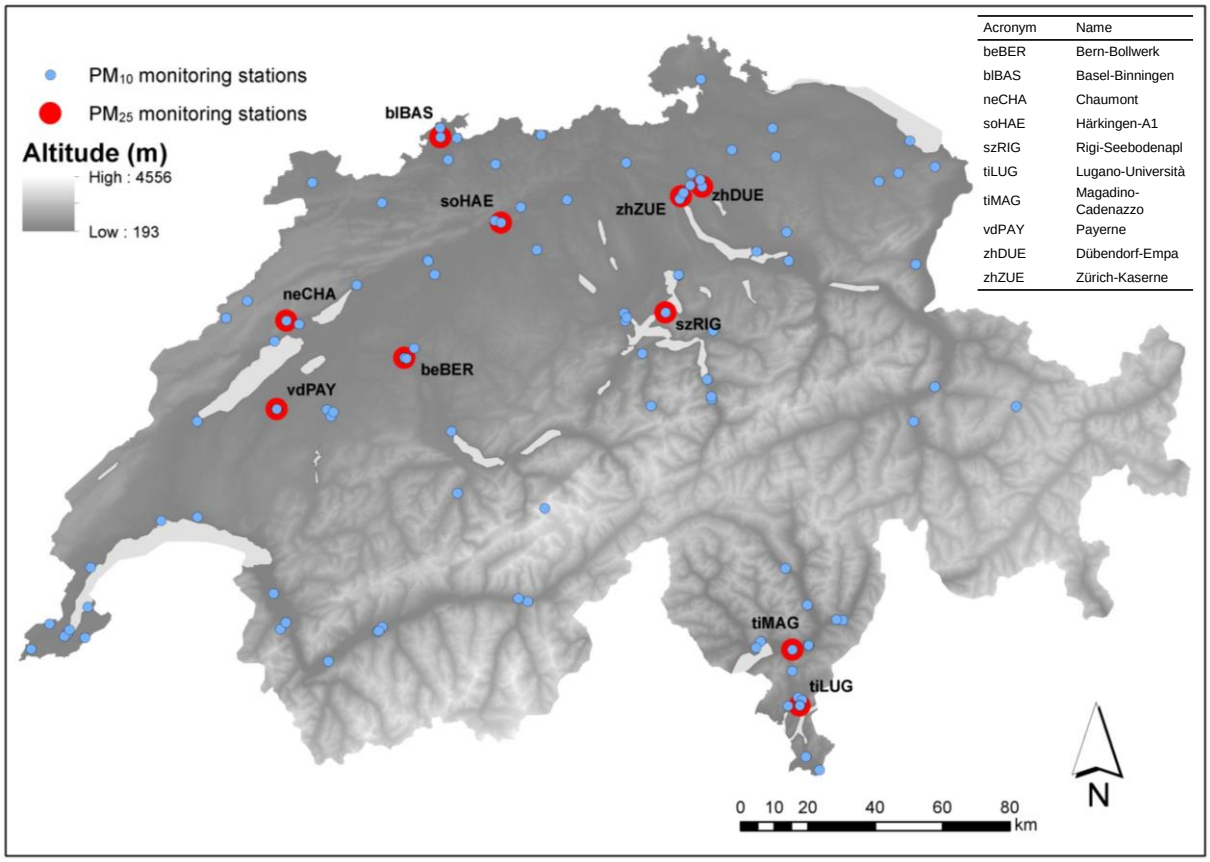

Figure A.1: $PM_{2.5}$ and PM_{10} monitoring locations in Switzerland; 2003 – 2013

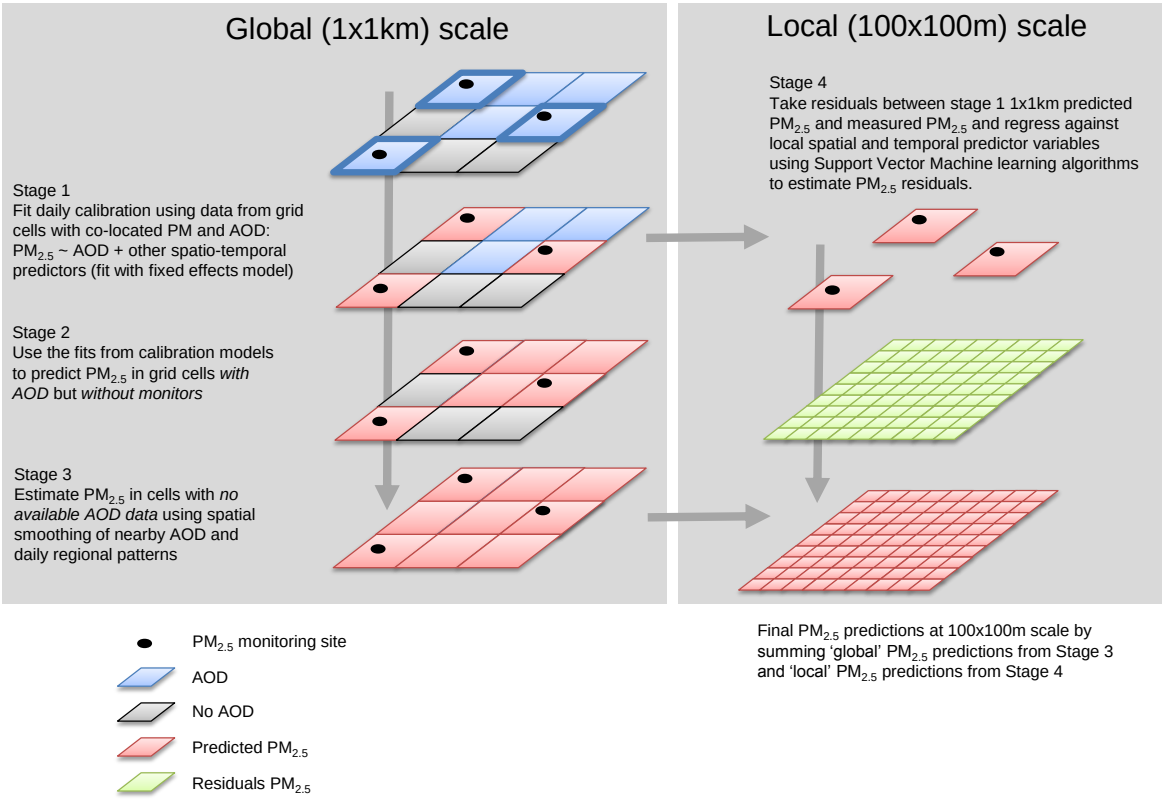

Figure A.2: Flow diagram depicting the 4 stages in the modelling framework

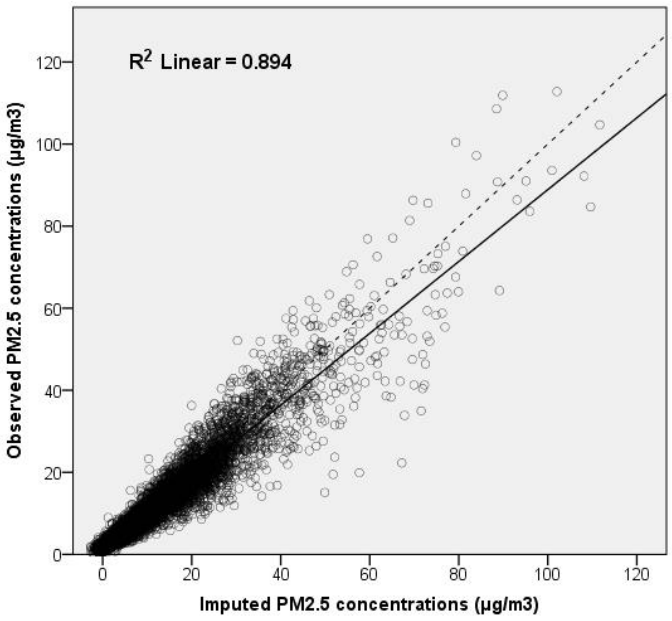

Figure A.3: Correlation of imputed and measured daily $PM_{2.5}$ concentrations, based on $PM_{2.5}/PM_{10}$ relationship at the nearest co-located site (calculated by year).

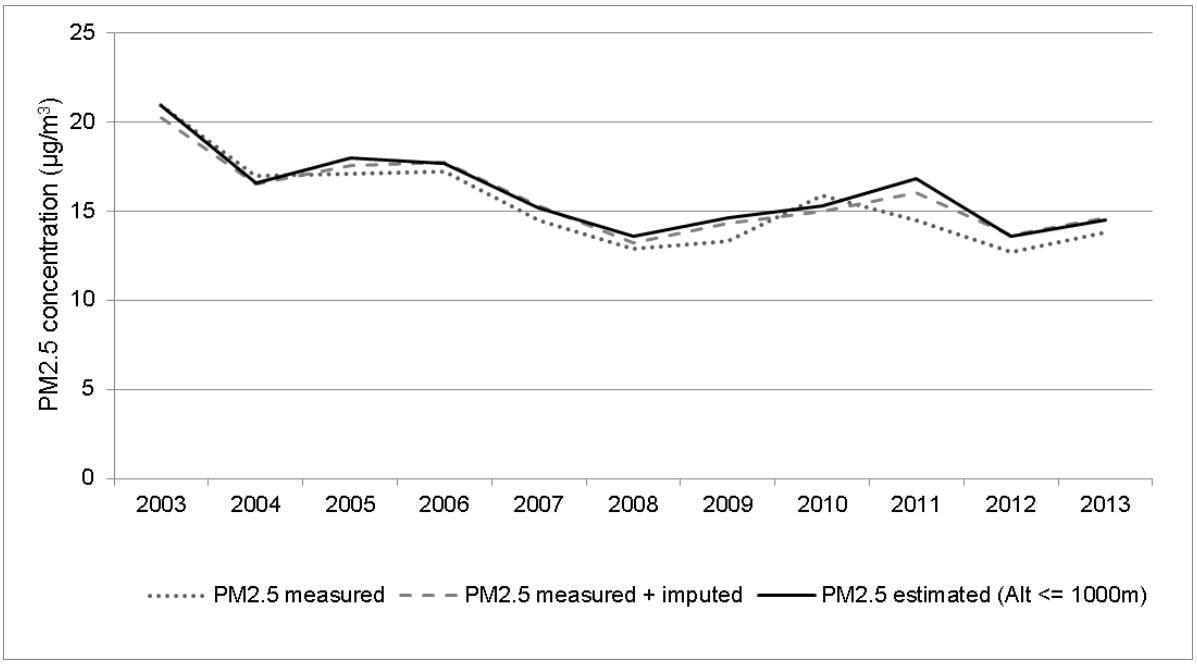

Figure A.4: Comparison by year of mean $PM_{2.5}$ concentrations for a) all $PM_{2.5}$ measurements; b) all $PM_{2.5}$ measurements + imputed $PM_{2.5}$ concentrations at the PM_{10} sites; and c) all $PM_{2.5}$ estimates across Switzerland for grid cells lying <= 1000m (1x1km, stage 3).

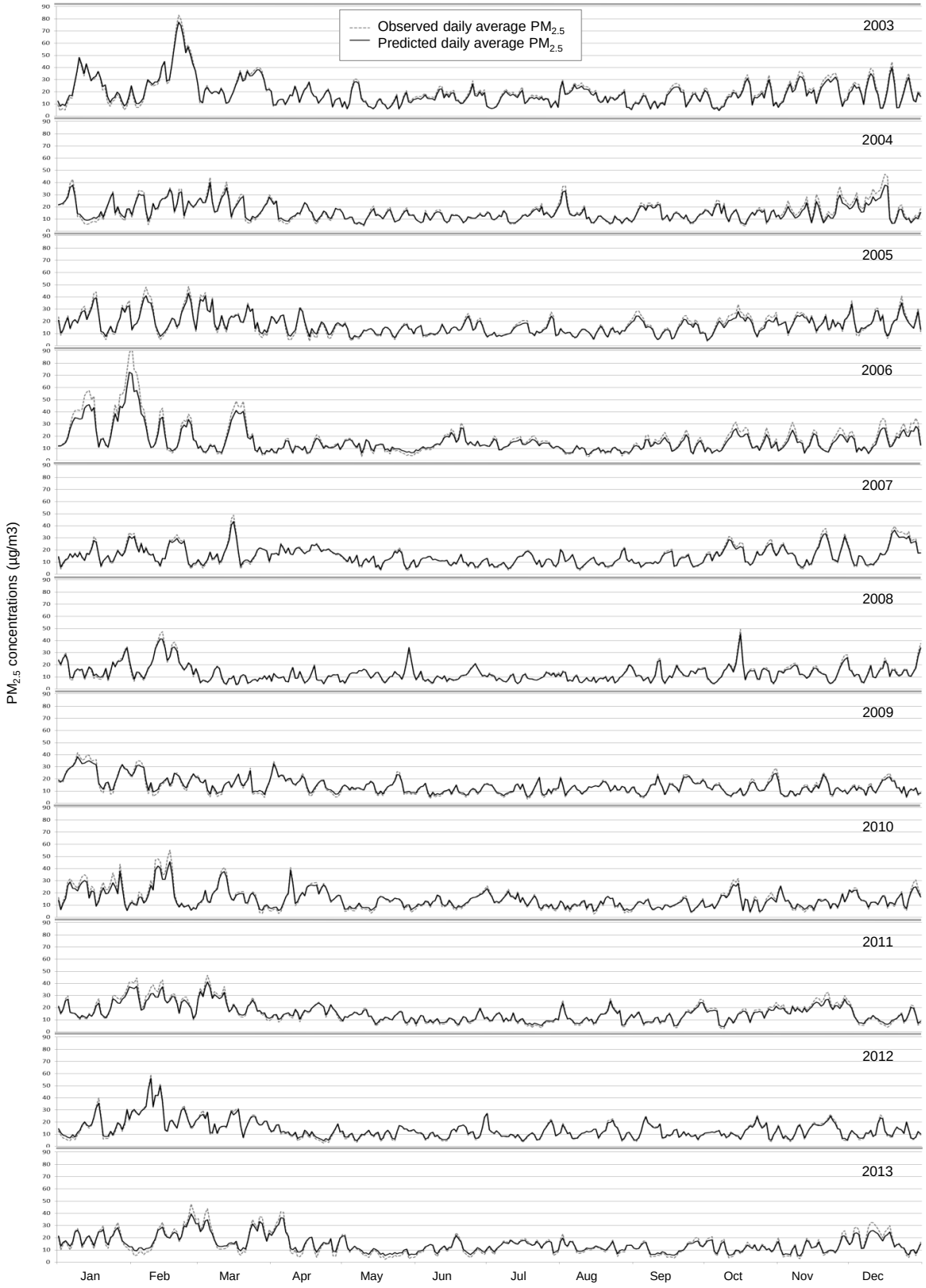

Figure A.5: Observed daily average versus predicted daily average $PM_{2.5}$ concentrations for 2003 to 2013; Observed daily $PM_{2.5}$ concentrations calculated as the average $PM_{2.5}$ across all 99 operating sites in Switzerland; Predicted daily $PM_{2.5}$ concentrations calculated as the average across all 1x1km cells covering Switzerland (stage 3)

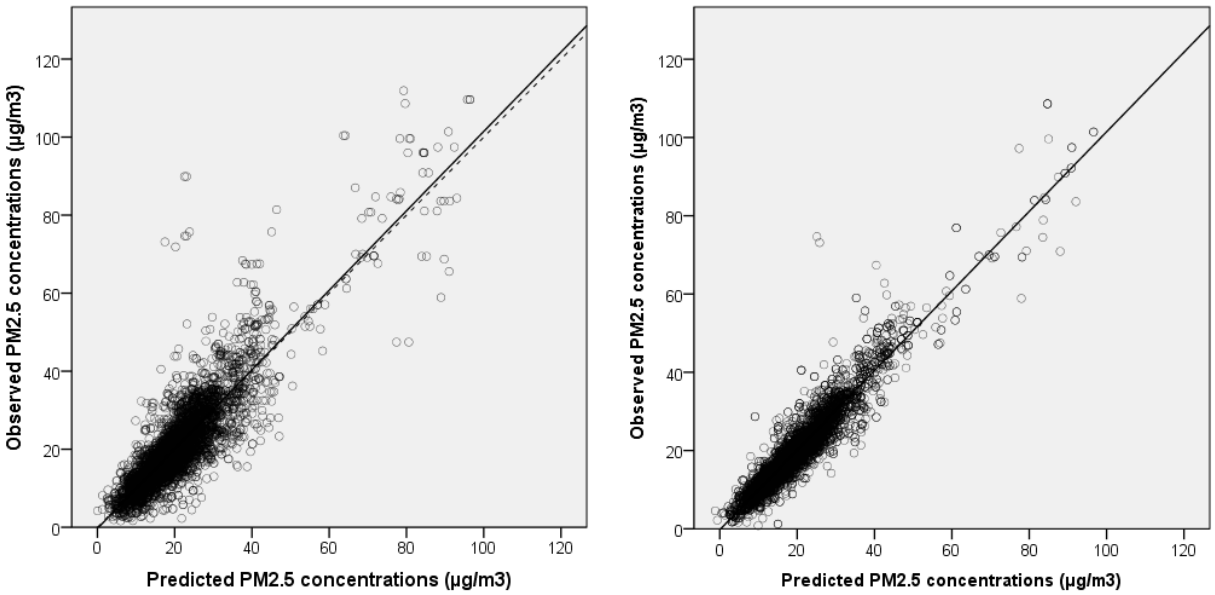

Figure A.6; Daily predicted versus observed $PM_{2.5}$ concentrations (µg/m3) at PM monitoring stations (n = 5210) for 2003 after stage 1 (CV R^2=.763; left) and stage 4 (R^2=.899; right).

8. Paper 5

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A systematic analysis of mutual effects of transportation noise and air pollution exposure on myocardial infarction mortality: a nationwide cohort study in Switzerland

Harris Héritier[1,2]; Danielle Vienneau[1,2]; Maria Foraster[1,2], Ikenna C. Eze[1,2], Emmanuel Schaffner[1,2] Laurie Thiesse[3], Franziska Rudzik[3], Manuel Habermacher[4], Micha Köpfli[4], Reto Pieren[5], Kees de Hoogh[1,2], Mark Brink[6], Christian Cajochen[3], Jean Marc Wunderli[5], Nicole Probst-Hensch[1,2], Martin Röösli [1,2] for the SNC study group

[1] Swiss Tropical and Public Health Institute, Basel, Switzerland

[2] University of Basel, Basel, Switzerland

[3] Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Basel, Switzerland

[4] N-sphere AG, Zürich, Switzerland

[5] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Dubendorf, Switzerland.

[6] Federal Office for the Environment, Bern, Switzerland

A systematic analysis of mutual effects of transportation noise and air pollution exposure on myocardial infarction mortality: a nationwide cohort study in Switzerland

Harris Héritier[1,2*], Danielle Vienneau[1,2*], Maria Foraster[1,2,3], Ikenna C. Eze[1,2], Emmanuel Schaffner[1,2] Kees de Hoogh[1,2], Laurie Thiesse[4,5], Franziska Rudzik[4,5], Manuel Habermacher[6], Micha Köpfli[6], Reto Pieren[7], Mark Brink[8], Christian Cajochen[4,5], Jean Marc Wunderli[7], Nicole Probst-Hensch[1,2], Martin Röösli [1,2] for the SNC study group

[1] Swiss Tropical and Public Health Institute, Basel, Switzerland

[2] University of Basel, Basel, Switzerland

[3] Barcelona Institute for Global Health (ISGlobal)

[4] Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Basel, Switzerland

[5] Transfaculty Research Platform Molecular and Cognitive Neurosciences, University of Basel, Basel, Switzerland

[6] N-sphere AG, Zürich, Switzerland

[7] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Dübendorf, Switzerland.

[8] Federal Office for the Environment, Bern, Switzerland

* both authors contributed equally

Correspondence:

Martin Röösli

Swiss Tropical and Public Health Institute

Socinstrasse 57

P.O. Box

CH-4002 Basel

E-Mail martin.roosli@unibas.ch

Tel. +41 (0)61 284 83 83

Fax +41 (0)61 284 85 01

Abstract

Aims: The present study aimed to disentangle the risk of the three major transportation noise sources – road, railway and aircraft traffic– and the air pollutants NO_2 and $PM_{2.5}$ on myocardial infarction (MI) mortality in Switzerland based on high quality/fine resolution exposure modelling.

Methods and Results: We modelled long term exposure to outdoor road traffic, railway and aircraft noise levels, as well as NO_2 and $PM_{2.5}$ concentration for each address of the 4.40 million adults (>30y) in the Swiss National Cohort (SNC). We investigated the association between transportation noise/air pollution exposure and death due to MI during the follow-up period 2000 to 2008, by adjusting noise (L_{den}(Road), L_{den}(Railway), L_{den}(Air)) estimates for NO_2 and/or $PM_{2.5}$ and vice versa by multipollutant Cox regression models considering potential confounders. Adjusting noise risk estimates of MI for NO_2 and/or $PM_{2.5}$ did not change the hazard ratios per 10 dB increase in road traffic (without air pollution: 1.032, 95% CI: 1.014-1.051, adjusted for NO_2 and $PM_{2.5}$: 1.034, 1.014-1.055), railway traffic (1.020, 1.007-1.033 vs. 1.020, 1.007-1.033) and aircraft traffic noise (1.025, 1.006-1.045 vs. 1.025, 1.005-1.046). Conversely, noise adjusted hazard ratios for air pollutants were lower than corresponding estimates without noise adjustment. HR per 10 µg/m³ increase with and without noise adjustment were 1.024 (1.005-1.043) vs. 0.990 (0.965-1.016) for NO_2 and 1.054 (1.013-1.093) vs. 1.019 (0.971-1.071) for PM2.5.

Conclusion: Our study suggests that transportation noise is associated with MI mortality, independent from air pollution. However, air pollution studies not adequately adjusting for transportation noise exposure may overestimate the cardiovascular disease burden of air pollution.

Keywords: Noise, Road traffic, Railway, Aircraft, Air pollution

Introduction

Several meta-analyses have highlighted the link between transportation noise and cardiovascular health. Babisch[1] reported a risk increase of 1.08 (95% CI; 1.04-1.13) for coronary heart diseases per 10 dB(A) increase in road traffic noise levels, and positive associations between myocardial infarction (MI) and exposure to road traffic and to aircraft noise have also been reported by Vienneau.[2] Air pollutants have also been shown to impact cardiovascular health. A recent meta-analysis focusing on MI reported a relative risk of 1.011 (95% CI; 1.006-1.016) and 1.025 (95% CI; 1.015-1.036) per 10 $\mu g/m^3$ increase in NO_2 and $PM_{2.5}$ concentrations, respectively.[3] Transportation noise and air pollution impact health through different pathways,[4] though they share many biologic pathways.

Mutual confounding is also of concern, since transportation noise and air pollution mainly originate from traffic. NO_2 and road traffic noise are often highly spatially correlated; aircraft and railway noise are less correlated with air pollution[5, 6] and thus offer the potential to elucidate their mutual independent impact on health. Correlations between long-term traffic noise and air pollution ranging from 0.16 to 0.72 were reported in a systematic review.[7] However, Fecht and colleagues[8] found that correlation depends on the spatial unit, with largest ranges seen when comparing across smaller vs. larger spatial units. Finally, correlation between transportation noise and air pollution has been shown to decrease with decreasing measurement error demonstrating the need of high quality exposure modeling.[9] Various studies have investigated the link between co-exposure to air pollution and noise, and CVD mortality. Some report independent noise effects[6, 10-14] while others suggest attenuation of the noise effect estimates after adjustment for air pollution thus indicating confounding or antagonistic interaction[15, 16] or did not adjust for exposure to air pollutants.[17, 18] A limitation in many of these studies, including our own previous analysis on this cohort,[19] is the fact that they do not model exposure at the same spatial scale. Depending on the main interest of a

study, the model used for adjustment may be less accurate than the main exposure model which would then yield partial confounding adjustment. A systematic review of nine studies comprising outcomes such as hospital discharge registers, self-reported medication intake, and mortality found that less than 10% of the effect estimate of noise was attenuated after adjustment for air pollution or vice versa[7] and thus concluded that confounding of cardiovascular effects by noise or air pollution is low. However, improvements in exposure assessment may change the situation.

As per our previous cohort study,[20, 21] we developed high-quality models to assess road, railway, and aircraft noise. In this study, we further included highly detailed NO_2 and $PM_{2.5}$ exposures to investigate the independent associations between transportation noise and air pollution at the participants' residence with MI mortality.

Methods

Study population

The SNC probabilistically links national census data with mortality and emigration records.[22] The data used in our study is based on the 4 December 2000 census and on mortality and emigration data for the period 5 December 2000 to 31 December 2008 and contains 7.28 million observations. We excluded subjects below 30 years of age (n=2.59 million) as myocardial mortality is very rare in this age group, observations for which residential coordinates were missing (n=0.19 million) or no buffers for the prediction of the air pollution levels could be calculated (n=0.01 million), subjects living in an institution such as special-care homes (n=0.25 million), and observations for which the cause of death was imputed (0.03 million) leaving 4.40 million observations for the analyses. The outcome under investigation was primary causes of death from MI (ICD-10: I21-I22). Immediately after death, primary and underlying causes of deaths are recorded by a physician, possibly verified

later by autopsy. Eventually, coding of causes is done centrally by the Federal Statistical Office using the German Modification of the ICD-10 system. The SNC was approved by the cantonal ethics boards of Bern and Zurich.

Noise exposure data

Within the framework of the SiRENE project (Short and Long Term Effects of Transportation Noise Exposure), we built a Swiss-wide noise exposure database for the year 2001 which included the three major transportation noise sources in Switzerland: road traffic, railway and aircraft noise.

The noise exposure database is described in detail elsewhere.[23] In brief, road traffic noise emissions were calculated using sonROAD [24] while propagation was computed via the propagation model of StL-86 [25]. For railway noise, the emissions were calculated using sonRAIL[26] and propagation was computed using the Swiss railway noise model SEMIBEL.[27] Aircraft noise exposure estimates were calculated via FLULA2.[28]

For each building in Switzerland, transportation noise exposure was estimated at pre-defined façade points with a maximum of 3 per facade.[23] For each façade point, we calculated the L_{den} for each noise source. Using the available geocodes and the information about floor of residence, we linked participants to their respective dwelling unit to assign noise exposure. Exposure was assigned on the basis of the façade point per dwelling unit with the highest L_{den} value. If information on the floor of residence was not available, we assigned the noise estimates corresponding to the middle floor of the building.

NO_2 exposure

The fine scale NO_2 model was based on data from cantonal air pollution monitoring authorities of Genève, Vaud, Neuchâtel, Jura, Fribourg, Berne, Basel-City, Basel-Country, Solothurn, Aargau, Ticino, the Inluft and Ostluft network, and the Sapaldia team.[29] The data

comprised 9,469 data points from 14-days passive measurements collected from 2000−2008 at a total of 1,834 locations. Missing data was imputed by considering available values from other monitoring sites within the same network based on inverse distance weighting. Subsequently, annual mean concentration for each year and site were calculated, and were regressed against various spatial predictors as outlined in the supplementary material including table S1.

The prediction for the SNC at residential address was performed for each year, from 2000−2008, and then averaged to obtain a long-term NO_2 exposure estimate for each participant.

$PM_{2.5}$ exposure

Daily $PM_{2.5}$ at 100 m grid cells across Switzerland was predicted for 2003−2008 from satellite, land use and meteorological data as described in detail in de Hoogh et al, 2017.[30] We used aerosol optical depth (AOD) data for the period of 2003−2008 at 1×1 km resolution and combined it in 4-staged modelling approach[31] with various predictors (see supplementary material) and with $PM_{2.5}$ ground measurements to refine model resolution to address level. The annual models for the period 2003−2008 were used to predict $PM_{2.5}$ exposure for each SNC study participant, and the average over the whole period was used as a long term $PM_{2.5}$ exposure measure in the epidemiological analysis.

Statistical Analysis

We analysed the data using the Cox proportional hazards model with age at date of entry into the cohort as the underlying time variable. Participants were followed until emigration, death or end of follow-up. In order to capture long term effects of noise and air pollution, exposure values representing the average during the follow-up period as described above were added as static covariate into the model. L_{den} variables were left censored at 35 dB (road traffic) or 30

dB (railway and aircraft noise). Linear hazard ratios (HR) were computed using multipollutant models adjusted for potential confounders excluding NO_2 and $PM_{2.5}$ in a first step, and stepwise including them in a second step. Additional potential confounders included in the model were sex, neighborhood index of socio-economic position,[32] civil status, educational level, nationality, and mother tongue using the categories depicted in Table 1. The latter was selected as it is expected to represent cultural variability in health behaviour in Switzerland.[33] To satisfy the Cox proportional hazard assumption, we stratified the baseline hazard function on the following variables; sex, neighbourhood index of socio-economic position, civil status, and education level. Potential multicollinearity between exposure variables in the Cox proportional hazards models was evaluated using the variance inflation factor (VIF).

In addition to linear HRs, we also conducted categorical noise analyses using L_{den}(Road) in 5 dB and L_{den}(Railway) and L_{den}(Air) in 10 dB categories to explore the effect of combined exposures of noise respectively with NO_2 and $PM_{2.5}$. All analyses were conducted with the statistical software R and the package "survival".[34, 35]

Results

The cohort contained 4.40 million observations amounting to 33.67 million person-years. There were 19,261 deaths from MI. The characteristics of the study population are displayed in table 1.

Table 1: Study population characteristics

Characteristics at baseline				
		Lden Road above median (54.09 dB)	NO_2 above median (26.95 µg/m^3)	$PM_{2.5}$ above median (18.74 µg/m^3)
Number of participants at baseline	4404046	2202848	2202744	2202025
Males (%)	48	48	47	48
Age: mean (SD)	52.4 (15.1)	52.47 (15.44)	52.57 (15.49)	52.51 (15.37)
Education level (%)				
Compulsory education or less	24	26	25	25
Upper secondary level education	52	50	50	50
Tertiary level education	22	21	23	22
Not known	2	3	3	3
Civil status (%)				
Single	14	15	17	16
Married	70	67	65	66
Widowed	8	8	8	8
Divorced	8	9	10	10
Socio-economic position (%) (38)				
low	33	37	32	33
medium	33	33	32	33
high	33	29	36	34
Mother tongue (%)				
German and Rhaeto-Romansch	65	59	60	60
French	19	22	19	19
Italian	7	9	10	10
Other	8	10	11	11
Nationality (%)				
Swiss	82	78	76	77
Rest of Europe (inclusive ex-USSR)	16	19	21	20
Other /unknown	12	2	3	3
NO_2 concentration in µg/m^3: mean (SD)	26.1 (7.3)	30.26 (7.78)	33.69 (5.6)	31.53 (7.17)
$PM_{2.5}$ concentration in µg/m^3: mean (SD)	20.2 (3.5)	20.16 (4.05)	21.12 (4.12)	22.11 (3.11)

For the NO_2 exposure model, R^2 values independently validated with NABEL data for each year, ranged from 0.70 to 0.82 with highest value for the year 2002 (table S2). R^2 values for 10 fold cross-validated elastic net models were ≥0.60 and the R^2 values for elastic net combined with kriged residuals were ≥0.84 (table S2). For the $PM_{2.5}$ model, cross validated temporal and spatial R^2 values ranged from 0.81 to 0.92 and 0.59 to 0.80.[30]

Mean NO_2 and $PM_{2.5}$ exposure concentrations were 26.1 and 20.2 µg/m^3 as depicted in figure S1 of supplemental material. The highest Spearman's correlation coefficient between the noise and the air pollution variables was 0.44, observed between L_{den}(Road) and NO_2 (table 2). Correlation between $PM_{2.5}$ and road (0.27), railway (0.20) and aircraft noise (0.24) was rather low. The correlation coefficient between NO_2 and $PM_{2.5}$ was 0.62.

Table 2: Spearman's rank correlation coefficients and variance inflation factor (VIF) for road traffic, railway, and aircraft noise as well as for $PM_{2.5}$ and NO_2

	Lden Road	**Lden Railway**	**Lden Air**	**$PM_{2.5}$**	**NO_2**
Lden Road	1				
Lden Railway	0.13	1			
Lden Air	0.09	-0.04	1		
$PM_{2.5}$	0.27	0.20	0.24	1	
NO_2	0.44	0.18	0.27	0.62	1

Without considering air pollution exposure, the hazard ratio for MI mortality per 10 dB noise increase was 1.032 (1.014-1.051) for road traffic, 1.020 (1.007-1.033) for railway traffic and 1.025 (1.006-1.045) for aircraft traffic (Table S3). Adjustment for air pollution exposure had little impact on the HR of the linear exposure-response relationship for all three transportation noise sources (Figure 1).

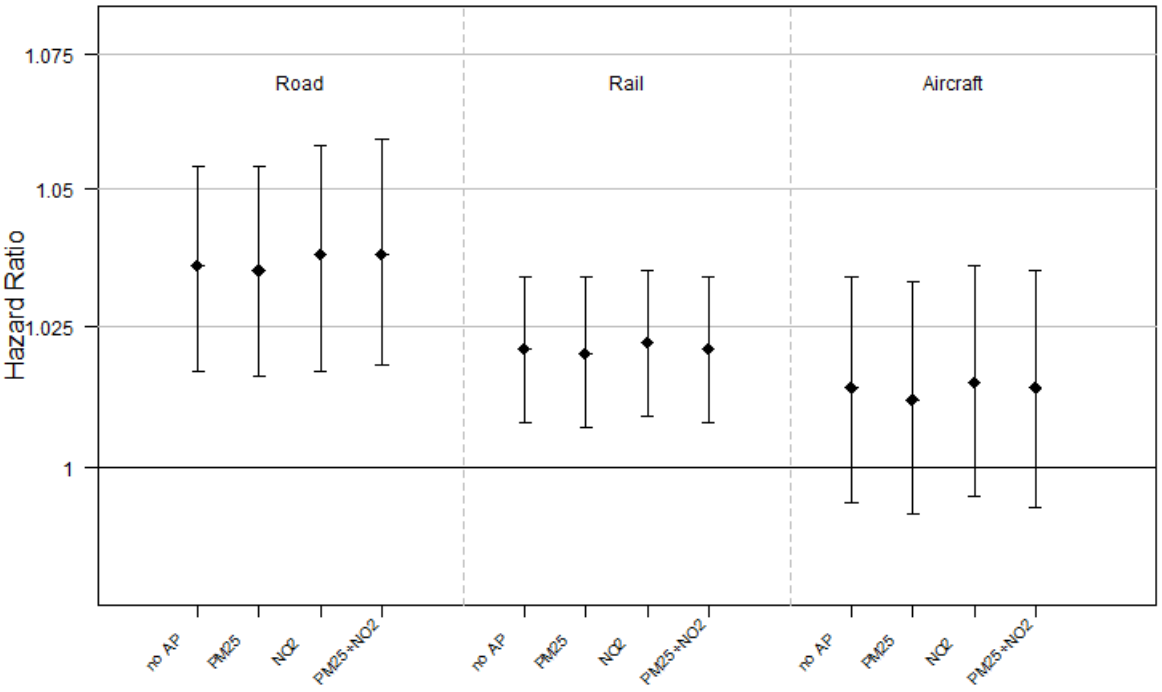

Figure 1: linear HRs for associations between road, railway, and aircraft noise exposure and myocardial infarction per 10 dB increase in L_{den}, not adjusted for air pollution (no AP), adjusted for $PM_{2.5}$ only ($PM_{2.5}$), adjusted for NO_2 only (NO_2), and adjusted for $PM_{2.5}$ and NO_2 ($PM_{2.5}$+NO_2). All models were adjusted for age, sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and the other noise sources.

Categorical models showed a slight attenuation of the noise estimates for MI in all road traffic noise exposure categories after adjustment for both air pollutants (table S4). A similar pattern was seen for the association between MI mortality and railway noise (table S5) but not aircraft noise (table S6).

Without considering noise exposure, the hazard ratio for MI mortality per 10 µg/m³ increase in NO_2 was 1.024 (1.005-1.043) and per 10 µg/m³ increase in $PM_{2.5}$ 1.052 (1.013-1.093) (table S7). In this case risk estimates for air pollution exposure unadjusted for noise tended to decrease upon adjustment for all noise sources (Figure 2).

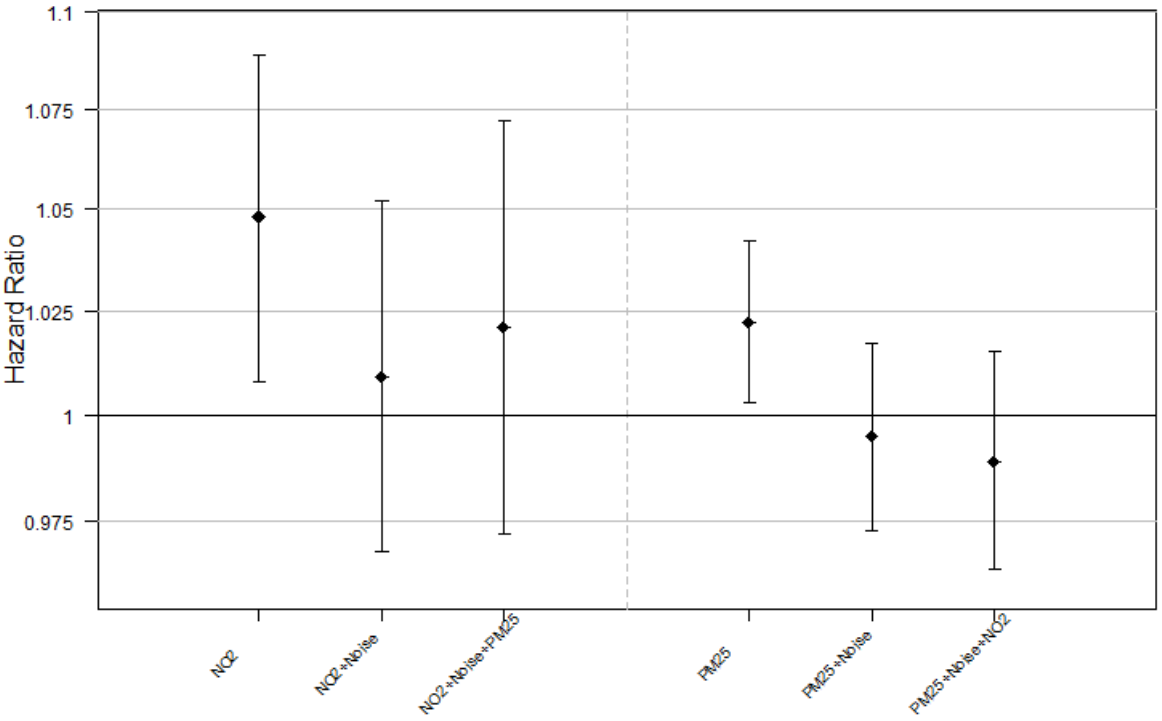

Figure 2: linear HRs for associations between $PM_{2.5}$ (left side) and NO_2 (right side) per 10 µg/m^3 and myocardial infarction in single exposure models, adjusted additionally for all noise sources, and adjusted for all noise sources and the complimentary air pollutant. All models were adjusted for age, sex, neighbourhood index of socio-economic position, civil status, education level, mother tongue, and nationality.

Multicollinearity between noise and air pollution exposure measures was not critical. In fully adjusted multipollutant models VIF was 1.25 for road traffic noise, 1.06 for railway traffic noise, 1.10 for aircraft noise, 1.65 for $PM_{2.5}$ and 1.92 for NO_2.

Synergistic or antagonistic effects between road traffic noise and $PM_{2.5}$ or NO_2 could not be seen in linear-exposure response models including interaction terms. Similarly, testing interactions in categorical models to evaluate potential thresholds for interaction did not reveal any relevant interactions for PM2.5 (table 3) or NO2 (table S8) with road traffic noise exposure. For instance, interaction terms were close to unity for people exposed to high levels of noise and air pollutants.

Table 3: Adjusted* HR for NO_2 and road traffic noise for MI in categorical interaction exposure models. Number of cases in each cell is shown in italics.

		Road traffic noise [dB]			
		<40 (reference) ***N=699***	**41-50** ***N=4393***	**51-60** ***N=8747***	**>60** ***N=5422***
NO2 [ug/m3]	**<20 (reference)** ***N=2824***	1.00 *N=325*	1.04 (0.93-1.18) *N=998*	1.12 (1.00-1.26) N=1172	1.09 (0.94-1.27) *N=329*
	21-30 ***N=9560***	1.11 (0.95-1.29) *N=311*	0.83 (0.70-0.98) *N=2583*	0.82 (0.70-0.97) *N=4566*	0.94 (0.77-1.13) *N=2100*
	31-40 ***N=5500***	0.74 (0.47-1.18) *N=19*	1.19 (0.74-1.91) *N=738*	1.19 (0.75-1.89) *N=2627*	1.21 (0.75-1.95) *N=2116*
	>40 ***N=1377***	2.24 (0.84-6.00) *N=44*	0.32 (0.12-0.90) *N=74*	0.37 (0.14-0.99) *N=382*	0.39 (0.15-1.06) *N=877*

* Age as the underlying time scale and additionally adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality, railway and aircraft noise, $PM_{2.5}$.

Discussion

Using fine scale exposure modelling, this study demonstrates that the association between transportation noise and death from MI is stable to air pollution adjustment but not vice versa.

NO_2 originates mainly from road traffic while $PM_{2.5}$ arises from multiple sources including those not related to traffic such as industrial areas, power plants and wood burning and is thus less correlated to road traffic noise than NO_2. In multipollutant air pollution studies, the impact of NO_2 on mortality was independent from $PM_{2.5}$.[36] Strikingly many of the air pollution studies have not adjusted for transportation noise and thus noise as confounder in these studies cannot be ruled out, in particular for associations with NO_2. In the present study we found that the effect estimates of both air pollutants were attenuated upon inclusion of the noise variables in the models, which indicates a confounding effect of transportation noise on air pollution. This finding would imply that many air pollution studies, which have not

adjusted for transportation noise, may have overestimated the effects of air pollution on MI mortality.

On the other hand, our associations between noise and MI mortality were robust to $PM_{2.5}$ adjustment and only slightly attenuated if NO_2 was considered in the analyses. However, we were not able to evaluate the effects of ultrafine particles (UFP) due to lack of a national model. Of all air pollutants, UFP may have the most similar propagation behaviour to noise. Ultrafine particles are moderately correlated to road traffic noise[37-41] and are a known risk factor for cardiovascular morbidity.[42, 43] Ultrafine particles, however, are poorly correlated with $PM_{2.5}$ and we cannot fully rule out that road traffic noise exposure is confounded by UFP. However, UFP are expected to be poorly correlated with railway noise in Switzerland, since railways are electrified, and thus the corresponding risk estimates are unlikely to suffer from such a bias.

Noise and air pollution exposure were both estimated at the residential address and thus bias due to different spatial resolution is expected to be minimized. Such bias may have occurred in our previous analysis[20, 21] and other studies.[7, 44-46] The models are further comparable in that both the noise and air pollution estimates reflect the ambient exposure, rather than indoor exposure, and therefore suffer similar bias. Nevertheless, bias cannot be completely excluded if accuracy of the models would differ; although R^2 in external validations available for NO_2 and road traffic noise were found to be similar (0.70-0.82 for all models).

In previous analyses conducted with the same cohort and noise data,[19] the noise effect models were adjusted for NO_2 derived from PolluMap, a 200×200m dispersion model for the year 2010. Here we improved the spatial resolution of the NO_2 exposure estimate, from the grid level to the address level, by further modelling with an extensive passive sampling network distributed across the country. PolluMap was included in this new NO_2 model as a predictor variable, and was found to be the most relevant predictor for NO_2. However including

additional factors like road and population density around the place of residence produced in improved NO_2 exposure estimates (Table S1). The correlation between NO_2 estimates from PolluMap used in our previous study[19] and from the NO_2 estimates used here is 0.83. In contrast to our previous analyses we did not consider the intermittency ratio - a measure of noise eventfulness - as an additional noise metric to the Leq.

Our results for noise are in line with most of the criteria for evaluating causality proposed by Sir Arthur Hill [47] such as consistency,[48, 49] temporality, biological gradient, plausibility[50] and coherence[51], although specificity cannot be expected for the multifactorial disease MI. The effect sizes are small and not of clinical relevance for an individual. Overall, however, public health burden is relevant as many people are exposed to transportation noise and air pollution. A previous health impact assessment for Switzerland concluded that transportation noise and air pollution caused 6,000 and 14,000 years of life lost in 2010.[52]

Strength and limitations

The strengths of this study include the large study population and the long follow-up time. We developed a detailed noise exposure model, which allowed for an individual exposure assessment at the address and floor level. Our air pollution models for NO_2 and $PM_{2.5}$ were both based on novel approaches using high resolution input data. Potential selection bias is minimal in this nationwide study based on census data. Finally, both the noise and air pollution models have been validated with independent data. The road traffic noise model for 2011 has been validated using 99 weekly measurements conducted in 2016 yielding an average difference between modelling and measurements of +0.5 dB(A) with a standard deviation of 4.0 dB(A).[53] This good agreement was obtained despite a time lag of five years between modelling and measurements demonstrating stable noise exposure in our study area. Similar stability is also expected for the time between baseline, time at which the noise exposure was assigned, and follow-up of this cohort study.

Though our models are adjusted for socioeconomic status and other demographic variables, we could not adjust for lifestyle and smoking as this information is not available in the SNC. We therefore cannot rule out that residual confounding of lifestyle may play a role for our analyses, although no indications for this were seen in a previous SNC noise study.[6] However, the non-significant effect estimates observed for NO_2 and $PM_{2.5}$ after noise adjustment may suffer from residual confounding. In Switzerland, mortality from cardiovascular diseases has been shown to be higher in rural areas,[33] where air pollution is lower on average. This mortality pattern is likely due to individual risk factors and the number of health facilities associated with urban areas, and thus correlated with NO_2 and $PM_{2.5}$, which may not be fully considered in our adjustment set. Since noise exposure varies on a very small scale, this type of bias is likely less relevant for these estimates.

Further, despite high quality exposure modelling, exposure misclassification is unavoidable due to uncertainty in the input data. For noise, uncertainty may arise from exposure assignment based on estimates for the loudest outdoor facade point while no information was available regarding indoor noise levels and noise attenuation factors. However, the impact of such misclassification on the study results are similar for noise and air pollution, and are more likely to dilute the association than introduce a spurious effect.

Conclusion

In this analysis on MI, mutually adjusted with fine-scale noise and air pollution modelling at address level, a consistent exposure-response association between long term transportation noise exposure and MI mortality was observed. This association was independent from the effects observed for air pollution. Conversely, air pollution effects decreased upon adjustment for transportation noise exposure. Future studies need high quality exposure models for both air pollution and transportation noise to better understand their clinical and public health relevance for cardiovascular disease in various settings.

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Conflict of interests

The authors declare no conflict of interests.

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References

1. Babisch W. Updated exposure-response relationship between road traffic noise and coronary heart diseases: A meta-analysis. *Noise and Health* 2014; **16**:1-9.
2. Vienneau D, Schindler C, Perez L, Probst-Hensch N, Röösli M. The relationship between transportation noise exposure and ischemic heart disease: a meta-analysis. *Environmental research* 2015; **138**:372-380.
3. Mustafić H, Jabre P, Caussin C, Murad MH, Escolano S, Tafflet M, Périer MC, Marijon E, Vernerey D, Empana JP, Jouven X. Main air pollutants and myocardial infarction: A systematic review and meta-analysis. *JAMA* 2012; **307**:713-721.
4. Münzel T, Sørensen M, Gori T, Schmidt FP, Rao X, Brook J, Chen LC, Brook RD, Rajagopalan S. Environmental stressors and cardio-metabolic disease: part I–epidemiologic evidence supporting a role for noise and air pollution and effects of mitigation strategies. *European Heart Journal* 2016:ehw269.
5. Dratva J, Phuleria HC, Foraster M, Gaspoz J-M, Keidel D, Künzli N, Liu LS, Pons M, Zemp E, Gerbase MW. Transportation noise and blood pressure in a population-based sample of adults. *Environmental health perspectives* 2012; **120**:50.

6. Huss A, Spoerri A, Egger M, Roosli M. Aircraft noise, air pollution, and mortality from myocardial infarction. *Epidemiology* 2010; **21**:829-36.
7. Tetreault LF, Perron S, Smargiassi A. Cardiovascular health, traffic-related air pollution and noise: are associations mutually confounded? A systematic review. *International journal of public health* 2013; **58**:649-66.
8. Fecht D, Hansell AL, Morley D, Dajnak D, Vienneau D, Beevers S, Toledano MB, Kelly FJ, Anderson HR, Gulliver J. Spatial and temporal associations of road traffic noise and air pollution in London: Implications for epidemiological studies. *Environment international* 2016; **88**:235-242.
9. Foraster M, Künzli N, Aguilera I, Rivera M, Agis D, Vila J, Bouso L, Deltell A, Marrugat J, Ramos R. High blood pressure and long-term exposure to indoor noise and air pollution from road traffic. *Environmental health perspectives* 2014; **122**:1193.
10. Correia AW, Peters JL, Levy JI, Melly S, Dominici F. Residential exposure to aircraft noise and hospital admissions for cardiovascular diseases: multi-airport retrospective study. 2013.
11. Floud S, Blangiardo M, Clark C, de Hoogh K, Babisch W, Houthuijs D, Swart W, Pershagen G, Katsouyanni K, Velonakis M, Vigna-Taglianti F, Cadum E, Hansell AL. Exposure to aircraft and road traffic noise and associations with heart disease and stroke in six European countries: a cross-sectional study. *Environmental Health* 2013; **12**:89.
12. Gan WQ, Davies HW, Koehoorn M, Brauer M. Association of long-term exposure to community noise and traffic-related air pollution with coronary heart disease mortality. *Am J Epidemiol* 2012; **175**:898-906.
13. Hansell AL, Blangiardo M, Fortunato L, Floud S, de Hoogh K, Fecht D, Ghosh RE, Laszlo HE, Pearson C, Beale L, Beevers S, Gulliver J, Best N, Richardson S, Elliott P. Aircraft noise and cardiovascular disease near Heathrow airport in London: small area study. *BMJ : British Medical Journal* 2013; **347**.
14. Sorensen M, Andersen ZJ, Nordsborg RB, Jensen SS, Lillelund KG, Beelen R, Schmidt EB, Tjonneland A, Overvad K, Raaschou-Nielsen O. Road traffic noise and incident myocardial infarction: a prospective cohort study. *PLoS One* 2012; **7**:e39283.
15. Beelen R, Hoek G, Houthuijs D, van den Brandt PA, Goldbohm RA, Fischer P, Schouten LJ, Armstrong B, Brunekreef B. The joint association of air pollution and noise from road traffic with cardiovascular mortality in a cohort study. *Occupational and environmental medicine* 2009; **66**:243-50.
16. Tonne C, Halonen JI, Beevers SD, Dajnak D, Gulliver J, Kelly FJ, Wilkinson P, Anderson HR. Long-term traffic air and noise pollution in relation to mortality and hospital readmission among myocardial infarction survivors. *Int J Hyg Environ Health* 2016; **219**:72-8.
17. Seidler A, Wagner M, Schubert M, Dröge P, Pons-Kühnemann J, Swart E, Zeeb H, Hegewald J. Myocardial Infarction Risk Due to Aircraft, Road, and Rail Traffic Noise. *Deutsches Ärzteblatt international* 2016; **113**:407.
18. Seidler A, Wagner M, Schubert M, Dröge P, Römer K, Pons-Kühnemann J, Swart E, Zeeb H, Hegewald J. Aircraft, road and railway traffic noise as risk factors for heart failure and hypertensive heart disease—A case-control study based on secondary data. *International Journal of Hygiene and Environmental Health* 2016; **219**:749-758.
19. Héritier H, Vienneau D, Foraster M, Eze IC, Schaffner E, Thiesse L, Rudzik F, Habermacher M, Köpfli M, Pieren R, Brink M, Cajochen C, Wunderli JM, Probst-Hensch N, Röösli M. Transportation noise exposure and cardiovascular mortality: a nationwide cohort study from Switzerland. *European Journal of Epidemiology* 2017:1-9.
20. Heritier H, Vienneau D, Foraster M, Eze IC, Schaffner E, Thiesse L, Rudzik F, Habermacher M, Kopfli M, Pieren R, Brink M, Cajochen C, Wunderli JM, Probst-Hensch N, Röösli M, group SNCs. Transportation noise exposure and cardiovascular mortality: a nationwide cohort study from Switzerland. *Eur J Epidemiol* 2017; **32**:307–315.
21. Heritier H, Vienneau D, Foraster M, Eze IC, Schaffner E, Thiesse L, Rudzik F, Habermacher M, Kopfli M, Pieren R, Schmidt-Trucksäss A, Brink M, Cajochen C, Wunderli JM, Probst-Hensch N, Röösli M, group SNCs. Diurnal variability of transportation noise exposure and cardiovascular mortality: a nationwide cohort study from Switzerland. *International journal of hygiene and environmental health* 2018:epub ahead of print.

22. Spoerri A, Zwahlen M, Egger M, Bopp M. The Swiss National Cohort: a unique database for national and international researchers. *International journal of public health* 2010; **55**:239-242.
23. Karipidis I, Vienneau D, Habermacher M, Köpflii M, Brink M, Probst-Hensch N, Röösli M, Wunderli J-M. Reconstruction of historical noise exposure data for environmental epidemiology in Switzerland within the SiRENE project. *Noise mapping* 2014; **1**.
24. Heutschi K. SonRoad: New Swiss road traffic noise model. *Acta Acustica united with Acustica* 2004; **90**:548-554.
25. BAFU *Computermodell zur Berechnung von Strassenlärm, Teil 1, Bedienungsanleitung zum Computerprogramm StL-86*; Bundesamt für Umweltschutz: Bern, 1987.
26. Thron T, Hecht M. The sonRAIL emission model for railway noise in Switzerland. *Acta Acustica United with Acustica* 2010; **96**:873-883.
27. FOEN *SEMIBEL: Schweizerisches Emissions- und Immissionsmodell für die Berechnung von Eisenbahnlaerm*; Swiss Federal Office for the Environment: Berne, 1990.
28. Thomann G, Bütikofer R, Krebs W. FLULA2 Ein Verfahren zur Berechnung und Darstellung der Fluglärmbelastung, Technische Programm-Dokumentation Version 2.1. In EMPA report, April: 2005.
29. Wüthrich B, Schmid-Grendelmeier P, Schindler C, Imboden M, Bircher A, Zemp E, Probst-Hensch N, Team S. Prevalence of atopy and respiratory allergic diseases in the elderly SAPALDIA population. *International archives of allergy and immunology* 2013; **162**:143-148.
30. de Hoogh K, Héritier H, Stafoggia M, Künzli N, Kloog I. Modelling daily PM2.5 concentrations at high spatio-temporal resolution across Switzerland. *Environmental Pollution* 2017.
31. Kloog I, Chudnovsky AA, Just AC, Nordio F, Koutrakis P, Coull BA, Lyapustin A, Wang Y, Schwartz J. A new hybrid spatio-temporal model for estimating daily multi-year PM2.5 concentrations across northeastern USA using high resolution aerosol optical depth data. *Atmospheric Environment* 2014; **95**:581-590.
32. Panczak R, Galobardes B, Voorpostel M, Spoerri A, Zwahlen M, Egger M. A Swiss neighbourhood index of socioeconomic position: development and association with mortality. *Journal of epidemiology and community health* 2012:jech-2011-200699.
33. Chammartin F, Probst-Hensch N, Utzinger J, Vounatsou P. Mortality atlas of the main causes of death in Switzerland, 2008-2012. *Swiss Med Wkly* 2016; **146**:w14280.
34. Therneau Terry M. GPM, *Modeling Survival Data: Extending the Cox Model*. Springer: New York, 2000.
35. TM T *A package for Survival Analysis in S*, 2015.
36. Faustini A, Rapp R, Forastiere F. Nitrogen dioxide and mortality: review and meta-analysis of long-term studies. *European Respiratory Journal* 2014; **44**:744-753.
37. Allen RW, Davies H, Cohen MA, Mallach G, Kaufman JD, Adar SD. The spatial relationship between traffic-generated air pollution and noise in 2 US cities. *Environmental research* 2009; **109**:334-342.
38. Can A, Rademaker M, Van Renterghem T, Mishra V, Van Poppel M, Touhafi A, Theunis J, De Baets B, Botteldooren D. Correlation analysis of noise and ultrafine particle counts in a street canyon. *The Science of the total environment* 2011; **409**:564-72.
39. Kheirbek I, Ito K, Neitzel R, Kim J, Johnson S, Ross Z, Eisl H, Matte T. Spatial Variation in Environmental Noise and Air Pollution in New York City. *Journal of Urban Health : Bulletin of the New York Academy of Medicine* 2014; **91**:415-431.
40. Meier R, Cascio WE, Ghio AJ, Wild P, Danuser B, Riediker M. Associations of Short-Term Particle and Noise Exposures with Markers of Cardiovascular and Respiratory Health among Highway Maintenance Workers. *Environmental Health Perspectives* 2014; **122**:726-732.
41. Morelli X, Foraster M, Aguilera I, Basagana X, Corradi E, Deltell A, Ducret-Stich R, Phuleria H, Ragettli MS, Rivera M, Thomasson A, Künzli N, Slama R. Short-term associations between traffic-related noise, particle number and traffic flow in three European cities. *Atmospheric Environment* 2015; **103**:25-33.
42. Du Y, Xu X, Chu M, Guo Y, Wang J. Air particulate matter and cardiovascular disease: the epidemiological, biomedical and clinical evidence. *Journal of Thoracic Disease* 2016; **8**:E8-E19.
43. Lanki T, Pekkanen J, Aalto P, Elosua R, Berglind N, D'Ippoliti D, Kulmala M, Nyberg F, Peters A, Picciotto S, Salomaa V, Sunyer J, Tiittanen P, von Klot S, Forastiere F. Associations of traffic

related air pollutants with hospitalisation for first acute myocardial infarction: the HEAPSS study. *Occupational and Environmental Medicine* 2006; **63**:844.
44. Cai Y, Hansell AL, Blangiardo M, Burton PR, BioShaRe, de Hoogh K, Doiron D, Fortier I, Gulliver J, Hveem K, Mbatchou S, Morley DW, Stolk RP, Zijlema WL, Elliott P, Hodgson S. Long-term exposure to road traffic noise, ambient air pollution, and cardiovascular risk factors in the HUNT and lifelines cohorts. *European heart journal* 2017; **38**:2290-2296.
45. Cai Y, Hodgson S, Blangiardo M, Gulliver J, Morley D, Fecht D, Vienneau D, de Hoogh K, Key T, Hveem K, Elliott P, Hansell AL. Road traffic noise, air pollution and incident cardiovascular disease: A joint analysis of the HUNT, EPIC-Oxford and UK Biobank cohorts. *Environ Int* 2018; **114**:191-201.
46. Fuks KB, Weinmayr G, Basagana X, Gruzieva O, Hampel R, Oftedal B, Sorensen M, Wolf K, Aamodt G, Aasvang GM, Aguilera I, Becker T, Beelen R, Brunekreef B, Caracciolo B, Cyrys J, Elosua R, Eriksen KT, Foraster M, Fratiglioni L, Hilding A, Houthuijs D, Korek M, Kunzli N, Marrugat J, Nieuwenhuijsen M, Ostenson CG, Penell J, Pershagen G, Raaschou-Nielsen O, Swart WJR, Peters A, Hoffmann B. Long-term exposure to ambient air pollution and traffic noise and incident hypertension in seven cohorts of the European study of cohorts for air pollution effects (ESCAPE). *European heart journal* 2017; **38**:983-990.
47. Hill AB. The Environment and Disease: Association or Causation? *Proc R Soc Med* 1965; **58**:295-300.
48. Babisch W. Updated exposure-response relationship between road traffic noise and coronary heart diseases: a meta-analysis. *Noise & health* 2014; **16**:1-9.
49. Vienneau D, Schindler C, Perez L, Probst-Hensch N, Röösli M. The relationship between transportation noise exposure and ischemic heart disease: a meta-analysis. *Environmental research* 2015; **138**:372-80.
50. Münzel T, Sörensen M, Gori T, Schmidt FP, Rao X, Brook J, Chen LC, Brook RD, Rajagopalan S. Environmental stressors and cardio-metabolic disease: part I-epidemiologic evidence supporting a role for noise and air pollution and effects of mitigation strategies. *European heart journal* 2016.
51. Münzel T, Daiber A, Steven S, Tran LP, Ullmann E, Kossmann S, Schmidt FP, Oelze M, Xia N, Li H, Pinto A, Wild P, Pies K, Schmidt ER, Rapp S, Kroller-Schon S. Effects of noise on vascular function, oxidative stress, and inflammation: mechanistic insight from studies in mice. *European heart journal* 2017; **38**:2838-2849.
52. Vienneau D, Perez L, Schindler C, Lieb C, Sommer H, Probst-Hensch N, Kunzli N, Röösli M. Years of life lost and morbidity cases attributable to transportation noise and air pollution: A comparative health risk assessment for Switzerland in 2010. *International journal of hygiene and environmental health* 2015; **218**:514-21.
53. Schlatter F, Piquerez A, Habermacher M, Ragettli M, Röösli M, Brink M, Cajochen C, Probst-Hensch N, Foraster M, Wunderli J-M. Validation of large scale noise exposure modelling by long-term measurements. *Noise Mapping* 2017; **4**:75-86.

Supplementary material

A systematic analysis of mutual effects of transportation noise and air pollution exposure on myocardial infarction mortality: a nationwide cohort study in Switzerland

Harris Héritier[1,2*], Danielle Vienneau[1,2*], Maria Foraster[1,2], Ikenna C. Eze[1,2], Emmanuel Schaffner[1,2] Laurie Thiesse[3], Franziska Rudzik[3], Kees de Hoogh[1,2], Manuel Habermacher[4], Micha Köpfli[4], Reto Pieren[5], Mark Brink[6], Christian Cajochen[3], Jean Marc Wunderli[5], Nicole Probst-Hensch[1,2], Martin Röösli [1,2] for the SNC study group

[1] Swiss Tropical and Public Health Institute, Basel, Switzerland

[2] University of Basel, Basel, Switzerland

[3] Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Basel, Switzerland

[4] N-sphere AG, Zürich, Switzerland

[5] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Dubendorf, Switzerland.

[6] Federal Office for the Environment, Bern, Switzerland

* both authors contributed equally.

Correspondence:

Martin Röösli

Swiss Tropical and Public Health Institute

Socinstrasse 57

P.O. Box

CH-4002 Basel

E-Mail martin.roosli@unibas.ch

Tel. +41 (0)61 284 83 83

Fax +41 (0)61 284 85 01

Predictors and modeling strategy of the NO2 model

The following spatial predictors to were used to construct a NO2 model for the whole country:

- Building footprints of the year 2008 derived from Vector25, the digital landscape model of Switzerland.[1]
- Population density data at a 100×100 m grid were available for 2011 from the Bundesamt für Statistik (BfS) for Switzerland,[2] supplemented with a 100×100 m resolution European population grid for 2000 for the border areas.[3]
- CORINE Land Cover from the European Environment Agency for the year 2006. This was reclassified into 6 classes (Industry, water bodies, urban, farming, natural, and rocky natural).
- Road network and modelled traffic intensity data from 2008 from Vector 25[1] with resolution 1:25,000 were available from Swisstopo.

The above predictors were calculated for various buffer sizes (i.e. 25, 50, 75, 100, 150, 250, 500, 750, 1000m) around the place of residence. The following predictors refer to the coordinate of the home address:

- Elevation at residential was extracted from a 25m digital height model of Switzerland (DHM25) from the Federal Office for Topography.[4]
- The Topex indicator,[5] which reflects 'topographic exposure,' was computed by subtracting the altitude averaged over the 1000m buffer by the altitude at the point coordinate.
- NO_2 concentration at the residential coordinates from the PolluMap dispersion model for Switzerland at a 200×200 m resolution, for the individual years 2000-2005.[6]

Predictor selection for the models was conducted by elastic net regularization using the package glmnet in R,[7] by selecting the minimum lambda value following a 10-fold cross-validation. The NO_2 concentrations for each year from 2000 to 2008 were predicted and we extracted the residuals which where fitted based on an exponential variogram. The resulting fitted variogram was subsequently kriged yielding a smoothed layer that corrected the over- and under-predicted values. These predictions were independently validated using data from the NABEL network (652 data points collected from 2000 to 2008 at 137 locations). The prediction for the SNC at residential address was performed for each year, from 2000 to 2008, in two stages. We first predicted the annual NO_2 concentrations based on the elastic net models and in the second stage, we extracted the residuals from the annual kriged surface. First stage prediction and second stage residuals were then added to yield the final prediction.

References

1. *VECTOR25–Das digitale Landschaftsmodell der Schweiz*. Informationsbroschüre des Bundesamts FÜR Landestopographie der Schweiz: Wabern, 2000.
2. Basner M, Muller U, Elmenhorst EM. Single and combined effects of air, road, and rail traffic noise on sleep and recuperation. *Sleep* 2011; **34**:11-23.
3. Gallego FJ. A population density grid of the European Union. *Popul Environ* 2010; **31**:460-473.
4. Digital height model of Switzerland. In Swisstopo, Ed. Swisstopo: Wabern, 2005.
5. Beelen R, Hoek G, Pebesma E, Vienneau D, de Hoogh K, Briggs DJ. Mapping of background air pollution at a fine spatial scale across the European Union. *Science of the Total Environment* 2009; **407**:1852-1867.
6. FOEN. NO2 ambient concentrations in Switzerland, Modelling results for 2005, 2010, 2015. In Federal Office for the Environment: Berne, 2011.
7. Friedman J, Hastie T, Tibshirani R. Regularization Paths for Generalized Linear Models via Coordinate Descent. *Journal of Statistical Software* 2010; **33**:1-22.

Table S1: Predictors and coefficients of the NO_2 model for years 2000 to 2008

Predictors	2000	2001	2002	2003	2004	2005	2006	2007	2008
(Intercept)	28.70	26.79	26.48	26.17	24.71	25.58	26.62	24.88	25.18
Pollumap 2000	3.20	-	-	-	-	-	-	-	-
Pollumap 2001	-	2.82	-	-	-	-	-	-	-
Pollumap 2002	-	-	4.70	-	-	-	-	-	-
Pollumap 2003	-	-	-	4.24	-	-	-	-	-
Pollumap 2004	-	-	-	-	2.66	-	-	-	-
Pollumap 2005	-	-	-	-	-	3.45	3.12	3.02	2.67
Elevation	-1.68	-1.07	-0.84	-0.94	-0.16	-0.72	-0.13	-0.01	
Topex	-	-	0.43	0.54	-	-	-	0.17	0.13
Industry 25m	-	-	-	-	0.62	-	-	-	0.06
Industry 50m	0.44	0.36	0.34	0.37	0.03	-	-	-	0.00
Industry 100m	-	-	-	-	-	-	0.16	0.11	0.23
Industry 150m	-	0.41	0.39	-	-	0.12	-	0.15	-
Industry 250m	-	-	-	-	-	0.16	0.02	-	-
Industry 750m	-	-	-	-0.32	-0.18	-	-	-	-
Industry 1000m	0.10	-	-	-	-	0.01	-	-	0.11
Water 25m	-	-	-	0.00	-	0.00	-	-	-
Water 50m	0.03	-	-	-	-	0.17	-	-	-
Water 75m	-	-	0.13	0.18	0.20	-	-	0.12	-
Water 100m	-	-0.17	-0.16	-0.23	-	-	-	-	-
Water 250m	-	-	0.05	0.01	-	-	-	-	0.23
Water 1000m	-0.73	-0.03		-0.34	-0.21	-0.79	-	-0.19	-0.47
Urban 100m	-	-	-	-	-0.12	-0.45	-	-	-
Urban 150m	-0.13	-	-	-0.66	-0.05	-	-	-0.05	-
Urban 1000m	0.28	0.42	0.51	0.23	0.79	0.50	-	-	0.06
Nature 100m	-	-	-	0.27	0.04	-	-	-	-
Nature 150m	0.38	-	-	0.02	0.15	-	-	-	-
Nature 500m	-	-0.83	-0.39	-1.04	-0.51	-0.58	-0.42	-0.47	-0.50
Nature 750m	-1.48	-0.43	-	-	-	-0.04	-	-	-
Agriculture 500m	-	-	0.29	0.35	-	-	-	-	-
Agriculture 1000m	-	-	-0.54	-1.22	-	-	-	-	-
Population 25m	-0.02	-	-0.04	-	-	0.06	-	-	-
Population 50m	-0.09	-	-0.15	-	-	-	-	-	-
Population 75m	-	-	0.09	-	-	-	-	-	-
Population 100m	0.42	0.13	0.35	0.22		0.43	0.14	0.14	0.16
Population 150m	-	-	-	-	0.21	-	-	-	-
Population 250m		0.25		0.09	0.31	0.10	0.08	0.12	0.19
Population 500m	-	-	0.22	-	-	-	-	-	-
Population 1000m	0.64	0.10	0.00	-	-	-	-	-	-

Table S1 continued

Predictors	2000	2001	2002	2003	2004	2005	2006	2007	2008
Class1 road 25m	1.51	1.57	1.92	2.10	1.84	2.20	2.21	2.32	2.42
Class1 road 50m	-	-	0.22	-	-	-	-	-	-
Class1 road 75m	1.46	1.21	0.45	0.40	0.07		0.48	0.97	0.90
Class1 road 100m	-	0.20	0.10	0.43	1.11	0.93	0.14	0.07	0.20
Class1 road 150m	-	-	-	0.12	0.10	0.01	-	-	-
Class1 road 250m	-	-	-	0.20	-	-	-	-	-
Class1 road 500m	-	0.20	-	-	-	-	-	-	-
Class1 road 750m	-	-	-	-	-	-	0.03	-	-
Class1 road 1000m	-	-	0.03	0.14	0.12	-	0.07	-	-
Class2 road 25m	0.27	0.54	0.45	0.53	0.39	0.16	-	0.09	0.13
Class2 road 50m	0.52	0.21	0.28	0.28	0.28	0.73	0.37	0.72	0.65
Class2 road 75m	0.05	-	0.28	0.09	-	-	-	-	-
Class2 road 100m	-	-	-	0.30	0.27	-	0.02	-	-
Class2 road 150m	-	-	-	-	-	-	-	-	0.00
Class2 road 500m	-0.48	-0.25	-0.22	-	-	-	-	-	-
Class2 road 750m	-	-	-	-	-	-	-	-	-0.04
Class2 road 1000m	-	-	-	0.28	-	-	-	-	-
Class3 road 25m	-	0.16	0.33	-	0.12	-	-	-	-
Class3 road 50m	0.24	-	-	-	-	-0.17	-	-	-
Class3 road 75m	-	-	-	-	-	-	-0.06	-0.12	-
Class3 road 100m	-	-	-	-	0.02	-	-	-	-
Class3 road 150m	-0.29	-	-0.02	-0.03	-	-	-	-	-
Class3 road 250m	-	-0.30	-0.06	-	0.00	-	-0.03	-0.25	-0.34
Class3 road 500m	-	-	-	-	-	-	-	-	-0.17
Class3 road 750m	-	-	-	-	0.01	-0.24	-	-	-
Class3 road 1000m	-	-	0.47	0.63	-	-0.14	-	-	-
Highway 25m	0.13	0.42	0.72	0.24	0.16	0.33	0.78	0.71	0.97
Highway 50m	0.19	-	-	0.04	-	0.50	0.36	-	0.20
Highway 75m	1.33	1.04	0.71	0.47	0.80	-	0.15	0.80	0.33
Highway 150m	-	-	-	0.43	0.75	0.64	0.34	-	-
Highway 250m	-	1.03	0.72	0.84	0.75	0.15	-	-	-
Highway 500m	0.32	0.06	-	-	0.58	-	0.11	0.28	0.50
Highway 750m	-	0.18	0.26	0.29	-	0.39	0.62	0.40	-
Highway 1000m	-	-	0.38	1.22	0.65	0.23	0.08	0.30	0.12

Table S1 continued

Predictors	2000	2001	2002	2003	2004	2005	2006	2007	2008
Building density 25m	0.34	-	0.27	0.22	-	0.14	-	0.00	-
Building density 50m	0.27	0.36	0.31	0.24	0.34	0.65	0.30	0.49	0.67
Building density 75m	0.86	0.43	-	0.15	0.16	0.01	0.12	0.33	-
Building density 100m	-	-	-	-	-	-	0.35	0.10	-
Building density 150m	0.40	0.59	0.85	0.95	1.13	1.23	0.76	0.77	1.14
Building density 250m	-	-	-	-	-	-	-	-	0.09
Building density 750m	-	-	-	-	-	-	0.31	-	-
Building density 1000m	-	0.00		0.46	0.89	0.80	1.04	1.22	0.71
N trucks 25m	1.37	1.09	0.56	1.05	1.04	0.92	0.03	0.17	0.12
N trucks 50m	-	-	-	-	-	0.15	0.20	0.26	0.32
N trucks 75m	-	-	-	-	-	-	0.06	-	0.00
N trucks 150m	-	-	-	-	-	-	-	-	0.13
N trucks 250m	0.19	0.14	-	0.09	0.08	0.19	-	0.37	0.28
N trucks 750m	-	-	-	-0.18	-	-	-	-	0.13
N trucks 1000m	-	-	-0.06	-0.07	-	-	-	-	-
N motorbikes 25m	-0.29	-	-	-0.38	-0.05	-0.47	-	-	-
N motorbikes 50m	0.09	0.14	0.35	0.39	0.29	-	-	0.16	-
N motorbikes 75m	-0.15	-	-	-	-	-0.14	-	-	-
N motorbikes 100m	-	-	-	-	-	-	0.61	0.02	0.45
N motorbikes 150m	0.30	-	-	0.27	-	0.25	0.06	-	-
N motorbikes 250m	-	-	-	0.05	-	-	-	-	-
N motorbikes 500m	0.23	0.01	-	-	-	-	-	-	-
N motorbikes 1000m	-	0.10	-0.07	-	-	0.00	-	-	0.15
N cars 25m	-	-	-	-	-	-	0.02	-	0.19
N cars 50m	-	-	-	-	-	0.23	0.19	-	-
N cars 75m	-	-	-	-0.44	-0.17	-	-	-	-
N cars 1000m	-	-	-0.47	-0.17	-	-	-	-	-

Table S2: NO_2 model building and validation statistics by year

Year	Cantonal passive sampling data				NABEL data		
	N sites training	Elastic net cross-validation R^2	Elastic net + kriging prediction R^2	Elastic net + kriging SEE	N sites for validation	Validation R^2	RMSE
2000	980	0.60	0.89	3.53	78	0.73	5.83
2001	886	0.61	0.89	3.31	81	0.78	4.79
2002	975	0.69	0.90	3.13	86	0.82	4.20
2003	1308	0.70	0.87	3.88	93	0.70	6.06
2004	947	0.65	0.87	3.46	99	0.75	5.15
2005	1169	0.68	0.88	3.44	98	0.75	5.32
2006	1097	0.65	0.85	3.72	114	0.71	5.65
2007	1145	0.64	0.84	3.75	135	0.72	5.64
2008	1194	0.63	0.85	3.67	146	0.74	5.39

Predictors and modeling strategy of the $PM_{2.5}$ model

The development of the PM2.5 model is described in detail in de Hoogh et al, 2017.[8] Global and local land use predictors for the PM2.5 model included:

- $PM_{2.5}$ emissions from agriculture, households, industry, traffic and wood smoke for the years 2005 and 2010 obtained from MeteoTest at a 200×200 m grid.
- Distance to nearest main road, computed from the VECTOR25 road network.
- Elevation from the DHM25 at a 200×200 m grid.
- Land use from the European CORINE land cover at 100×100 m resolution.
- Meteorological data including daily modelled planetary boundary layer data, daily temperature, wind speed, wind direction and precipitation at a ~10×10 km resolution from 1 January 2003 until 31 December 2013 from the European Centre for Medium-Range Weather Forecasts.
- Normalized Difference Vegetation Index in a 30×30 m raster for year 2014, mosaicked from scenes downloaded from the USGS EarthExplorer website (http://earthexplorer.usgs.gov).[9]

A 4-staged modelling approach[10] was adapted and further developed to calibrate the satellite data with measured $PM_{2.5}$ concentrations. The measured $PM_{2.5}$ dataset consisted of 10 daily measurement sites. To increase the number of measurement sites we applied the ratio between $PM_{2.5}$ and PM_{10} data (daily; Jan 2003 - April 2010 and four-daily; April 2010 – Dec 2013) from 10 co-located monitoring sites of the NABEL network to predict $PM_{2.5}$ concentrations at sites where only PM_{10} measurements were available, supplementing the monitoring dataset with an extra 89 measurement sites. At the 1km scale, mixed effect models (stage 1) were generated regressing $PM_{2.5}$ measurements against day-specific random intercepts, fixed and random AOD and boundary layer height slopes, and fixed effects for spatial covariates. These mixed effect models were then used to predict $PM_{2.5}$ in cells were AOD was available, but without a $PM_{2.5}$ measurement (stage 2). Next, a generalized additive mixed model with spatial smoothing was applied to generate grid cell predictions for those grid cells where AOD was missing (stage 3). To estimate 100m localized $PM_{2.5}$ predictions, the residuals from the stage 1 model at each monitoring site were regressed against the local spatial and temporal variables at each monitoring site (stage 4) using machine learning techniques. This resulted in a $PM_{2.5}$ model for each year for the period 2003-2008.

References

8. de Hoogh K, Héritier H, Stafoggia M, Künzli N, Kloog I. Modelling daily PM2. 5 concentrations at high spatio-temporal resolution across Switzerland. *Environmental Pollution* 2017.
9. Vienneau D, de Hoogh K, Faeh D, Kaufmann M, Wunderli JM, Röösli M, Group SNCS. More than clean air and tranquillity: Residential green is independently associated with decreasing mortality. *Environ Int* 2017; 108:176-184.
10. Kloog I, Chudnovsky AA, Just AC, Nordio F, Koutrakis P, Coull BA, Lyapustin A, Wang Y, Schwartz J. A new hybrid spatio-temporal model for estimating daily multi-year PM2.5 concentrations across northeastern USA using high resolution aerosol optical depth data. *Atmospheric Environment* 2014; 95:581-590.

Figure S1: Distribution of mean NO_2 (2000 to 2008) and mean $PM_{2.5}$ (2003 to 2008) exposure in the study population.

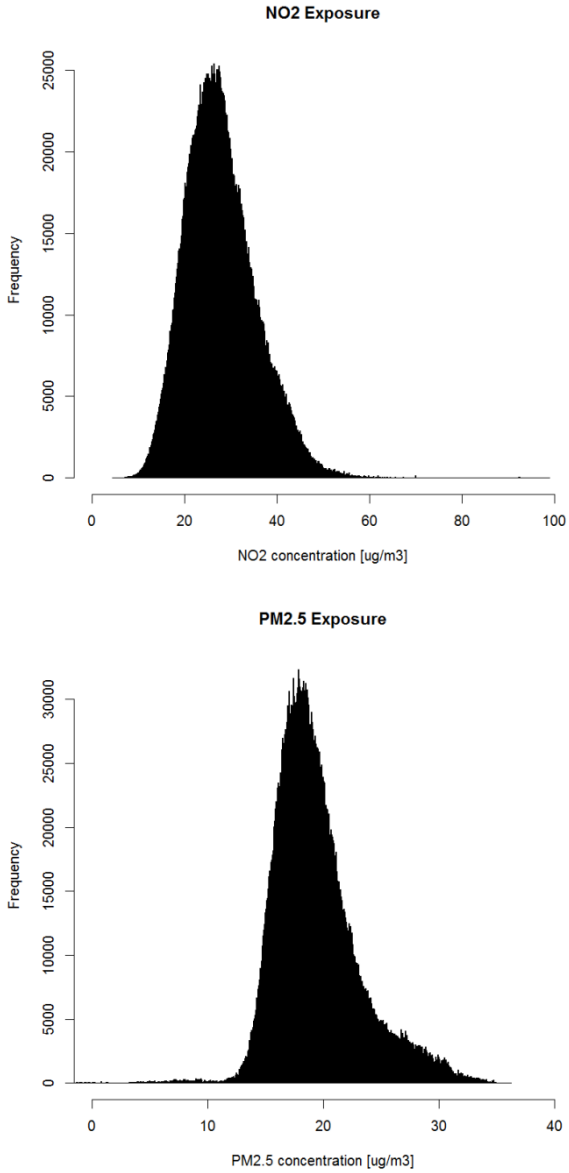

Table S3: linear HRs for associations between road, railway, and aircraft noise exposure and MI per 10 dB increase in Lden for models: crude, adjusted for sociodemographics but not air pollution (AP), adjusted for $PM_{2.5}$, adjusted for NO_2, and adjusted for $PM_{2.5}$ and NO_2

Noise source	Crude[1]	Adjusted (no AP) (95% CI)[2]	Adj for $PM_{2.5}$ (95% CI)[2]	Adj for NO_2 (95% CI)[2]	Adj for $PM_{2.5}$ and NO_2 (95% CI)[2]
Road	1.039 (1.020-1.057)	1.032 (1.014-1.051)	1.031 (1.013-1.051)	1.034 (1.014-1.055)	1.034 (1.014-1.055)
Railway	1.024 (1.011-1.037)	1.020 (1.007-1.033)	1.019 (1.007-1.033)	1.020 (1.008-1.034)	1.020 (1.007-1.033)
Aircraft	0.971 (0.952-0.990)	1.025 (1.006-1.045)	1.024 (1.004-1.045)	1.026 (1.006-1.047)	1.025 (1.005-1.046)

[1] age as the underlying time scale

[2]Models additionally adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and the other noise sources.

Table S4: categorical HR for road traffic noise exposure and MI for models: not adjusted for air pollution (AP), adjusted for $PM_{2.5}$, adjusted for NO_2, and adjusted for $PM_{2.5}$ and NO_2

Road traffic	≤45 dB	45-50 dB	50-55 dB	55-60 dB	60-65 dB	>65 dB
Adjusted (no AP)	1	1.040 (0.982-1.100)	1.057 (1.002-1.115)	1.087 (1.029-1.148)	1.072 (1.012-1.135)	1.093 (1.028-1.162)
Adj for $PM_{2.5}$	1	1.039 (0.982-1.100)	1.057 (1.002-1.115)	1.086 (1.028-1.148)	1.071 (1.011-1.135)	1.092 (1.026-1.162)
Adj for NO_2	1	1.041 (0.983-1.102)	1.060 (1.004-1.118)	1.091 (1.031-1.153)	1.077 (1.015-1.142)	1.101 (1.032-1.176)
Adj for $PM_{2.5}$ and NO_2	1	1.041 (0.983-1.102)	1.059 (1.004-1.118)	1.090 (1.031-1.153)	1.077 (1.015-1.142)	1.102 (1.032-1.177)

Age as the underlying time scale and additionally adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and railway and aircraft noise.

Table S5: categorical HR for railway noise exposure and MI for models: not adjusted for air pollution (AP), adjusted for $PM_{2.5}$, adjusted for NO_2, and adjusted for $PM_{2.5}$ and NO_2

Railway	≤30 dB	30-40 dB	40-50 dB	50-60 dB	>60 dB
Adjusted (no AP)	1	1.030 (0.991-1.072)	1.066 (1.026-1.107)	1.034 (0.984-1.087)	1.070 (1.009-1.134)
Adj for $PM_{2.5}$	1	1.030 (0.989-1.071)	1.065 (1.024-1.107)	1.033 (0.982-1.086)	1.068 (1.007-1.133)
Adj for NO_2	1	1.033 (0.993-1.076)	1.069 (1.028-1.112)	1.036 (0.985-1.090)	1.072 (1.011-1.137)
Adj for $PM_{2.5}$ and NO_2	1	1.032 (0.992-1.075)	1.068 (1.027-1.111)	1.035 (0.984-1.088)	1.070 (1.009-1.135)

Age as the underlying time scale and additionally adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and road and aircraft noise.

Table S6: categorical HR for aircraft noise exposure and MI for models not adjusted for air pollution (AP), adjusted for $PM_{2.5}$, adjusted for NO_2, and adjusted for $PM_{2.5}$ and NO_2

Aircraft	≤30 dB	30-40 dB	40-50 dB	50-60 dB	>60 dB
Adjusted (no AP)	1	1.001 (0.952-1.053)	1.048 (1.007-1.091)	1.050 (0.978-1.127)	0.979 (0.827-1.159)
Adj for $PM_{2.5}$	1	1.000 (0.951-1.053)	1.047 (1.005-1.091)	1.049 (0.976-1.127)	0.978 (0.826-1.158)
Adj for NO_2	1	1.003 (0.953-1.056)	1.053 (1.009-1.098)	1.053 (0.980-1.131)	0.981 (0.829-1.161)
Adj for $PM_{2.5}$ and NO_2	1	1.002 (0.952-1.055)	1.051 (1.008-1.096)	1.051 (0.978-1.129)	0.978 (0.826-1.158)

Age as the underlying time scale and additionally adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality and road and railway noise.

Table S7: linear HR for $PM_{2.5}$ and NO_2 (per 10 µg/m³) for MI in single exposure models, in models adjusted for noise, and in models adjusted for noise and the opposite air pollutant.

Air pollutant	Crude[1]	Single exposure model[2]	Adj for all noise sources[2]	Adj for all noise sources and for the other air pollutant[2]
$PM_{2.5}$	0.991 (0.955-1.029)	1.052 (1.013-1.093)	1.010 (0.969-1.052)	1.019 (0.971-1.071)
NO_2	0.968 (0.950-0.986)	1.024 (1.005-1.043)	0.996 (0.974-1.018)	0.990 (0.965-1.016)

[1]age as the underlying time scale

[2]Models additionally adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, and nationality.

Table S8: Adjusted* HR (95% confidence intervals) and number of deaths (N=) for NO_2 and road traffic noise for MI in categorical (quartiles) interaction exposure models. Cells with interaction terms[$] are shown in italics.

		Road traffic noise [dB]			
		≤49.0 (main effect) N=4249	**49.0-54.1 N=4775**	**54.1-60.3 N=5016**	**>60.3 N=5221**
NO_2 [µg/m³]	**≤22.3 (main effect) N=4728**	1.00 (reference) N=1834	1.04 (0.97-1.12) N=1287	1.05 (0.97-1.13) N=999	1.05 (0.96-1.15) N=608
	22.3-27.0 N=4643	1.02 (0.95-1.10) N=1181	*0.97 (0.87-1.08) N=1282*	*1.02 (0.92-1.15) N=1223*	*1.11 (0.98-1.26) N=957*
	27.0-32.3 N=4851	1.05 (0.96-1.14) N=848	*1.01 (0.90-1.13) N=1335*	*1.00 (0.89-1.12) N=1413*	*0.97 (0.85-1.10) N=1255*
	>32.3 N=5039	1.02 (0.91-1.14) N=386	*0.98 (0.85-1.12) N=871*	*1.00 (0.87-1.15) N=1381*	*0.98 (0.85-1.13) N=2401*

[*] Age as the underlying time scale and additionally adjusted for sex, neighborhood index of socio-economic position, civil status, education level, mother tongue, nationality, railway and aircraft noise, NO_2.

[$] Overall effects ($HR_{tot}$) in cells with interaction terms are obtained by multiplying the HR of these cells with the HRs of corresponding main effects (E.g. HR_{tot} for people in the highest NO_2 (>32.3 µg/m^3) and highest noise category (>60.3 dB): 0.98*1.02*1.05=1.05)

9. Summary of the main findings

The main findings from the sections 4. to 8. are summarized here. The objectives of this thesis are defined by the research questions in section 3.1. and will be briefly answered here. Detailed results and discussions are given in the respective chapters. More general aspects of the thesis are discussed in section 10.

9.1. Summary of results of section 4.

Over 4000 individuals were contacted for the QUALIFEX study, 1375 individuals participated in the baseline and 1122 returned the follow-up questionnaire one year later, thus accounting for 2497 observations. Mean exposure Ldn noise levels (SD) to road traffic was 52.0 dB (6.2). The proportion of considerable and heavy annoyed subjects was highest for aircraft (21.4%), road traffic (13.8%) and neighbour noise (10.2%) and smaller for railway (2.4%) and industry noise (1.9%).

A 10 dB increase of the road traffic noise Ldn was associated with a 0.47 (95% CI: −0.01, 0.95) point increase of the von Zerssen symptom score meaning an increase of symptoms and thus worsened physical HRQOL. A substantial increase in the von Zerssen symptom score for annoyance to road, industry and neighbour noise was observed.

The SF-36 mental health score was not associated with road traffic noise, whereas it was positively associated with most types of annoyance with the exception of annoyance to aircraft noise. An increase in annoyance category meant a worsening of mental HRQOL.

Search for interaction revealed that sleep disturbance was an important effect modifier of the relation with the von Zerssen score. No variable was found to modify the relationships with the SF-36 mental health score.

In the SEM, upon inclusion of other explanatory variables, the direct association between road traffic noise and the von Zerssen score vanished. Road traffic noise was indirectly associated with the von Zerssen, while the annoyance and sleep disturbance were directly associated with this indicator. The second SEM for the SF-36 revealed that the direct relationship between road traffic noise and the SF-36 was not robust to the addition of other explanatory variables. Road traffic noise and annoyance were indirectly associated with the SF-36, while sleep disturbance was directly associated with this indicator.

9.2. Summary of results of section 5.

The study population amounted to 4.41 mio observations and the number of deaths from CVD was 142,955, of which 42.2%, 15.7% and 9.4% were deaths from IHD, stroke and BP-related diseases, respectively. The proportion of the population exposed to Lden noise levels above 45 dB was 89%, 25% and 14% for road traffic, railway and aircraft noise, while more than 39% of the population were living in areas with highly intermittent noise (i.e. exposed to IR above 80%). Road traffic noise was

significantly associated with CVD, BP, IHD, MI, heart failure and ischemic stroke. The risk increase for CVD, BP, IHD and heart failure were 2.5% (95% CI: 1.8-3.2%), 5.3% (95%CI: 3.0-7.5%), 2.3% (95% CI: 1.2-3.4%) and 5.1% (95% CI: 2.7-7.4%) per 10 dB increase in road traffic noise. For railway noise the same tendencies than for road traffic were observed but the associations were weaker. Aircraft noise was significantly associated with MI, heart failure and ischemic stroke for which the risk increases were 2.7% (95% CI: 0.6-4.9%), 5.6% (2.8-8.5%), 7.4% (2.0-2.7%), respectively. Categorical analysis revealed that the risk for BP started to increase from levels as low as 30 dB for railway noise.

For IR, a bell-shaped relationship was observed with stronger associations in the 3rd and 4th quintiles. The risk increase for CVD per 10% increase of IR were 1.9% (95% CI, 0.2-3.7%) and 2.1% (0.3-3.8%) in the 3rd and 4th quintiles, respectively.

Stratification analyses yielded stronger associations between road traffic and railway noise exposure, and CVD for subjects with more than 5 years of residence in the same community. With regard to gender, males tended to be at higher risk for CVD than females for exposure to road traffic and railway noise.

9.3. Summary of results of section 6.

The study population amounted to 4.41 mio observations and the number of deaths from CVD was 142,955, of which 42.2%, 15.7% and 9.4% were deaths from IHD, stroke and BP-related diseases, respectively. During the core night hours from 01 to 05h and during daytime hours from 07 to 19h, 37.7% and 89.6% of the study population were exposed to Leq,Comb transportation noise levels above 45 dB. Spearman's rank correlations between the different Leq,Comb for different time windows were higher than 0.94. With the exception of stroke, all outcomes were associated with Leq,Comb in the different time windows. For BP, IHD, and heart failure, the risk described a sinus-shaped relationship, increasing and decreasing throughout the day. For CVD and IHD a diurnal pattern was found with highest HR during the core night (01-05h). For heart failure, a reversed pattern was found, where the highest HR was seen during the day (07-19h), while for BP highest HR was seen in the morning (06-07h). Categorical analysis showed that the risk for BP started to increase during the core night (01-05h) from Leq,Comb levels as low as 35 dB.

With the exception of heart failure, IR at night rather than during day was found to be more strongly associated with the outcomes. For most outcomes, the risk for IR night described a bell-shaped relationship with highest values in the 3rd and 4th quintiles, while for IR day the highest risk values tended to be shifted towards the 2nd and 3rd quintiles.

9.4. Summary of results of section 8.

The study population amounted to 4.40 mio observations and the number of deaths from MI was 19,261. For road traffic noise, neither adjustment for NO_2 nor for $PM_{2.5}$ did change the risk estimates. Similarly, adjusting for both sources did not significantly attenuate the estimates. For railway and

aircraft noise, adjustment for both air pollutants did not attenuate the risk estimates for MI. Categorical models showed that the increase of the estimates for road traffic noise, and MI was more pronounced in the higher noise categories 60-65 dB and >65 dB. The increase was mainly driven by adjustment for NO_2. For railway noise and both outcomes, estimates were robust to adjustment to air pollution. For aircraft noise the categorical risk estimates for MI exhibited a slight increase in all noise categories below 60 dB. This increase was more marked when adjusting for NO_2.

10. General discussion and conclusions

In the following section, the findings of the present work are compared to the international pee-reviewed scientific literature and their implications for further research are examined. Finally, the public health relevance of this work will be discussed.

10.1. Noise annoyance and HRQOL

Annoyance to road traffic has been reported to be more strongly associated with decreased activity than with objective road traffic noise itself (57). This thesis strengthens this web of relationship between noise, annoyance to noise and health effects by reporting that annoyance from road traffic noise seems to be more strongly associated with poor health than physical exposure to road traffic noise itself. In terms of physical and mental health, two different studies with road traffic noise (58) or annoyance to road traffic noise (42) as exposure variable and the SF-36 questionnaire as response variable for assessing physical and mental health have highlighted this phenomenon. Indeed, while an association of road traffic noise with physical and mental health could not be demonstrated (58), annoyance to this particular noise source was significantly associated with physical and mental health (42). With regards to this HRQOL study, using the SF-36 for assessing mental health as in the previous mentioned studies, the same tendencies could be highlighted. Moreover, assessing physical health using a different indicator (von Zerssen (52)), a weaker association for road traffic noise compared to annoyance to road traffic noise was still found.

In order to clarify the direct and indirect effects between road traffic noise, annoyance to road traffic and HRQOL indicators, an analysis was conducted using structural equation modelling. Two previous studies by Fyhri et al. (59, 60) indicate that 1) there is no direct link between road traffic noise and health outcomes and 2) annoyance is the mediator of further health effects, of which sleep disturbances belongs to. The findings of this thesis confirm 1) and 2) and further indicate that road traffic noise is indirectly associated with HRQOL. Furthermore, the results of the HRQOL study indicate that annoyance influences physical health directly and indirectly via sleep disturbances, while mental health is influenced only indirectly by annoyance via sleep disturbances.

Overall, the findings indicate a strong relationship between annoyance to road traffic noise and HRQOL but do not exclude a link between road traffic noise and HRQOL. The fact that the direct association between road traffic noise and health effect vanishes when considering annoyance could be due to the use of self-reported health indicators. More annoyed people may tend to have more health complaints and this would introduce a bias, thus cancelling the relationship between road traffic noise and health. This is in line with a previous analysis conducted on the same data that found no direct association between road traffic noise exposure and subjective sleep quality (61). However, objective sleep quality measured by actimetry was significantly associated with road traffic noise exposure.

10.2. Road traffic noise

Section 5. of this work shows that road traffic noise is a significant risk factor for heart failure, BP-related death, ischemic stroke and MI. With one exception (99), previous studies focusing on road traffic noise and stroke produced inconclusive results (42, 95-97). In this thesis, ischemic stroke but not hemorrhagic stroke was linked with road traffic noise. This will be discussed in section 10.6.2. Slightly lower risk and a lower threshold were found compared to previous meta-analysis (62-64). Indeed, while one of these meta-analysis reported pooled risk estimates for CVD with regards to road traffic noise of 1.08 (95% CI: 1.04-1.13) (65) this thesis reports a smaller risk of 1.025 (95% CI: 1.018-1.032). The difference may due to population characteristics, as well as uncertainties regarding confounding effects and exposure misclassification that will be the topic of section 10.7 and section 10.8 of this thesis. In summary the association between road traffic noise and CVD appears as an established fact confirmed by a report from the WHO (11).

Given the ubiquity of road traffic noise exposure, even a small increase in exposure levels could lead to an increase in the prevalence of CVD in the population. For this reason many governments have already taken some measures to tackle the problem. The most common approach in Switzerland aims at lowering the noise levels at the source. This comprises fostering the availability of silent tires (66), installation of low noise road pavement (66), speed zones (67), and vehicle taxing (66). More radical measures have been adopted in developing countries, in the city of Cairo, Egypt, a ban on horn, trucks and noisy buses has permitted to lower L_{eq} noise levels from road traffic by 6.0 to 10.2 dB in residential areas (68). Similarly, a ban on rickshaws in Lanzhou, China has decreased road traffic noise L_{eq} by 1.7dB (69).

Noise levels are expected to be reduced with the advent of electric cars. Indeed, a simulation showed that a 90% electrical vehicle fleet would lead to an overall 3 dB reduction of noise levels, while the proportion of annoyed and severely annoyed people would decrease by 33 and 36%, respectively (70). Further, in the industrialized nations, rate of driver's licenses follows a decreasing trend in the generations of the millennials (born between 1979 and 2000) (71). As a consequence, car ownership and therefore the total number of kilometers driven is decreasing (72). It is unclear whether these change will hold due to the advent of the mobile technologies (73), or will vanish once millennials will have access to more economic resources (74). However, the momentum towards fewer cars in the cities has already begun. In order to prevent traffic congestion, cities like Stockholm or London have introduced a tax that limits the number of cars entering the central urban area (75, 76). In Switzerland, between 2000 and 2010, the share of households without a car in the cities of Basel and Bern has increased from 45.3 to 54.9% and from 42.2 to 53.2% (77). Following the trends in north European cities like Amsterdam and Copenhagen, in Swiss cities more and more people use bicycles for moving

around. This translates in less cars on the road and therefore in a reduction of noise levels. In Geneva, between 2001 and 2010 a 128% increase of the number of cyclists has been observed (78). All these tendencies are expected to reduce exposure to road traffic noise in cities. A reduction of road traffic has been shown to decrease annoyance and activity disturbance, while an increase in general well-being was observed (79), and it seems plausible that the incidence of CVD decreases with less noise exposure.

In the developing world a reversed trend arises; increased urbanization and wealth fuel the rates of car ownership (80). Indeed, in China, Chile, Mexico, Korea and Thailand, the number of motor vehicles is increasing by more than 10% annually (80). As reported by a Vietnamese study, road traffic noise levels are in tendency higher in Vietnam than in Japan (13). The urban populations in developing countries are already vulnerable in terms of lifestyle due to changes in social relationships, and enhanced exposure to microbiological hazards due to poor sanitation facilities (81). In those settings the projected increase of road traffic noise will render the burden of CVD heavier.

10.3. Railway noise

The study presented in section 5. is the first study examining the link between railway noise and CVD mortality. This study reports an association between railway noise and IHD, MI and CVD as primary cause of death. Categorical models show an increase of risk from noise levels as low as 30dB. This finding has a particular meaning in terms of noise protection policy. Indeed, the current night noise guidelines from WHO recommend a long-term target of 45 dB (11) at the most exposed façade, and, according to the results this value would still expose the populations to high increased risk for CVD. A German study conducted in the surroundings of Frankfurt airport reported an association between railway noise, and blood pressure and heart failure (82), which could not be confirmed.

This thesis shows that railway noise induce less deleterious effects than road traffic noise. This is in line with the less steep exposure response curve of annoyance to railway noise compared to other transportation noise sources (27). Further, railway noise in Switzerland is quite predictable. Indeed, given the high punctuality of the Swiss railway company, people living close to railway tracks may anticipate their exposure and develop coping strategies. Further, noise mitigation strategies may contribute to lower the risk estimates for CVD. Without noise mitigation, freight trains have been shown to produce stronger cardiac responses (83) during sleep than passenger trains. Switzerland is at the lead in term of railway noise mitigation in Europe. Indeed according to Swiss railways a large share of the goods wagons fleet have been equipped with special brakes, while noise barriers are being built along railway lines. A plan has been set to ban noisy freight wagons from the Swiss rail network by 2020 (84).

In China the railway network length has grown by 7% in average between 2009 and 2015 (85). In many parts of the world, railways tracks are built for noisy high speed trains. Recently, China has

decided to ship a larger share of its exportation to Europe by train. Without noise mitigation strategies, an increase of the number of freight trains may increase noise levels and therefore lead to a higher incidence of CVD cases.

10.4. Aircraft noise

The risk estimates for MI reported for aircraft noise in section 5., confirm the results from Huss et al. (86). Further, an association with ischemic stroke and heart failure could be demonstrated.

In the two largest Swiss airports of Geneva and Zürich a ban on night operations restricts the number of night-time aircraft movements between 00:30 and 05:00, and 23:30 and 06:00, respectively. Given these restrictions on air traffic, the results more likely reflect the impact of daytime aircraft noise exposure. Further, as in the NORAH study (82), it can be shown that aircraft noise has less impact on CVD than road traffic or railway noise.

In the study presented in section 5., aircraft noise exposure has been found to be protective against cardiovascular diseases in general and ischemic heart diseases. This contra-intuitive effect can be explained in many different ways. The proportion of foreigners (first and second generation migrants) is higher around the airports. Indeed, based on data from the Federal Office of Statistics (87), the proportion of foreigners living in municipalities within the 55 dB aircraft noise contour for the airports of Zürich, Bern, Geneva, and Samedan was calculated. In 2014, the proportion of foreigners was 31.5% while it was 24.3% at the national level. Yet a study conducted with the SNC data (88) shows that first and second generation migrants live longer than Swiss nationals. Though the models were adjusted for nationality, there was not enough information to adjust for second generation migrants that may have already undergone the naturalization process and hence, are considered Swiss nationals. In addition, self-selection may play a role in explaining the protective effect of aircraft noise exposure: subjects not able to cope with aircraft noise tend to move away from noisy areas, while subjects with enough psychological resources tend to habituate to aircraft noise. Further, in Switzerland, airports have to subsidize the installation of sound-proof windows in the area exceeding certain noise levels. Better sound insulation may lead to a large overestimation of the indoor noise exposure. In addition, the aircraft noise model used in this study is based on noise level contour data, which may introduce exposure misclassification. Indeed, people living on the quiet side of a building may have been considered as highly exposed while in fact the building would have a shielding effect.

Aircraft noise has been at the center of the media attention for years in Switzerland. While a substantial effort has been made to mitigate the deleterious effects of aircraft noise by airport authorities, activist citizens are still requiring an extension of the existing night curfews. This thesis demonstrates a non-negligible impact of aircraft noise on health in a country where virtually no night operations are allowed between 00:30 and 05:00. Most major European airports have adopted mitigations strategies such as allowing noisy aircraft during daytime only as well as night curfew (89)

to protect the surrounding populations. However, a French exception emerges from this picture; Indeed, at Roissy Charles-de-Gaulle airport – the largest French airport- a maximum of 55 night flights are allowed between 00:00 and 05:00 (89). In general, in western countries, the issue has been taken seriously and substantial progress has been made to reduce aircraft noise nuisances. In addition, initiatives to reduce aircraft noise at the source are bourgeoning (90). Today's aircraft are much less noisy than last generation ones. Indeed the Airbus A-380, which is the current largest aircraft launched in 2005 has been shown to generate less noise than the Boeing B747-400 launched in 1988 (91). These trends will contribute to alleviate the noise exposure in the surroundings of airports.

The situation is a matter of concern in fast developing countries where the health of the populations is often considered after the economic interests. At the Beijing Capital International Airport, Beijing, China and Chhatrapati Shivaji International Airport, Mumbai, India about 150 landings and take-offs are scheduled each night between 00:00 and 06:00 (92). Both airports are located next to dense urban areas. Figure 4 shows that two hospitals and a dense residential area are located in the axis of one of Mumbai's airport runways. Given that China and India are forecasted to become the first and third world largest aviation markets by 2025, the governments of these nations urgently need to take action in order to protect their populations (93).

Figure 4: Satellite picture of a runway at Chhatrapati Shivaji International Airport, Mumbai, India (Google earth)

10.5. Different exposures – different effects?

This thesis highlights different relationships between noise sources as well as timing of exposure and outcomes. This section is devoted to the discussion of the variety of relationships found between the different exposures variables and the different outcomes.

10.5.1. Timing of exposure

This study presented in section 6. assessed the impact of transportation noise exposure on CVD in different time windows of the day and night. Stronger associations between transportation noise exposure during nighttime from 23:00 to 06:00 for IHD have been found. For heart failure, this pattern was opposite, with higher risk during the day from 06:00 to 23:00. For BP-related deaths, exposure during the morning hours from 05:00 to 06:00 appeared to be more deleterious. The risk for BP was found to increase from noise levels as low as 35 dB during the night from 01:00 to 05:00, while this threshold was 45 dB and 50 dB for IHD and heart failure.

Differences in outcome risk for different time exposure windows may reflect different mechanism in the etiology of those particular outcomes. Indeed, acute coronary events have been shown to occur in the morning hours following sympathetic nervous system activation (94), which induces a cascade of processes leading to thrombus formation mechanisms. Impact of noise exposure during the previous night may act as an additional stressor at wake time. Higher risk for heart failure was found for noise exposure during the day. Deaths by heart failure are caused by sudden cardiac death or acute worsening of ischemia (95), occurring most often during the day. Therefore, it seems plausible that noise exposure during the day induces a stress that acts as a trigger for death by heart failure.

Only a few studies investigated on the impact of noise exposure during the night and the day (42-44). Results are not conclusive, because some found a higher risk of cardiovascular hospital admissions for road traffic during the day (43), while others report stronger effects during the night for hypertension and aircraft noise (44) as well as for systolic blood pressure and railway noise (42). Since the analysis in those studies was not conducted using the same time windows as the study of this thesis, a comparison remains difficult.

10.5.2. Stroke – a special case

Stroke was neither associated with a noise source nor was it associated with a particular time window of exposure to combined transportation noise. Stroke does not share a common etiology with IHD and heart failure (96, 97). Previous studies focusing on road traffic noise and stroke found positive associations (98, 99) while not considering the subgroups of stroke. In this work, no association was found when considering stroke in general. Considering subgroups of this outcome it was shown that ischemic stroke but not hemorrhagic stroke was linked with road traffic noise. This may indicate that different mechanisms are at play in the etiology of stroke and that road traffic noise does not belong to the risk factors influencing hemorrhagic stroke.

10.5.3. Different impact magnitude by source?

The relationship between transportation noise exposure and the annoyance reaction to transportation noise sources has been extensively documented (27, 100). Miedema et al. (27), derived dose-response relationships for aircraft, railway and road traffic noise sources and their respective annoyance. The results reported in the HRQOL study in section 4. are in line with Miedema's findings. Indeed, for the same equivalent noise levels, annoyance to aircraft noise is higher than for road traffic and railway noise (27). Since annoyance to noise has been postulated to be the precursor of further health effects, this raises the question whether different noise sources have different impact magnitude. In terms of physiological effects, aircraft noise has been shown less likely than the other transportation noise sources to induce cortical and vegetative arousal in sleeping subjects (39, 101). The lower risk for CVD observed in section 5. for aircraft noise compared to the other sources is in line with the physiological mechanisms. This implies that a particular noise source is more likely to trigger a particular outcome than others noise sources. Given that aircraft noise exposure in Switzerland is very unlikely to occur during the night, daytime mechanisms may be at play in the etiology of CVD from this noise source. Indeed, high annoyance prevalence reported for aircraft noise shows that aircraft noise generates psychosocial stress in the conscious state which can be further internalized and processed physiologically and somatically (28). The risk for heart failure being higher during the day, may indicate that an annoyance-mediated pathway may play a role in the etiology of this outcome. Further, the prevalence of annoyance to road traffic and railway noise is substantially lower than for aircraft noise, but physiological effects during sleep have been shown to be stronger (39, 101). This would indicate a sleep-mediated pathway for IHD with regards to road traffic and railway noise exposure. The findings from section 5. show that IHD is more strongly associated with these noise sources, while the findings in section 6. showing a higher risk during the core night for IHD further support this view. However, as shown in section 4. an indirect link between annoyance and health outcomes via sleep disturbance may also be present. Further, as discussed in section 10.1, results shows that for different outcomes, different pathways may be at play. This pathway-mediated impact hypothesis deserves further research.

Further, this raises the question whether for a given noise source different impact magnitudes can be expected? This can be reformulated as follows; is exactly the same impact of noise expected regardless of the noise characteristics and the noise levels? As shown in the study from Phan et al. (13), there can be substantial differences in terms of road traffic noise levels and characteristics between countries. More insight on the impacts of noise characteristics on health are discussed in section 10.6. of this work.

10.6. Gender specific impacts

Stratification analyses conducted in section 5. of this work show gender-specific responses to transportation noise exposure. In general, males were more affected by transportation noise than their

counterparts. With regards to MI, the results of this thesis confirm the findings from other studies (47, 102, 103). While the findings from the literature in terms of vulnerability for hypertension yield mixed results (104-106), this thesis shows that for BP-related deaths, males are more vulnerable to all transportation noise. Interestingly, females are more vulnerable to noise exposure in terms of changes in waist circumference (107-109), a marker for obesity, which may lead to diabetes for which females tend to be more vulnerable than men (108, 110). These findings suggest that different mechanisms are at play; leading to either higher risk of CVD for males or diabetes mellitus for females. Females tend to report higher annoyance levels (102, 111), and annoyance has been associated with a decrease in physical activity (57), thus leading to weight increase and obesity (107-109). Different noise exposure pattern between genders could also be an explanation; A study reports stronger association for males with MI and work sound levels, while environmental noise levels affected both gender (112).

10.7. **Refining the exposure: Intermittency ratio**

The IR was introduced based on the hypothesis that highly peaking noise sources would be more detrimental for health. The results surprisingly show that for most outcomes IR values in the 3rd and 4th quintiles are more harmful to health. The soundscape in those quintiles categories is a mix between background noise and intermittent noise sources as opposed to the 1st and 2nd quintiles where background noise is continuous and almost no single event can be distinguished. In the 5th quintile, each single event can be distinguished. The median number of noise events during the night can support understanding of the soundscape in the different quintiles: 139 and 173 in the 1st and 2nd quintiles, while it was 123 and 99 in the 3rd and 4th quintiles and finally it was 58 in the 5th quintile. The median number of noise events in the latter quintile shows that exposure to noise is mostly intermittent with periods of silence in-between. Based on the hypothesis that silence between two noise events is more relevant for health than the noise events themselves, this explains why high IR has been found to be less strongly associated with the outcomes.

The purpose of the IR was to capture the impact of noise variability on health that is otherwise not captured by average noise levels (113). In other terms, IR allows to refine the exposure. This refinement aimed to capture the remaining variance not captured by average noise levels variables. This refining effect of the IR is particularly obvious when observing the effect of the inclusion of IR in the models on the risk estimates for road traffic noise. Indeed, inclusion of IR tended to attenuate the risk estimates indicating that IR successfully captured the remaining variance, and further indicating that IR contributes to the risk of those outcomes. This tendency is not observable for railway and aircraft noise because IR is mostly driven by road traffic noise. Aircraft and railway noise are very intermittent noise sources and populations exposed to those noise sources are exposed to high IR at the same time. They therefore enjoy more periods of silence between noise events and their health is less affected by intermittency that acts as a protective factor. As a result, the risk estimate for exposure to L_{den}(Air) and L_{den}(Rail) aircraft and railway is increased. HR unadjusted and adjusted for IR for each

of the three noise sources road traffic, railway and aircraft are displayed in table 1.

Table 1: HR unadjusted and adjusted for IR for various CVD outcomes for each of the three noise sources road traffic, railway and aircraft

	HR L_{den}(Road) (95% CI)	**HR L_{den}(Road) (95% CI) adj for IR**	**HR L_{den}(Rail) (95% CI)**	**HR L_{den}(Rail) (95% CI) adj for IR**	**HR L_{den}(Air) (95% CI)**	**HR L_{den}(Air) (95% CI) adj for IR**
CVD	**1.025 (1.018-1.032)**	**1.023 (1.016-1.031)**	**1.005 (1.000-1.010)**	**1.006 (1.001-1.012)**	0.994 (0.985-1.002)	0.994 (0.985-1.002)
IHD	**1.023 (1.012-1.034)**	**1.021 (1.010-1.032)**	**1.013 (1.005-1.020)**	**1.013 (1.005-1.021)**	0.991 (0.979-1.003)	0.990 (0.978-1.003)
Stroke	1.011 (0.993-1.028)	1.008 (0.990-1.026)	0.995 (0.983-1.008)	0.996 (0.983-1.009)	1.013 (0.993-1.034)	1.012 (0.992-1.033)
BP	**1.054 (1.030-1.078)**	**1.052 (1.028-1.076)**	1.011 (0.995-1.027)	1.014 (0.998-1.032)	1.012 (0.985-1.040)	1.013 (0.985-1.041)
HF	**1.052 (1.027-1.077)**	**1.053 (1.028-1.079)**	0.997 (0.980-1.014)	1.003 (0.985-1.021)	**1.058 (1.028-1.088)**	**1.060 (1.030-1.091)**
HS	1.004 (0.969-1.041)	0.999 (0.963-1.037)	1.021 (0.996-1.046)	1.025 (0.998-1.052)	0.991 (0.952-1.032)	0.991 (0.952-1.032)
IS	**1.051 (1.002-1.103)**	**1.050 (1.000-1.103)**	0.989 (0.956-1.024)	1.000 (0.964-1.037)	**1.076 (1.020-1.135)**	**1.080 (1.023-1.139)**
MI	**1.041 (1.021-1.060)**	**1.038 (1.018-1.058)**	**1.020 (1.007-1.033)**	**1.021 (1.007-1.036)**	**1.027 (1.006-1.050)**	**1.027 (1.005-1.049)**

The road traffic component of intermittency has been modeled on the basis of the energy of noise events from car engines and tires (113). This implies that intermittency is based on the same noise frequencies as average noise levels. However, intermittency could arise from different frequencies due to differences in the vehicle fleet or by neither engine nor tire-related vehicle noises such as horns. Indeed, in many countries horn noise is a common phenomenon (13, 68, 69, 114) contributing substantially to noise levels (68). A frequency analysis of road traffic noise comparing two Vietnamese cities with a Japanese city has shown that a peak of frequencies related to horn noise could be clearly identified (13). Therefore, noise phenomenon like horns are more likely to induce changes in the magnitude of IR. In the light of this reasoning, future investigations will have to answer the

question whether in a different setting, a bell-shaped relationship between intermittency and risk for CVD still prevails.

However, a general decrease in noise levels as discussed in sections 10.2. and 10.3. could change the IR levels. In the case of road traffic noise, if the vehicle fleet changes toward more electric cars and fewer combustion motor engines, the background noise would be generated by the electric cars and noisier combustion cars would generate single events that would be easily noticed. This is most likely to happen on busy roads where current IR is low because of high traffic. A decrease in the noise levels, may then expose populations to harmful IR levels.

10.8. Confounding

In this section the issue of confounding of the associations between noise and health in the context of the results of this thesis will be discussed.

10.8.1. Confounding by air pollution

The results of section 8. demonstrate quite robust association between transportation noise in general and MI after adjustment for air pollution. Only the associations involving road traffic noise appeared to be confounded by NO_2 while no major change of the risk estimates occurred upon adjustment for $PM_{2.5}$. NO_2 is more related to traffic than $PM_{2.5}$ which can arise from different sources and this is most probably the reason why no confounding effect from this air pollutant could be observed. These results are in line with the extent of confounding reported in a meta-analysis (115). Categorical analyses revealed a larger extent of confounding by NO_2 in higher road traffic noise exposure categories for MI. This indicates that at NO_2 exposure has an impact on the relationship between road traffic noise and health above a certain threshold only. Further, this implies that a mutual confounding between road traffic noise and NO_2 occurs where noise and air pollution levels are more harmful for health. Consequently, it remains difficult to evaluate the contribution of health damaging effects from either pollutant for individuals in these high exposure categories. However, the results for railway and aircraft noise show independent health effects from air pollution. Using the exposure-response relationships from these sources in order to disentangle the effect from road traffic and air pollution is not advised, for railway and aircraft noise do not share the same characteristics and do not trigger the same health effects as road traffic noise.

Confounding of the association between road traffic noise and MI by NO_2 was observed at noise levels as low as 45 dB for which mean NO_2 concentrations was 21.21 $\mu g/m^3$ which is well below the WHO guideline value of 40$\mu g/m^3$ for long-term exposure. The extent of confounding increases in higher noise exposure categories. In the present thesis the estimated average NO_2 concentration was 33.05 $\mu g/m^3$ for road traffic noise levels above 65 dB. This raises the question about the extent of confounding by air pollution in other settings. Indeed, in Shanghai, China and Mumbai, India where long-term NO_2 levels range from 48 to 83 $\mu g/m^3$ (116) and 24 to 127 $\mu g/m^3$ (117), independent health

effects of noise and air pollution may be even more difficult to assess. Further, these ranges of hazardous air pollutant concentrations come along with very high noise exposure. One may raise the question about the relationship of these pollutants with health in this particular context. Are the health effects from transportation noise dwarfed by the massive air pollution levels? Does transportation noise contribute evenly with air pollution to higher risks? Is there a synergistic effect between air pollution and noise at high exposure levels? Although no synergistic effect between air pollutants and transportation noise was identified in this thesis, a publication has reported that such an effect can be observed on cognitive performance at lower exposure levels (118).

10.8.2. Confounding by neighbour noise

In the HRQOL study presented in section 4., annoyance to neighbour noise has been examined. The prevalence of neighbour noise and its impact on health have been rarely investigated. The LARES study (119), reports a prevalence of 12.4% of subjects strongly annoyed by neighbour noise, while in the present sample the proportion of subjects considerably and heavily annoyed was 10.2%. A longitudinal study conducted by Weinhold and colleagues using self-reported binary measurement for defining exposure and outcome has found that neighbour noise was significantly associated with lower quality of life and higher risk for cardiovascular diseases (120). Similar to the HRQOL study where neighbour noise annoyance was more strongly associated with HRQOL outcomes than road traffic noise annoyance, the study of Weinhold and colleagues shows stronger effect estimates for neighbour noise than for road traffic noise (120). From a methodological point of view, the HRQOL study is stronger, since validated scores were used for both exposure and outcome in place of binary variables.

Further the LARES study reports a dose-response increase of the risk for cardiovascular symptoms and medically diagnosed hypertension, arthrosis, allergy, depression and migraine with increasing neighbour noise annoyance category (119). These results support the association between neighbour noise annoyance and the von Zerssen indicator reported in the HRQOL study. Further, the HRQOL study shows that not only physical but mental health may be affected by neighbour noise annoyance. Neighbour noise annoyance is highly prevalent in the population and studies show an impact on health that should not be underestimated. Whether neighbour noise exposure is as high as annoyance to neighbour noise remains an open question, but this raises the question of confounding by neighbour noise. Indeed, neighbour noise is expected to co-occur along with road traffic noise in cities where population density is high. The stratified analysis in section 5. show that in general the risk for CVD due to road traffic noise exposure is higher in urban settings where neighbour noise is more prevalent. Given that a clear impact of neighbour noise on HRQOL has been demonstrated, it may be likely that other health effects could be observed. Being correlated with both road traffic noise exposure and the outcomes, neighbour noise fulfills the definition of a confounder and should deserve particular attention. For this reason, future studies should seek to elucidate the importance of neighbour noise for health.

10.9. Strengths and limitations

In the study presented in section 4., the relationship between road traffic noise exposure, annoyance, sleep disturbance and HRQOL was examined using advanced statistical methods. This study provides prevalence values of the –often underreported noise annoyance types- annoyance to industrial and neighbour noise as well as associations between these annoyance types and HRQOL indicators. SEM has rarely been conducted in the field of environmental noise and robust baseline and follow-up values have been reported in separate models, thus strengthening the consistency of the present results. This study has used the von Zerssen indicator for the first time in the field of environmental noise epidemiology.

Limitation related to the low-response rate of 37% which could have introduced a bias could be ruled out by comparison of the noise exposure levels of the non-respondent vs respondents. However a bias related to the use of self-reported scales for annoyance and HRQOL could not be excluded. All associations were adjusted with the set of variables deemed necessary in this kind of study. However, the presence of some residual confounding cannot be ruled out. Indeed, noise sensitivity, which is an important parameter to consider when dealing with annoyance and mental health (121), was not available.

Studies reported in sections 5., 6., and 8. have been conducted on a cohort of about 4.41 mio observations over 8 years of follow-up without selection bias. The size of the study population ensures robust estimates through high statistical power. The high definition model for all three transportation sources allowed us to perform a cutting edge individual exposure assessment taking into account the place and floor of residency. For both air pollutant models, high definition data and advanced statistical methods were used in order to guarantee a high level of accuracy of the predicted exposure. The impact of all three noise sources concurrently was assessed which has rarely been done in previous large noise impact studies. Further, the IR metric allows us to refine noise exposure and this metric is expected to open new research avenues and will affects noise policy and regulations. The evaluation of the effects of transportation noise exposure in different time windows is also a novelty brought by this thesis.

All models were adjusted for socioeconomic status and other demographic variables, but the SNC did not contain information on smoking status or lifestyle. Therefore, residual confounding related to lifestyle cannot be ruled out. However, an uneven distribution of smokers or lifestyle factors in different noise categories seems not plausible and bias from this side is therefore not expected. Similarly, a bias on the mortality records would have occurred only if mortality was related to the exposure, which is very unlikely. Mortality data as used in this study cannot be considered incidence data because time to death after the disease has been diagnosed can vary greatly. However, a meta-analysis of studies on MI and road traffic ruled out systematic differences between mortality and incidence studies (64).

A great effort has been invested in acquiring the input data for the noise and air pollution models. Each model contains errors and uncertainties stemming from missing information and modelling assumptions. However, these errors are more likely to be non-differential, thus resulting in an underestimation of the true risks.

The majority of studies (as all the noise studies from this thesis) in the field of environmental noise epidemiology suffer potential exposure misclassification. Indeed, missing historical data, missing information about residential history and location of bedroom, and personal behavior (such as window closing behavior and other coping strategies), no information about exposure modifiers such as façade insulation and location of the bedroom could have biased the estimates.

10.10. Public health relevance

This thesis fills many knowledge gaps in environmental noise epidemiology. While confirming the associations between transportation noise and CVD in international studies, new insights about the impacts of noise characteristics, time of exposure and confounding by air pollution have been revealed. This thesis reports valuable knowledge for public health authorities, urban planners as well as physicians and could potentially lead to a paradigm change in protection from environmental noise pollution. Moreover, this work can be used to inform the population about the deleterious effects of transportation noise exposure.

The attributable risk gives the proportion of cases among the exposed attributable to the exposure. In the study population analyzed in the framework of the present thesis, the attributable risk for CVD due to road traffic is 2.5%. This means that over all the individuals that deceased from CVD between the years 2000 to 2008, 2.5% of them could have been avoided if they were not exposed to road traffic noise. Further the sum of attributable risk to all transportation noise sources translates into 4146 cases that could have been avoided by eliminating the exposure.

Table 2: Table of attributable risk (in %) for selected outcomes due to road traffic, railway, and aircraft noise.

	Road % (95% CI)	Rail % (95% CI)	Aircraft % (95% CI)	n avoidable cases from all noise sources
CVD	2.4 1.8-3.1)	0.5 (0.0-1.0)	-	4146
IHD	2.2 (1.2-3.4)	1.1 (0.5-2.0)	-	1989
MI	3.6 (2.1-5.5)	2.0 (0.7-3.1)	2.6 (0.6-4.3)	1584
Heart failure	4.8	-	5.3	1247

	(0.2-9.0)		(0.2-11.2)	

While a total elimination of the risk attributable to transportation noise is unrealistic for practical reasons, the measures discussed in sections 10.2. and 10.3. of this work will help to lessen the impact of transportation noise. The results of this thesis support noise mitigation strategies adopted in Switzerland. Furthermore, the higher risk observed during nighttime pleads for the introduction of low speed zones (Tempo 30, Figure 5) from 22h to 06h as in many German cities like Hamburg and Frankfurt.

Figure 5: German street sign limiting the speed during nighttime

This thesis also show that air pollution and noise have independent effect and a reduction of both sources can easily achieved with modest means. Indeed, since years, due to improved infrastructure, the number of cyclists has been increasing in Swiss cities. The use of bicycles helps alleviate both air pollution levels and noise emissions while also being a protective factor for CVD prevention (122).

Compared to major risk factors for CVD, transportation noise exposure seems negligible. However, there are substantial differences between classical risk factors for CVD such as lack of physical activity and smoking behavior, and environmental risk factors like transportation noise and air pollution. Various risk factors for CVD mortality are displayed in table 3.

Table 3: Risk factors for CVD mortality and their estimates

Risk factor	**Risk increase in % (95% CI)**
Smoking >10 cigarettes/day (123)	80 (61-111)
Lack of physical activity (124)	90 (60-120)
Heavy alcohol consumption (125)	105 (9-286)
High salt intake (126)	14 (0-32)
Air pollution (100 µg/m^3) (127)	6 (5-7)
Road traffic noise (10 dB) (65)	8 (4-13)

While smoking, lack of physical activity, high alcohol and salt consumption belong to individual lifestyle and can in some extent be controlled, air pollution and transportation noise are much more

difficult to control at the individual scale. The high prevalence of annoyance to transportation noise reported in this thesis expresses this feeling of lack of perceived control.

Further, while the prevalence of smokers in Switzerland has been reported to be around 25% (128), this thesis reports that >89% of the Swiss population is exposed to noise levels above 45 dB. In terms of public health, lower risk and high prevalence raises the importance of transportation noise compared to the high risk and lower prevalence of smoking. 25% of the population has made the choice of smoking, while the 89% of the population exposed to road traffic noise has not decided to be voluntarily exposed. Therefore a sound decision would be to allocate more resources in order to protect the whole population from risk factors that affects almost everyone.

11. Outlook

Exposure misclassification has been shown to have a non-negligible impact on the risk estimates (129-132). In order to assess exposure misclassification, the SIAS (SiRENE Innen-Aussenstudie) study has been launched at the Swiss TPH. The aim of this project is to evaluate the extent of exposure misclassification and assess the impact of such misclassification on the association with health outcomes by considering additional factors such as the age of building, date of renovation, and the presence of double-glazed windows.

The Holy Grail in terms of environmental noise research is the true exposure. The technology to achieve an assessment of the true exposure is available. Personal exposure tools have been developed and can be loaded in smartphone apps, thus allowing for a real-time evaluation of the noise exposure and can be deployed for small studies. For large scale studies however, assessing the true exposure remains difficult.

Within the framework of the SiRENE project, noise modelling was conducted for the years 1991, 2001 and 2011. The analyses in the present thesis were conducted using the SNC data from 2000 to 2008 and the noise model for the year 2001. The SNC contains a variable informing about the length of stay of the subject in the same community. The SNC team is expected to extend the observations to years subsequent to 2008. A longer duration of observation of the cohort, time-varying noise exposure and the use of the "length of stay" variable will allow comparison of the health impact of longer term noise exposure as well as the effect of a change in noise exposure due to address change.

12. References for sections 1. to 3. and 9. to 11.

1. WHO. Fact sheet: Cardiovascular diseases (CVDs) Geneva: World Health Organization; 2016 [updated 16.02.2017. Available from: http://www.who.int/mediacentre/factsheets/fs317/en/.
2. Ezzati M, Obermeyer Z, Tzoulaki I, Mayosi BM, Elliott P, Leon DA. Contributions of risk factors and medical care to cardiovascular mortality trends. Nature reviews Cardiology. 2015;12(9):508-30.
3. Di Cesare M, Khang Y-H, Asaria P, Blakely T, Cowan MJ, Farzadfar F, et al. Inequalities in non-communicable diseases and effective responses. The Lancet. 2013;381(9866):585-97.
4. WEF. The Global Economic Burden of Non-communicable Diseases. Geneva: World Economic Forum/ Harvard school of Public Health; 2011.
5. WHO. The global burden of disease 2004 update. Geneva, Switzerland: World Health Organization; 2008.
6. OFS. Maladies cardiovasculaires Neuchatel2017 [Available from: https://www.bfs.admin.ch/bfs/fr/home/statistiques/sante/etat-sante/maladies/cardiovasculaires.html.
7. Wieser S, Tomonaga Y, Riguzzi M, Fischer B, Telser H, Pletscher M, et al. Die Kosten der nichtübertragbaren Krankheiten in der Schweiz. Bundesamt für Gesundheit BAG, Bern. 2014.
8. Tzoulaki I, Elliott P, Kontis V, Ezzati M. Worldwide Exposures to Cardiovascular Risk Factors and Associated Health Effects. Circulation. 2016;133(23):2314-33.
9. Ezzati M, Hoorn SV, Rodgers A, Lopez AD, Mathers CD, Murray CJ. Estimates of global and regional potential health gains from reducing multiple major risk factors. Lancet (London, England). 2003;362(9380):271-80.
10. Babisch W, Dutilleux G, Paviotti M, Backman A, Gergely B, McManus B, et al. Good practice guide on noise exposure and potential health effects. European Environmental Agency Technical report2010.
11. WHO. Night noise guidelines for Europe. Copenhagen: World Health Organization; 2009.
12. WHO. Urban populaion growth. 2014.
13. Phan HYT, Yano T, Sato T, Nishimura T. Characteristics of road traffic noise in Hanoi and Ho Chi Minh City, Vietnam. Applied Acoustics. 2010;71(5):479-85.
14. Trombetta Zannin PH, Bunn F. Noise annoyance through railway traffic - a case study. Journal of Environmental Health Science and Engineering. 2014;12:14-.
15. Banerjee D, Chakraborty SK, Bhattacharyya S, Gangopadhyay A. Evaluation and analysis of road traffic noise in Asansol: an industrial town of eastern India. International journal of environmental research and public health. 2008;5(3):165-71.
16. Onuu MU. ROAD TRAFFIC NOISE IN NIGERIA: MEASUREMENTS, ANALYSIS AND EVALUATION OF NUISANCE. Journal of Sound and Vibration. 2000;233(3):391-405.
17. Xie H, Li H, Kang J. The characteristics and control strategies of aircraft noise in China. Applied Acoustics. 2014;84:47-57.
18. HE Z-h, CHEN C, ZHU D-m. Noise pollution assessment of Baiyun International Airport, Guangzhou. The Administration and Technique of Environmental Monitoring. 2004;1:009.
19. EEA. Noise in Europe 2014. Copenhagen: European Environmental Agency; 2014.
20. OFEV. Etat de l'exposition au bruit en Suisse Berne2015 [Available from: https://www.bafu.admin.ch/bafu/fr/home/themes/bruit/info-specialistes/exposition-au-bruit/etat-de-l_exposition-au-bruit-en-suisse.html.
21. Munzel T, Sorensen M, Gori T, Schmidt FP, Rao X, Brook FR, et al. Environmental stressors and cardio-metabolic disease: part II-mechanistic insights. European heart journal. 2016.

22. Stansfeld SA, Matheson MP. Noise pollution: non-auditory effects on health. British Medical Bulletin. 2003;68(1):243-57.
23. Bundesamt für Umwelt (BAFU) BfSB. Umwelt Report 2007. Neuchâtel and Bern2007.
24. Guski R. Personal and social variables as co-determinants of noise annoyance. Noise Health. 1999;1(3):45-56.
25. Fields JM. Effect of personal and situational variables on noise annoyance in residential areas. The Journal of the Acoustical Society of America. 1993;93(5):2753-63.
26. Stallen PJ. A theoretical framework for environmental noise annoyance. Noise and Health. 1999;1(3):69-79.
27. Miedema HM, Oudshoorn CG. Annoyance from transportation noise: relationships with exposure metrics DNL and DENL and their confidence intervals. Environmental Health Perspectives. 2001;109(4):409-16.
28. Recio A, Linares C, Banegas JR, Diaz J. Road traffic noise effects on cardiovascular, respiratory, and metabolic health: An integrative model of biological mechanisms. Environ Res. 2016;146:359-70.
29. Ising H, Kruppa B. Health effects caused by noise: evidence in the literature from the past 25 years. Noise Health. 2004;6(22):5-13.
30. Basner M, Glatz C, Griefahn B, Penzel T, Samel A. Aircraft noise: effects on macro- and microstructure of sleep. Sleep medicine. 2008;9(4):382-7.
31. Basner M, Griefahn B, Berg M. Aircraft noise effects on sleep: mechanisms, mitigation and research needs. Noise Health. 2010;12(47):95-109.
32. Griefahn B, Bröde P, Marks A, Basner M. Autonomic Arousals Related to Traffic Noise during Sleep. Sleep. 2008;31(4):569-77.
33. Mullington JM, Haack M, Toth M, Serrador J, Meier-Ewert H. Cardiovascular, Inflammatory and Metabolic Consequences of Sleep Deprivation. Progress in Cardiovascular Diseases. 2009;51(4):294-302.
34. Cappuccio FP, Cooper D, D'Elia L, Strazzullo P, Miller MA. Sleep duration predicts cardiovascular outcomes: a systematic review and meta-analysis of prospective studies. European heart journal. 2011;32(12):1484-92.
35. Babisch W, Van Kamp I. Exposure-response relationship of the association between aircraft noise and the risk of hypertension. Noise \& Health. 2009;11(44).
36. Van Kempen E, Babisch W. The quantitative relationship between road traffic noise and hypertension: a meta-analysis. Journal of hypertension. 2012;30(6):1075-86.
37. Huss A, Spoerri A, Egger M, Röösli M, Group SNCS, others. Aircraft noise, air pollution, and mortality from myocardial infarction. Epidemiology. 2010;21(6):829-36.
38. Hansell AL, Blangiardo M, Fortunato L, Floud S, de Hoogh K, Fecht D, et al. Aircraft noise and cardiovascular disease near Heathrow airport in London: small area study. BMJ: British Medical Journal. 2013;347.
39. Basner M, Müller U, Elmenhorst E-M. Single and Combined Effects of Air, Road, and Rail Traffic Noise on Sleep and Recuperation. Sleep. 2011;34(1):11-23.
40. Basner M, Samel A, Isermann U. Aircraft noise effects on sleep: application of the results of a large polysomnographic field study. The Journal of the Acoustical Society of America. 2006;119(5 Pt 1):2772-84.
41. Brink M, Lercher P, Eisenmann A, Schierz C. Influence of slope of rise and event order of aircraft noise events on high resolution actimetry parameters. Somnologie - Schlafforschung und Schlafmedizin. 2008;12(2):118-28.
42. Dratva J, Zemp E, Felber Dietrich D, Bridevaux PO, Rochat T, Schindler C, et al. Impact of road traffic noise annoyance on health-related quality of life: results from a population-based study. Quality of life research : an international journal of quality of life aspects of treatment, care and rehabilitation. 2010;19(1):37-46.
43. Halonen JI, Hansell AL, Gulliver J, Morley D, Blangiardo M, Fecht D, et al. Road traffic noise is associated with increased cardiovascular morbidity and mortality and all-cause mortality in London. European heart journal. 2015;36(39):2653-61.
44. Jarup L, Babisch W, Houthuijs D, Pershagen G, Katsouyanni K, Cadum E, et al. Hypertension and Exposure to Noise Near Airports: the HYENA Study. Environmental Health Perspectives. 2008;116(3):329-33.

45. Beelen R, Hoek G, Houthuijs D, van den Brandt PA, Goldbohm RA, Fischer P, et al. The joint association of air pollution and noise from road traffic with cardiovascular mortality in a cohort study. Occup Environ Med. 2009;66(4):243-50.
46. Gan WQ, Davies HW, Koehoorn M, Brauer M. Association of long-term exposure to community noise and traffic-related air pollution with coronary heart disease mortality. Am J Epidemiol. 2012;175(9):898-906.
47. Huss A, Spoerri A, Egger M, Roosli M. Aircraft noise, air pollution, and mortality from myocardial infarction. Epidemiology. 2010;21(6):829-36.
48. Selander J, Nilsson ME, Bluhm G, Rosenlund M, Lindqvist M, Nise G, et al. Long-term exposure to road traffic noise and myocardial infarction. Epidemiology. 2009;20(2):272-9.
49. Sorensen M, Luhdorf P, Ketzel M, Andersen ZJ, Tjonneland A, Overvad K, et al. Combined effects of road traffic noise and ambient air pollution in relation to risk for stroke? Environ Res. 2014;133:49-55.
50. Tonne C, Halonen JI, Beevers SD, Dajnak D, Gulliver J, Kelly FJ, et al. Long-term traffic air and noise pollution in relation to mortality and hospital readmission among myocardial infarction survivors. Int J Hyg Environ Health. 2016;219(1):72-8.
51. Eurostat. European Health Interview Survey (EHIS wave 2). Luxembourg; 2013.
52. Von Zerssen D, Petermann F. B-LR—Beschwerden-Liste—Revidierte Fassung. Hogrefe: Göttingen, Germany. 2011.
53. Ware JE, Kosinski M, Dewey JE, Gandek B. SF-36 health survey: manual and interpretation guide: Quality Metric Inc.; 2000.
54. FSO. Swiss Health Survey 2007. Neuchatel; 2012.
55. FOEN. NO2 Ambient Concentrations in Switzerland. Modeling Results for 2005, 2010 and 2020. Berne Federal Office for the Environment; 2011.
56. Kloog I, Chudnovsky AA, Just AC, Nordio F, Koutrakis P, Coull BA, et al. A new hybrid spatio-temporal model for estimating daily multi-year PM2.5 concentrations across northeastern USA using high resolution aerosol optical depth data. Atmospheric Environment. 2014;95:581-90.
57. Foraster M, Eze IC, Vienneau D, Brink M, Cajochen C, Caviezel S, et al. Long-term transportation noise annoyance is associated with subsequent lower levels of physical activity. Environment International. 2016;91:341-9.
58. Roswall N, Høgh V, Envold-Bidstrup P, Raaschou-Nielsen O, Ketzel M, Overvad K, et al. Residential Exposure to Traffic Noise and Health-Related Quality of Life—A Population-Based Study. PLoS ONE. 2015;10(3):e0120199.
59. Fyhri A, Aasvang GM. Noise, sleep and poor health: Modeling the relationship between road traffic noise and cardiovascular problems. Science of The Total Environment. 2010;408(21):4935-42.
60. Fyhri A, Klæboe R. Road traffic noise, sensitivity, annoyance and self-reported health—A structural equation model exercise. Environment International. 2009;35(1):91-7.
61. Frei P, Mohler E, Roosli M. Effect of nocturnal road traffic noise exposure and annoyance on objective and subjective sleep quality. Int J Hyg Environ Health. 2014;217(2-3):188-95.
62. Babisch W. Updated exposure-response relationship between road traffic noise and coronary heart diseases: a meta-analysis. Noise Health. 2014;16(68):1-9.
63. van Kempen E, Babisch W. The quantitative relationship between road traffic noise and hypertension: a meta-analysis. J Hypertens. 2012;30(6):1075-86.
64. Vienneau D, Schindler C, Perez L, Probst-Hensch N, Roosli M. The relationship between transportation noise exposure and ischemic heart disease: a meta-analysis. Environ Res. 2015;138:372-80.
65. Babisch W. Updated exposure-response relationship between road traffic noise and coronary heart diseases: A meta-analysis. Noise and Health. 2014;16(68):1-9.
66. Heutschi K, Bühlmann E, Oertli J. Options for reducing noise from roads and railway lines. Transportation Research Part A: Policy and Practice. 2016;94:308-22.
67. Zürich S. Tempo 30 Zürich2017 [Available from: https://www.stadt-zuerich.ch/pd/de/index/dav/themen_projekte/tempo_30.html.

68. Ali SA, Tamura A. Road traffic noise levels, restrictions and annoyance in Greater Cairo, Egypt. Applied Acoustics. 2003;64(8):815-23.
69. Ma G, Tian Y, Ju T, Ren Z. Assessment of Traffic Noise Pollution from 1989 to 2003 in Lanzhou City. Environmental Monitoring and Assessment. 2006;123(1):413-30.
70. Verheijen E, Jabben J. Effect of electric cars on traffic noise and safety. RIVM letter report. 2010;680300009:2010.
71. Delbosc A, Currie G. Causes of Youth Licensing Decline: A Synthesis of Evidence. Transport Reviews. 2013;33(3):271-90.
72. Polzin SE, Chu X, Godfrey J. The impact of millennials' travel behavior on future personal vehicle travel. Energy Strategy Reviews. 2014;5:59-65.
73. Lyons G. Transport's digital age transition. Journal of Transport and Land Use. 2015;8(2):1-19.
74. Klein NJ, Smart MJ. Millennials and car ownership: Less money, fewer cars. Transport Policy. 2017;53:20-9.
75. Eliasson J, Hultkrantz L, Nerhagen L, Rosqvist LS. The Stockholm congestion–charging trial 2006: Overview of effects. Transportation Research Part A: Policy and Practice. 2009;43(3):240-50.
76. Leape J. The London congestion charge. The Journal of Economic Perspectives. 2006;20(4):157-76.
77. Jud P. Städtevergleich Mobilität-Vergleichende Betrachtung der Städte Basel, Bern, Luzern, St.Gallen, Winterthur und Zürich. Kloten; 2012.
78. DETA. Les transports genevois en chiffre. In: DÉPARTEMENT DE L'ENVIRONNEMENT DTEDLA, editor. Geneva2014.
79. Ohrstrom E. Longitudinal surveys on effects of changes in road traffic noise-annoyance, activity disturbances, and psycho-social well-being. The Journal of the Acoustical Society of America. 2004;115(2):719-29.
80. Gakenheimer R. Urban mobility in the developing world. Transportation Research Part A: Policy and Practice. 1999;33(7–8):671-89.
81. McMichael AJ. The urban environment and health in a world of increasing globalization: issues for developing countries. Bulletin of the World Health Organization. 2000;78:1117-26.
82. Seidler A, Wagner M, Schubert M, Dröge P, Römer K, Pons-Kühnemann J, et al. Aircraft, road and railway traffic noise as risk factors for heart failure and hypertensive heart disease—A case-control study based on secondary data. International Journal of Hygiene and Environmental Health. 2016;219(8):749-58.
83. Tassi P, Saremi M, Schimchowitsch S, Eschenlauer A, Rohmer O, Muzet A. Cardiovascular responses to railway noise during sleep in young and middle-aged adults. European journal of applied physiology. 2010;108(4):671-80.
84. SBB. Noise: Swiss railways; 2017 [Available from: https://www.sbb.ch/en/group/the-company/der-umwelt-verpflichtet/laerm-erschuetterungen/laerm.html.
85. China NBoS. 2017 [Available from: http://data.stats.gov.cn/.
86. Huss A, Spoerri A, Egger M, Röösli M, Group SNCS. Aircraft noise, air pollution, and mortality from myocardial infarction. Epidemiology. 2010;21(6):829-36.
87. BFS. STAT-TAB - Interaktive Tabellen. In: (BFS) BfS, editor. Neuchâtel2015.
88. Zufferey J. Pourquoi les migrants vivent-ils plus longtemps ?: analyse des différentiels de mortalité des populations suisse et étrangères 1990-2008 Geneva: University of Geneva; 2014.
89. J. F, L. B, M. S. Night flight restrictions and airline responses at major European airports. Delft: CE Delft; 2012.
90. Dowling A, Greitzer E. The silent aircraft initiative—Overview. AIAA Paper No AIAA-2007-0452. 2007.
91. D. R. Airbus A380 noise measurments. Civil Aviation Authority; 2015.
92. Flightstats. 2017 [Available from: http://www.flightstats.com/go/Home/home.do.
93. IATA. IATA Forecasts Passenger Demand to Double Over 20 Years 2016 [Available from: http://www.iata.org/pressroom/pr/Pages/2016-10-18-02.aspx.

94. Muller JE. Circadian variation and triggering of acute coronary events. American heart journal. 1999;137(4):S1-S8.
95. Ørn S, Dickstein K. How do heart failure patients die? European Heart Journal Supplements. 2002;4(suppl_D):D59-D65.
96. Arboix A, Alio J. Acute cardioembolic cerebral infarction: answers to clinical questions. Current cardiology reviews. 2012;8(1):54-67.
97. Ferro JM. Cardioembolic stroke: an update. The Lancet Neurology. 2003;2(3):177-88.
98. Halonen JI, Hansell AL, Gulliver J, Morley D, Blangiardo M, Fecht D, et al. Road traffic noise is associated with increased cardiovascular morbidity and mortality and all-cause mortality in London. European heart journal. 2015;36(39):2653-61.
99. Sørensen M, Hvidberg M, Andersen ZJ, Nordsborg RB, Lillelund KG, Jakobsen J, et al. Road traffic noise and stroke: a prospective cohort study. European heart journal. 2011;32(6):737-44.
100. Ouis D. Annoyance Caused by Exposure to Road Traffic Noise: An Update. Noise Health. 2002;4(15):69-79.
101. Marks A, Griefahn B, Basner M. Event-related awakenings caused by nocturnal transportation noise. Noise Control Engineering Journal. 2008;56(1):52-62.
102. Babisch W, Beule B, Schust M, Kersten N, Ising H. Traffic noise and risk of myocardial infarction. Epidemiology. 2005;16(1):33-40.
103. Sorensen M, Andersen ZJ, Nordsborg RB, Jensen SS, Lillelund KG, Beelen R, et al. Road traffic noise and incident myocardial infarction: a prospective cohort study. PLoS One. 2012;7(6):e39283.
104. Barregard L, Bonde E, Ohrstrom E. Risk of hypertension from exposure to road traffic noise in a population-based sample. Occup Environ Med. 2009;66(6):410-5.
105. Bluhm GL, Berglind N, Nordling E, Rosenlund M. Road traffic noise and hypertension. Occupational and Environmental Medicine. 2007;64(2):122-6.
106. Eriksson C, Bluhm G, Hilding A, Ostenson CG, Pershagen G. Aircraft noise and incidence of hypertension--gender specific effects. Environ Res. 2010;110(8):764-72.
107. Christensen JS, Raaschou-Nielsen O, Tjonneland A, Nordsborg RB, Jensen SS, Sorensen TI, et al. Long-term exposure to residential traffic noise and changes in body weight and waist circumference: A cohort study. Environ Res. 2015;143(Pt A):154-61.
108. Eriksson C, Hilding A, Pyko A, Bluhm G, Pershagen G, Ostenson CG. Long-term aircraft noise exposure and body mass index, waist circumference, and type 2 diabetes: a prospective study. Environ Health Perspect. 2014;122(7):687-94.
109. Pyko A, Eriksson C, Oftedal B, Hilding A, Ostenson CG, Krog NH, et al. Exposure to traffic noise and markers of obesity. Occup Environ Med. 2015;72(8):594-601.
110. Sorensen M, Andersen ZJ, Nordsborg RB, Becker T, Tjonneland A, Overvad K, et al. Long-term exposure to road traffic noise and incident diabetes: a cohort study. Environ Health Perspect. 2013;121(2):217-22.
111. Willich SN, Wegscheider K, Stallmann M, Keil T. Noise burden and the risk of myocardial infarction. Eur Heart J. 2006;27(3):276-82.
112. Willich SN, Wegscheider K, Stallmann M, Keil T. Noise burden and the risk of myocardial infarction. European heart journal. 2006;27(3):276-82.
113. Wunderli JM, Pieren R, Habermacher M, Vienneau D, Cajochen C, Probst-Hensch N, et al. Intermittency ratio: A metric reflecting short-term temporal variations of transportation noise exposure. Journal of Exposure Science & Environmental Epidemiology. 2016;26(6):575-85.
114. Goswami S. Road Traffic Noise: A Case Study of Balasore Town, Orissa, India. International Journal of Environmental Research. 2009;3(2):309-18.
115. Tetreault LF, Perron S, Smargiassi A. Cardiovascular health, traffic-related air pollution and noise: are associations mutually confounded? A systematic review. Int J Public Health. 2013;58(5):649-66.
116. Mi YH, Norbäck D, Tao J, Mi YL, Ferm M. Current asthma and respiratory symptoms among pupils in Shanghai, China: influence of building ventilation, nitrogen dioxide, ozone, and formaldehyde in classrooms. Indoor Air. 2006;16(6):454-64.

117. Patankar AM, Trivedi PL. Monetary burden of health impacts of air pollution in Mumbai, India: Implications for public health policy. Public Health. 2011;125(3):157-64.
118. Tzivian L, Dlugaj M, Winkler A, Weinmayr G, Hennig F, Fuks KB, et al. Long-Term Air Pollution and Traffic Noise Exposures and Mild Cognitive Impairment in Older Adults: A Cross-Sectional Analysis of the Heinz Nixdorf Recall Study. Environ Health Perspect. 2016;124(9):1361-8.
119. Niemann H, Bonnefoy X, Braubach M, Hecht K, Maschke C, Rodrigues C, et al. Noise-induced annoyance and morbidity results from the pan-European LARES study. Noise Health. 2006;8(31):63-79.
120. Weinhold D. Sick of noise: the health effects of loud neighbours and urban din. Grantham Research Institute on Climate Change and the Environment; 2015.
121. Stansfeld SA, Shipley M. Noise sensitivity and future risk of illness and mortality. The Science of the total environment. 2015;520:114-9.
122. Oja P, Titze S, Bauman A, de Geus B, Krenn P, Reger-Nash B, et al. Health benefits of cycling: a systematic review. Scandinavian Journal of Medicine & Science in Sports. 2011;21(4):496-509.
123. Jacobs DR, Jr, Adachi H, Mulder I, et al. Cigarette smoking and mortality risk: Twenty-five–year follow-up of the seven countries study. Archives of Internal Medicine. 1999;159(7):733-40.
124. Berlin JA, Colditz GA. A META-ANALYSIS OF PHYSICAL ACTIVITY IN THE PREVENTION OF CORONARY HEART DISEASE. American Journal of Epidemiology. 1990;132(4):612-28.
125. Malyutina S, Bobak M, Kurilovitch S, Gafarov V, Simonova G, Nikitin Y, et al. Relation between heavy and binge drinking and all-cause and cardiovascular mortality in Novosibirsk, Russia: a prospective cohort study. The Lancet. 2002;360(9344):1448-54.
126. Strazzullo P, D'Elia L, Kandala N-B, Cappuccio FP. Salt intake, stroke, and cardiovascular disease: meta-analysis of prospective studies. BMJ. 2009;339.
127. Schwartz J. Air Pollution and Daily Mortality: A Review and Meta Analysis. Environmental Research. 1994;64(1):36-52.
128. BAG. Tabak Bern: Bundesamt für Gesundheit; 2016 [Available from: https://www.bag.admin.ch/bag/de/home/themen/mensch-gesundheit/sucht/tabak.html.
129. Babisch W, Ising H, Gallacher JE, Sweetnam PM, Elwood PC. Traffic noise and cardiovascular risk: the Caerphilly and Speedwell studies, third phase--10-year follow up. Archives of environmental health. 1999;54(3):210-6.
130. Babisch W, Swart W, Houthuijs D, Selander J, Bluhm G, Pershagen G, et al. Exposure modifiers of the relationships of transportation noise with high blood pressure and noise annoyance. The Journal of the Acoustical Society of America. 2012;132(6):3788-808.
131. Foraster M, Künzli N, Aguilera I, Rivera M, Agis D, Vila J, et al. High blood pressure and long-term exposure to indoor noise and air pollution from road traffic. Environmental health perspectives. 2014;122(11):1193.
132. Lercher P, Widmann U, Kofler W, editors. Transportation noise and blood pressure: the importance of modifying factors. Proceedings of the 29th International Congress and Exhibition on Noise Control Engineering (Cassereau D, ed) InterNoise; 2000.

13. Appendix A

This paper does not belong to this thesis. It describes the data used in this thesis

Reconstruction of historical noise exposure data for environmental epidemiology in Switzerland within the SiRENE project

Ioannis Karipidis[1], Danielle Vienneau[2,3], Manuel Habermacher[4], Micha Köpfli[4], Mark Brink[5],

Nicole Probst-Hensch[2,3], Martin Röösli[2,3], and Jean-Marc Wunderli[1]

[1] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Dubendorf, Switzerland.

[2] Swiss Tropical and Public Health Institute, Basel, Switzerland

[3] University of Basel, Basel, Switzerland

[4] N-sphere AG, Zürich, Switzerland

[5] Federal Office for the Environment, Bern, Switzerland

Research Article **Open Access**

Ioannis Karipidis, Danielle Vienneau, Manuel Habermacher, Micha Köpfli, Mark Brink, Nicole Probst-Hensch, Martin Röösli, and Jean-Marc Wunderli

Reconstruction of historical noise exposure data for environmental epidemiology in Switzerland within the SiRENE project

Abstract: In 2014 the three-year interdisciplinary study SiRENE (Short and Long Term Effects of Traffic Noise Exposure) was launched in Switzerland. The goal of SiRENE is to investigate acute, short- and long-term effects of road, railway and aircraft noise exposure on annoyance, sleep disturbances and cardio-metabolic risk.

The study is based on a detailed Swiss-wide assessment of transportation noise exposure, including diurnal distributions. The exposure analysis comprises current as well as historical exposure calculations for up to 20 years in the past.

We present the major challenges of compiling sufficient data to support a Swiss-wide model for all buildings and including all transport infrastructure as a basis for the subsequent SiRENE sound exposure analysis for the years 2011, 2001 and 1991. The task is particularly challenging for the early years due to poor data quality and/or lack of availability. We address the integration of geo-referenced input datasets from various sources and time periods, the assignment of traffic noise exposure from façade points to dwelling units, as well as the processing of traffic information and statistics. Preliminary results of the noise exposure calculations are presented.

Ioannis Karipidis: Empa, Swiss Federal Laboratories for Materials Science and Technology, Dübendorf, Switzerland
Danielle Vienneau: Swiss Tropical and Public Health Institute, Basel, Switzerland,
and University of Basel, Basel, Switzerland
Manuel Habermacher, Micha Köpfli: n-Sphere AG, Zürich, Switzerland
Mark Brink: Federal Office for the Environment, Bern, Switzerland
Nicole Probst-Hensch, Martin Röösli: Swiss Tropical and Public Health Institute, Basel, Switzerland,
and University of Basel, Basel, Switzerland
Jean-Marc Wunderli: Empa, Swiss Federal Laboratories for Materials Science and Technology, Dübendorf, Switzerland

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1 Introduction

Transportation noise, largely from road, railway and aircraft traffic, is one of the most widespread sources of environmental stress and discomfort in daily life. Viewed mainly as an urban problem, it impacts a large proportion of the population in Europe. Based on the EU strategic noise maps, covering approximately 20% of the total population in 2010, an estimated 50% of residents in agglomerations with >250,000 inhabitants are exposed to road traffic noise of ≥55 dB Lden [1]. This led European legislation to adopt the European Noise Directive, END [2], which is currently being implemented by the member states [3]. END aims at monitoring and mapping environmental noise, to inform the public and to initiate noise mitigation and abatement actions.

Little is known, however, about how acute and short-term noise effects translate into long-term health consequences. In particular, it is unclear which acoustical characteristics of noise from different transportation sources are most detrimental for human health and wellbeing. To date, research and noise policy has mainly focused on annoyance and metrics capturing average exposure (e.g. Lden, Lnight) [4–6]. Despite certain limitations, for example in communicating noise exposure in a comprehensible way to the public [7], such energy-based exposure measures seem to predict sufficiently well annoyance or disturbance [8]. However their application to forecast the impact of noise on sleep has not met with much success [9–13]. Whereas the probabilities of event-related awakenings and cardiovascular arousals clearly increase with the maximum sound pressure level of noise events (Lmax,

SEL) [4, 14, 15], average noise metrics usually fail to predict sleep disturbances sufficiently [16–18]. Thus, depending on whether the noise source is intermittent (e.g., passing flights or trains) or continuous (e.g., road traffic from a highway), the effects of noise on sleep might be better predicted by the number of noise events and their characteristics [19]. There is, for example, evidence that railway noise at comparable average levels across a night elicits stronger reactions in the sleeping individuals compared to noise from other transportation modes [20]. It has been proposed that the distribution of maximum sound pressure levels and the relatively short rise time of railway noise events are primarily responsible for the increased reaction probability, as compared for example to road traffic noise events. For these reasons, it is essential not only to evaluate the average exposure levels for possible long-term health effects but also investigate more detailed exposure assessment characteristics, in particular the *degree of intermittence* and the *distribution of maximum sound pressure levels*, and also the *diurnal variation of exposure*.

The SiRENE project (Short and Long Term Effects of Transportation Noise Exposure) aims at investigating the health effects of exposure to transportation noise in the Swiss population. We focus on the relationship between noise exposure from individual and combined transportation sources and three main outcomes: annoyance, sleep disturbances and cardio-metabolic risk. Cardio-metabolic morbidity and mortality is investigated, with the aim to derive exposure-response associations for the various transportation noise sources, using data from two on-going Swiss epidemiological studies (SAPALDIA and SNC, see below) dating back to the 1990s. Underpinning and linking these three components is the detailed, nationwide modelling of road, railway and aircraft noise exposure to cohort participants' residential and occupational address.

The SAPALDIA (Swiss Study on Air Pollution and Lung Disease in Adults) Biobank is a large chronic disease cohort initiated in 1991 to investigate the effects of air pollution on respiratory and cardiovascular health in adults [21, 22]. The baseline survey (1991, SAPALDIA 1) included 9,561 subjects residing in eight geographically distinct urban and rural communities across Switzerland. Two follow-up surveys have since been conducted, including 8,047 subjects in 2000 (SAPALDIA 2) and ~6,000 in 2010-11 (SAPALDIA 3). Over the 20 year period, detailed questionnaire information on lifestyle, sleep problems, noise annoyance, medical history and risk factors as well as blood and DNA samples have been collected and stored. Full residential history has also been compiled over the entire follow-up period, as well as addresses for secondary residence and work place. The database contains exposure relevant information including the building floor level where each participant lives, the orientation of the sleeping room relative to the street, the type and age of the building, and personal window-opening habits. The Swiss National Cohort (SNC) is a national longitudinal research platform which uses a probabilistic linkage of census with mortality records [23]. Participation in the censuses (years 1990, 2000 and 2010) was mandatory and enumeration is near-complete. The database is continuously updated with mortality and migration data, and the fully linked SNC includes an individual record for each person in Switzerland, a household record, and a building record including exact geocodes for the home address. It also contains exposure-relevant information including whether persons live (or lived) on the ground floor or above, type of the housing (e.g. detached house, apartment), and age of the building or date of the last renovation.

The accuracy of geocodes in SAPALDIA and the SNC is essential, directly impacting the accuracy of the noise exposure assessment as a whole. Pilot work in SAPALDIA suggested that annoyance can be used as an internal validation criterion to improve exposure assignment, and that building age is one of the most important modifying factors [24]. Furthermore, the SAPALDIA and SNC cohorts span several decades over which transportation in Europe has shown steady growth [25], altering the soundscape and exposure situation for many residents. Recognizing this spatial and temporal change in noise levels across the population is crucial for the epidemiological analysis, in order to minimize exposure misclassification in the assessment of long-term health effects.

In the following, we present the noise exposure analysis of SiRENE and the major challenges of this task. We address the integration of geo-referenced input datasets from various sources (infrastructure and building datasets, population data address directories and geocodes of study participants), the processing of traffic information, the noise calculations as well as the assignment of traffic noise exposure characteristics from façade points to dwelling units.

2 Methods

2.1 Overview

According to the overall objective of SiRENE, noise exposure data is required for virtually all residential households in Switzerland - essentially each dwelling of every building. Precondition for the corresponding exposure

assessment is detailed and accurate information about building coordinates and heights as well as the number of floors per building. For each building and floor, façade points are defined for which noise calculations for road, railway and aircraft noise are performed for the years 2011, 2001 and 1991. Due to the three decades of interest in the SiRENE project, represented by three specific years, data compilation and processing of adequate input data is repeated for each year.

The number of dwellings per building can be extracted from census data. The façade points are assigned to dwelling units, allowing for a detailed exposure analysis per unit. Using the census data (GWS 2011), each building record is geocoded and has the number of residents attributed. The same applies for participants of SAPALDIA and SNC cohorts, which are also assigned to distinct dwelling units as part of this noise exposure analysis.

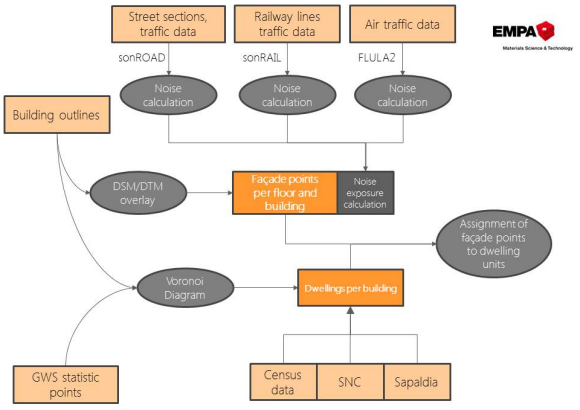

Fig. 1. Flow chart depicting the data processing, from the input data to the exposure analysis. Further information about sonROAD is available in [26], for sonRAIL in [27] and FLULA2 in [28].

In SiRENE we implement a more detailed temporal evaluation of noise exposure than recommended in the Swiss Noise Abatement Ordinance (NAO) [29] to have estimates relevant to the actual sleeping behavior of the population. Time periods in the SiRENE project are: 07:00-19:00 L_{Day}, 19:00-23:00 $L_{Evening}$, 23:00-07:00 L_{Night}. A particular focus is on the late night and early morning hours when a large proportion of the population is still asleep. The night period is thus subdivided into four further partitions: 23:00-01:00, 01:00-05:00, 05:00-06:00, and 06:00-07:00.

Our analysis is based on and closely related to the sonBASE Project [30], the Swiss Noise Geodatabase application. Some of the input data used for SiRENE is already incorporated in sonBASE, such as building outlines and the definition of the façade points. Another main reason for using sonBASE is the pre-calculated attenuations database between emission source and façade points. The following sections present the detailed data processing and methodology for the calculation year 2011, as well as strategies to overcome data availability issues for the calculation years 2001 and 1991.

2.2 Building outlines and definition of façade points

Building outlines from the period 1998 – 2006 are defined in the digital landscape model of Switzerland (1:25,000 VECTOR25) from the Swiss Federal Office of Topography, Swisstopo [31]. VECTOR25 displays connected houses as single buildings. Thus, one building can have several entrances and dwellings. The accuracy of position of the building polygons is in the order of 3 – 8 metres.

The building heights are derived by incorporating information from a digital surface model (DSM) and digital terrain model (DTM) from Swisstopo. The DSM provides three-dimensional information for all artificial and natural obstacles and objects on earth's surface, while the DTM displays solely the natural terrain. The deviation between these two datasets is used to determine the height of the buildings, and subsequently the number of floors. The number of floors is based on an assumed 2.5 to 3.3 metre floor height, with a base height of 1.5 metres.

On the defined buildings we assign façade points. Façade or receiver points represent the 3-D point location at which the incoming transportation noise is calculated. A maximum of 3 façade points, spaced a minimum distance of 5 metres, are specified for each façade and floor of all buildings. We created a total of 54,300,000 façade points, assigned to 1,813,000 buildings throughout Switzerland.

Buildings and façade points for earlier periods are also necessary to calculate noise exposure for 2001 and 1991. The earliest useful digital dataset for buildings in Switzerland was created by Swisstopo in 2004, with data originating from 1996 to 2000. Although this dataset lacks streets and buildings built between 2000 and 2001, it is sufficient for the definition of façade points, dwellings within buildings and the subsequent exposure calculation. The terrain model is the same as used for the analysis 2011, as there is little change assumed. The same buildings dataset created for 2001 is used for 1991, as there is no more accurate data available.

2.3 Definition of dwelling units and assignment of façade points

In a further step, these façade points are assigned to dwelling units within the buildings. The building footprints in VECTOR25, however, do not include separating walls between dwellings. In order to define distinct dwellings, we use geo-coded raster point information about dwelling units, obtained from the building and dwelling statistics (GWS) from the Swiss Federal Statistics Office. These GWS points are located inside each VECTOR25 building. The number of points contained in each VECTOR25 building indicates how many distinct dwellings are contained within. Hence, multiple GWS coordinates can point to the same VECTOR25 building and, given that a building may have more than one floor with an assumed one dwelling per floor, multiple dwellings may be associated with the same GWS coordinates.

The GWS points allowed us to subdivide buildings into dwelling units by creating Voronoi (or Thiessen) polygons in ArcGIS (Figure 2). This approach does not claim to generate a realistic representation of the dwelling units. It is, however, sufficient for delineating the correct number of dwellings per building.

Due to a temporal mismatch between the GWS data (compiled in 2010) and older VECTOR25 data, these two datasets do not necessarily match. For example, if a building was replaced by a new construction, the GWS point might be displayed outside the VECTOR25 building outline. GWS points are thus assigned to the nearest building outline with a threshold distance of 20 metres. All GWS points beyond this threshold are eliminated. A possible solution to this problem could be the generation of a default building around GWS points more than 20 metres from an existing building outline. While this approach fell short of computing capacity and this is a very rare case, we decided to erase these entrance points without substitution.

The number of inhabitants per GWS point is next assigned to the subdivided buildings respectively to individual dwelling units. SNC datasets already contain the number of inhabitants per dwelling and furthermore the floor on which they reside [23].

The final step is to assign the façade points available in sonBASE to dwelling units (Figure 3). Dwellings typically span buildings, and have more than one facade point. The subsequent epidemiological analysis in SiRENE considers both the highest exposure of each dwelling and also the potential benefit of quiet façades. Since we use Voronoi polygons and existing façade points from sonBASE, there is the possibility that some dwellings are left without a façade point. Dwelling units A and B in Figure 3 contain

Fig. 2. Buildings and Voronoi polygons to represent dwellings based on the GWS points.

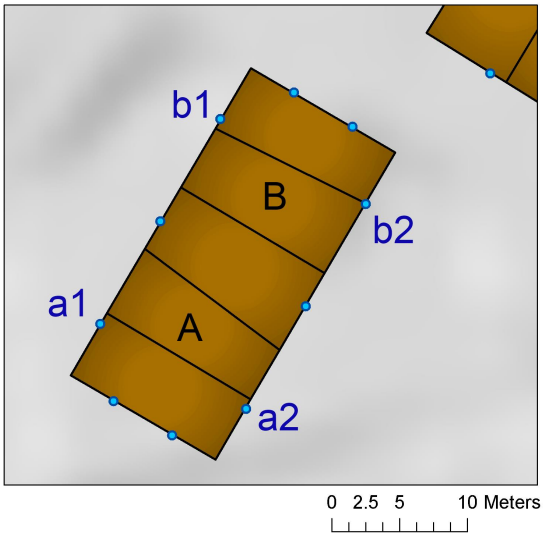

Fig. 3. Façade points and dwelling units within a building showing assignment of the nearest façade point to units with none.

no façade point within their boundaries, thus the closest façade points are assigned (e.g., façade points a1 and a2 are assigned to A; b1 and b2 are assigned to B).

2.4 Data processing and noise calculation

2.4.1 Road traffic

Road traffic noise exposure calculation requires information about the geometry and classification of all streets as

well as detailed traffic statistics, including yearly averages and diurnal variations, for all vehicle categories. Statistics and datasets concerning this information vary in quality and availability, and historical information is often particularly difficult to obtain.

The geometries of all existing roads in Switzerland are based on the sufficiently accurate VECTOR25 street section dataset. We supplement this with further geo-referenced data including:

- slope of each road section,
- road type and width,
- speed limit,
- bridge construction height, and
- traffic statistics.

To obtain the slope of each road section, we projected the VECTOR25 data on the DSM and calculated change in slope. Consequently the road is subdivided into sections corresponding to the changes of slope. Information about the correct height of bridge constructions is derived by manually overlaying VECTOR25 with DSM.

Information about noise barriers was obtained from the Federal Road Office (FEDRO) and the cantonal offices for infrastructure and traffic. The year of construction is available for ∼20% of the noise barriers. Adequate data specifically for the years 2001 and 1991 is not available. State and cantonal expenditures on noise barriers indicate that the large majority of the noise barriers in Switzerland have been built after 2000 (∼85%), hence we assume that if there is no construction date, it did not exist in 1991 or 2001 [32].

By comparing the geo-data with satellite images we controlled and corrected the dataset. False noise barriers, such as bushes and shrubs which were sometimes coded as noise barriers by mistake, were manually deleted. We rendered the surface resolution more precisely by generating a Triangulated Irregular Network (TIN) from a 2 × 2 metre resolution DTM using ArcGIS 3D Analyst. Thus more obstacles such as small hills or slopes, which influence the sound propagation similarly to a noise barrier, could be identified. In the case of missing values for height of the noise barriers, we assumed a default value of 3.5 metres for barriers proximate to highways and 2 metres for all other barriers. In total we incorporated 1,700 noise barriers (with an overall length of 238 km) at cantonal roads and 870 noise barriers (284 km) at national highways. Default height values were assigned to 30% of the barriers at cantonal roads. As there was no information for the barrier heights at national highways, 100% of these barriers were assigned the default height.

Traffic information can be gathered from traffic census monitoring systems. However such data is typically only available for highways and major roads. For Switzerland the federal department of the environment, transport and energy DETEC provides annual statistics that currently cover about 17% of all streets and about 60% of the total traffic volume. In sonBASE, a sophisticated traffic model developed by Arendt Consulting, is implemented [33] that combines this dataset with Swiss federal census data of population and business enterprises to generate traffic information also for the street sections not represented by monitoring stations. Based on assumptions of a distinct traffic behavior between dwellings and enterprises or other targets, the model creates daily average traffic statistics for passenger cars and heavy traffic. The Arendt road traffic model [34, 35] also includes a classification of the road type and the number of lanes. In combination with information about speed limits the specifications for each street section were defined. Due to its high temporal and spatial resolution and traffic statistics from each street category, the model allows for a highly detailed calculation of road traffic noise for the year 2011.

For the past situations we performed a back scaling based on the simulation of 2011. In a first step the street network was adjusted. Comparing the infrastructure of 2011 with 2001 for example this resulted in a reduction of the total street network of 3%. Then available traffic census data was again combined with information on local population density and the number of workplaces for the years 1991 and 2001. After a reevaluation with the Arendt model, scaling factors were derived for each street section.

Finally we performed the traffic noise calculation using the emission model of sonROAD [26] and the propagation model of StL-86 [36].

2.4.2 Railway traffic

Basic geometry for railway tracks is derived from information provided by the Federal Office of Transport (FOT). Supplemental data about the rail system, including the start and end of route sections and the coordinates of switch points, was provided by the Swiss Federal Railways (SBB). SBB also provided detailed information dating back to the 1970s about the year of construction, x-y-coordinates and height of noise barriers. Datasets contained information about a total of 2,055 noise barriers with an overall length of 244 km.

SBB provided the aggregated daily traffic for 2011 for each of the main train types, including actual driving speed. The diurnal variation of railway traffic, which is cru-

cial information for the subsequent exposure calculation, is also available and will be incorporated. Historical data of railway traffic statistics will be incorporated as soon as it will be available. However it already became apparent that traffic information will not be on hand for all railway lines, especially not for the year 1991. For these railway lines, where historical data is not available, a back-scaling will be performed using global scaling parameters for passenger and freight trains.

The traffic statistics are based on a classification system with 9 different train types including intercity trains, commuter trains and freight trains. The composition of these trains is varied for each year of calculation, representing the rolling stock in use during this period. Of great importance on the resulting noise exposure level is thus the noise remediation program for rolling stock which has been conducted in Switzerland since 2000. In this program the braking systems of all Swiss vehicles with cast-iron brakes are replaced with braking blocks made of composite material, resulting in a noise reduction of up to 10 dB(A) [37].

The emission of railway noise is calculated with the recently developed calculation model sonRAIL [38]. An essential part of the model is the emission data base, which has been developed based on an extensive measurement campaign and contains emission data for the major track constructions and rolling stock types used in Switzerland.

The corresponding propagation model [27] however has so far not been applied for sonBASE. As a propagation calculation for entire Switzerland was not possible within the given time frame, an attenuation database of calculations with the previous Swiss railway noise model SEMIBEL [39] was used instead.

2.4.3 Air traffic

The nature and level of detail for air traffic data is entirely different to that for road and rail. Ongoing radar monitoring of each single aircraft movement for the main airports Zurich, Geneva and Basel provide a high level of detail, including exact event times and the actual flown flight tracks. Combined with traffic statistics delivered by the airports, each radar track can be assigned to a specific aircraft type. The main runways infrastructure at Swiss airports did not change since 1991. The radar data we used contains information about the x-y- and z-axis in time steps of approximately 4 seconds, from which the actual flown speed is deduced.

The aircraft noise calculation is performed with FLULA2 [28, 40], which includes a sound source database with information about sound directivity patterns of almost every conventional aircraft type operated in Switzerland. The sound source database is based on measurements of real traffic, distinguishing landings and take-offs. FLULA2 performs a time-step simulation of individual flights and yields maximum noise levels ($L_{A,max}$) and sound exposure levels (L_{AE}) as results.

The acoustical footprint of an aircraft is the noise exposure, on average, created by a certain aircraft type on a specific route, calculated for a receiver grid with a constant height of 4 metres above terrain. The average is based on a sufficiently large number of aircraft of the same type flying the specified route. By assuming that the existing air routes remain unchanged, acoustical footprints can be taken from other years if radar data is not available. In FLULA2 these acoustical footprints can be imposed with an assumed air traffic volume to calculate the air traffic noise exposure for the survey years where radar data is missing [41].

Radar data for Zurich, the largest Swiss airport, is available for each survey year at an hourly resolution, with information about aircraft types and the flown air routes. For Geneva, the second largest Swiss airport, data availability is analogue only for the calculation year 2011. Radar data is not available for the years 2001 and 1991. For the exposure calculation of 2001 and 1991 traffic statistics including the fleet mix, from the Federal Office of Civil Aviation (FOCA), will be used with available acoustical footprints dating from the year 2000.

Yearly calculations for Basel airport have always been conducted in France, so data availability is limited. We therefore use data derived from a calculation made by Empa in 2001, based on highly detailed traffic statistics for this year, and 1999 footprints. Due to the lack of information, these footprints are also assumed for the exposure calculations for 2011 and 1991. For 2011 and 1991 traffic statistics from FOCA are used.

Military airfields are not taken into account with the exception of Payerne, as Payerne is of great importance for SAPALDIA. Information about military aircraft operations is generally poor. Radar data is not accessible, thus solely idealized flight paths are used. The aircraft types and the number of flights are known. However the hours of operation are only accessible in very generalized way, differentiating morning, afternoon and evening flights.

Small airfields used for private and business aviation are not taken into account as there is no reliable data available.

3 Results

Key interim results for the ongoing SiRENE noise exposure analysis are presented below.

3.1 Improvement in sound propagation calculation using topography

As already mentioned in section 2.4.1 topography is represented more precisely for the actual calculations compared to the sonBASE 2006 and previous versions. Figure 4 and Figure 5 show as an example a section of a highway close to Geneva, where the highway is embedded in a cutting before entering a tunnel. As a consequence of the rough resolution of the terrain model DHM25 of swisstopo used for sonBASE 2006 no noticeable reduction of the source takes place and sound seems to propagate freely resulting in the levels as depicted in Figure 4. With the refined topography the basin situation is represented more accurately, yielding a prominent barrier attenuation and consequently significantly lower exposure levels in the vicinity of the source, as shown in Figure 5. It can be concluded that a more detailed representation of topography results in major increase of the accuracy of the resulting exposure analysis.

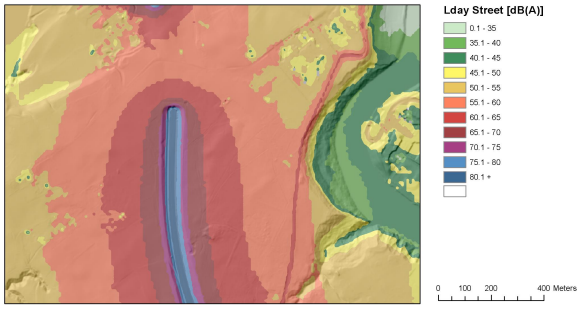

Fig. 4. Sound propagation and topography in sonBASE 2006 based on the terrain model DHM25 of swisstopo, calculated for a height of 4 m above terrain.

3.2 Calculation at facade points

The exposure calculation was performed on 54,3 million façade points of 1,8 million buildings of the VECTOR25 dataset. As an example, Figure 6 shows an excerpt of a noise map including the exposure at façade points for road traffic for the period of 06:00-18:00 (L_{Day}).

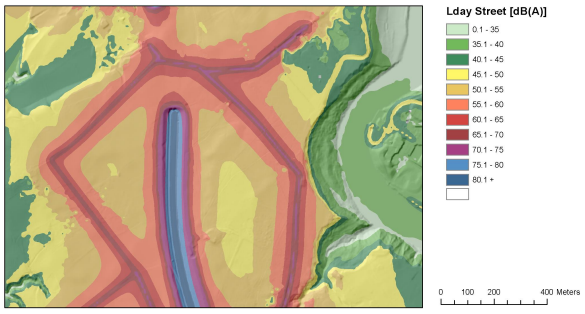

Fig. 5. Sound propagation and topography in sonBASE 2010 based on the terrain model swissALTI3D of swisstopo with a 2 × 2 meter resolution.

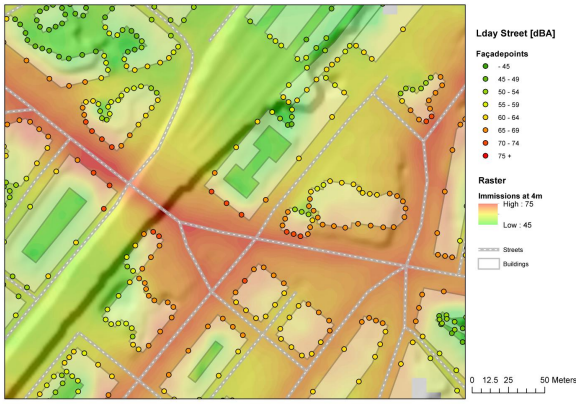

Fig. 6. Noise map and exposure at façade points of road traffic for the period of 06:00-18:00 (L_{Day}).

Exposure results can be broken down to single buildings, allowing the analysis of distinct buildings or areas of interest. Calculation results of a detached house parallel to a street with no reflections from surrounding buildings are shown in Figure 7. Of each building side, the façade point with the highest exposure is represented. As can be seen, the levels vary substantially for the different façades. This illustrates the importance of the orientation of dwelling units as discussed in section 2.3.

3.3 Statistical analysis for the year 2011

A total of 7,9 million inhabitants, which represent the entire Swiss population, were assigned to buildings and subsequently to distinct dwellings and façade points. For our preliminary results we assigned the inhabitants to the façade point with the highest L_{eq} value, taking into account a potential overestimation of the true exposure as rooms like sleeping and living rooms are likely to be oriented toward less noisy façades. Figure 8 shows histograms of exposure of the Swiss population, in noise ex-

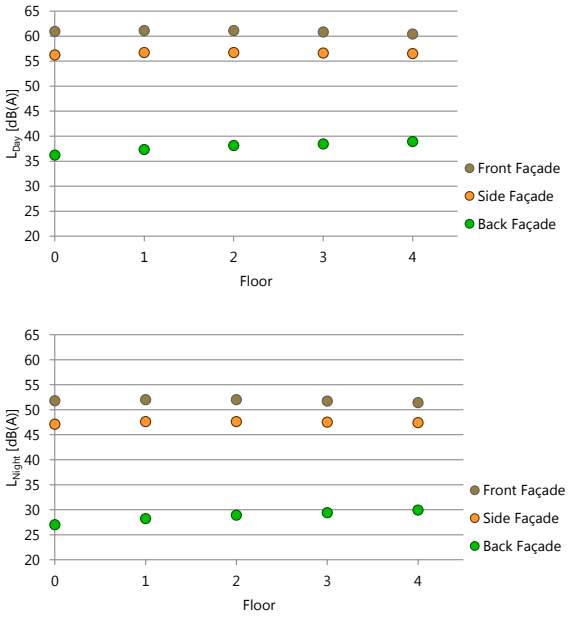

Fig. 7. Road traffic noise exposure at façade points for a single building for the period of 06:00-18:00 (L_{Day}) and 22:00-06:00 (L_{Night}).

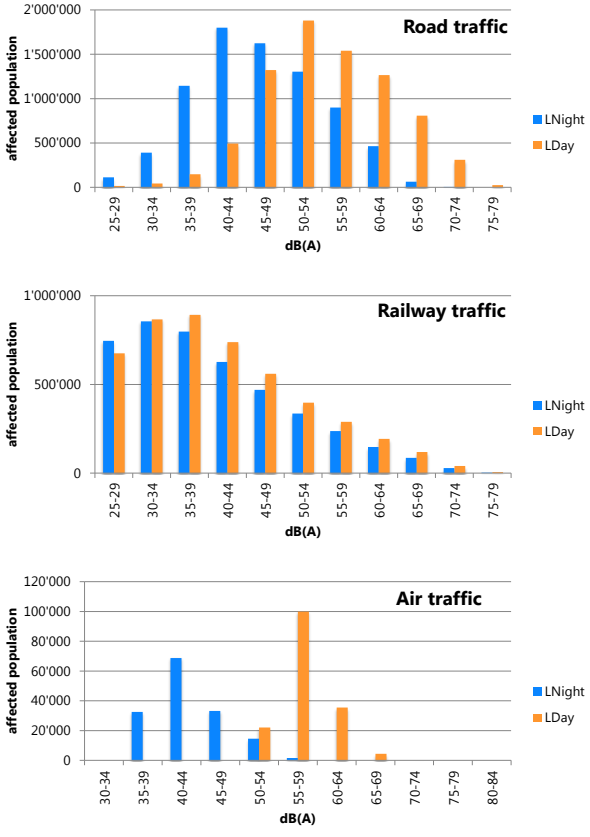

Fig. 8. Distribution of the Swiss population exposure to road, railway and air traffic noise, L_{Night} and L_{Day}, classification in 5 dB(A) steps for the year 2011.

posure categories in 5 dB(A) steps, respectively for L_{Day} and L_{Night} for the year 2011. The figures depict clearly the shift from lower to higher exposure levels during the day relative to the night period for road and air traffic noise. In the case of railway noise this shift is not present, although the overall exposure is slightly lower during the night period.

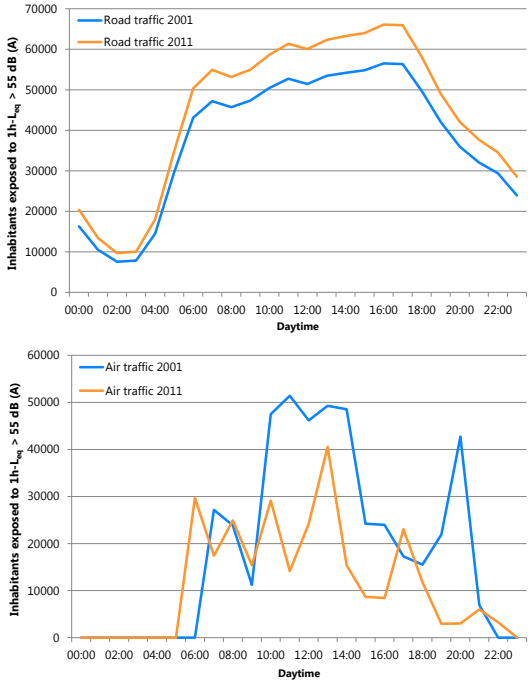

Fig. 9. Number of affected inhabitants by road and air traffic noise (1h-L_{eq}) >55 dB (A) for an area of 35 km^2 south of Zurich airport.

3.4 Comparison of noise exposure 2001 and 2011

Figure 9 yields a comparison of the number of people exposed to 1h-Leq levels above 55 dB(A) for road and air traffic in the course of the day for the years 2001 and 2011. The calculation was performed for an area south of the Zurich international airport of 7 × 5 km. The area was inhabited by 103'727 persons in 2001 and 115'145 in 2011. Primarily as a consequence of population growth a general increase of the number of affected people can be seen for road traffic, which is to a large extent independent of the time of the day.

The situation for air traffic differs substantially from road traffic. First an overall decrease of the number of af-

fected people can be seen. This decrease of overall air traffic noise at Zurich airport has also been stated by other investigations [42]. Second it is interesting to see how much the temporal pattern varies between the two years, indicating the great operational variability of the airport as a consequence of its three runways with differing orientation.

4 Discussion

4.1 Noise exposure assessment for environmental health studies

Accurate assessment of exposure is an essential component of any epidemiological study aiming at elucidating the exposure-response relationships. Noise models which are used for regulatory strategic noise maps (i.e. in accordance with the END) only reproduce long-term equivalent continuous sound level and do not meet the needs of SiRENE. We thus present a novel approach for calculating a highly detailed, nation-wide assessment of (individual) noise exposure. In addition to average exposure levels the potential health effects of other noise characteristics including distribution of maximum sound pressure levels, number of events, and the diurnal variation are studied as well.

Typically the exposure assessment in socio-acoustic surveys or epidemiological studies on the cardiometabolic effects of transportation noise is based on the gridded (mapped) output from these models [43–47], while only a few use the calculation results assigned to the home address and distinct dwellings [48–50]. To our knowledge, none have considered the additional noise characteristics we compute for SiRENE in epidemiological research on long term effects. Furthermore, except for Huss et al. [47] which was also conducted in Switzerland, none of these examples have been conducted on a nation-wide basis.

4.2 Uncertainty of the exposure analysis

The quality of SiRENE's exposure-response analyses and risk assessments depends strongly on the reliability and credibility of the input data and the models used for the exposure calculations. As nationwide exposure data cannot be taken from measurements, models have to be used to provide a realistic representation of the exposure situation. Models as such pose a potential risk of failing to depict reality sufficiently. With respect to epidemiological studies, therefore, systematic errors are more significant than stochastic uncertainties. Validation results of all calculation models used in SiRENE have been published before. In combination with the uncertainties of input parameters, such as the traffic modeling or the accuracy of geo-referenced data, a combined standard uncertainty of 2.7 dB for daytime road traffic noise calculations and 3.1 dB for nighttime calculations can be deduced. The resulting overall uncertainty of the A-weighted equivalent continuous sound level ($L_{A,eq}$) calculations of railway noise with sonRAIL remains below 2.5 dB for day and night [51]. Given the real fleet mix of aircraft and the corresponding noise source data, the remaining standard uncertainties of the aircraft noise calculation software FLULA2 add up to 0.5 dB for daytime calculations and 1.0 dB for nighttime calculations [28, 52]. Detailed information about this topic has been previously published [30, 51, 52].

On the noise exposure assessment side, achievements of SiRENE include compiling and refining data about existing buildings and traffic infrastructure and integrating their true spatial position and dimensions in the subsequent exposure calculations. By not only using existing standard models, but additionally refining and combining them with other sources, we created the most sophisticated model of the current situation in Switzerland with respect to noise. Shortcomings deriving from the temporal mismatch of input data, as described in section 2.3, have to be taken into account. However, there is no systemic error emerging from this issue and the absolute error has been kept to a minimum.

The quality of input data regarding traffic statistics has essential influence on the accuracy of the subsequent noise exposure analysis. In contrast to railway and air traffic, statistics for road traffic were taken from a model. The Arendt model used for the road traffic statistics has proved to be relatively precise in its modelling, with a weighted deviation of 4.6% from real traffic [34], which translates into an insignificant variation of +0.2 to −0.2 dB(A).

4.3 Challenges related to historical noise exposure estimation

Data quality and availability improved markedly over the past 20 years, leading to the current situation where a large variety of often high quality information is available. Although the datasets representing the present situation cannot be used without corrections and processing, the 2011 information is by far more reliable compared to datasets available from 2001 or 1991. This particularly ap-

plies to the geo-referenced data such as buildings, streets and railway lines, and statistics on traffic and population.

One of the main objectives of SiRENE is to depict the temporal variation in noise exposure of the Swiss population from 1991 to 2011 and to assess the long-term health effects of noise. Thus, noise calculations representing the past years have to be performed as similar as possible to those done with the state-of-the-art models available for the current situation in 2011. As pointed out in the methods, this led to a variety of challenges mainly related to data availability.

Through our overall approach, we solved many of these problems. The solutions rely mainly on certain simplifications, as the required datasets for the years of interest are simply not available. This includes road, railway and air traffic statistics of 1991, leading to larger uncertainties for this particular year. The uncertainty, however, is expected to be lowest for major sources such as highways, major railways and national airports. Thus, the identification of highly exposed buildings will be more accurate than in the low exposure range. The impact of this exposure assessment error on the epidemiological results will be addressed by sensitivity analyses and simulation studies.

4.4 Conclusions and impact of SiRENE on further surveys

It is well recognized that noise exposure from transportation is one of the most widespread sources of environmental stress, especially in densely populated areas such as those in Europe and mega-cities outside of Europe. Based on population noise exposure derived from the strategic noise maps through END, the annual burden of environmental noise in Western Europe is an estimated loss of 1 – 1.6 million healthy life-years [1]. As demonstrated in a recent special issue on noise, development of a common methodological framework under END for noise assessment in Europe is ongoing [53]. Improvements to the strategic noise maps in future will no doubt help to improve calculation of disease burden. Equally important, however, is the derivation of reliable exposure-response relationships based on large population studies. To support this particular type of research, more refined exposure models for a broader range of metrics are needed compared to the outputs for strategic noise modeling.

A key aspect of our exposure assessment is the evaluation of noise in three-dimensions. This allows us to link exposures to individuals based on the floor of their particular dwelling. Also, given that we estimate exposures for the entire Swiss population our subsequent exposure database can be used for other studies besides SiRENE.

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Additional members of the SiRENE study group are: Cajochen C, Clark I, Eze I, Foraster Pulido M, Héritier H, Lang C, Pieren R, Rudzik F, Schaffner E, Thiesse L.

References

[1] WHO, 2011. Burden of Disease from Environmental Noise: Quantification of Healthy Life Years Lost in Europe. ed. L.B. Fritschi, Lex; Kim, Rokho; Schwela, Dietrich; Kephalopolous, Stelios and Place. World Health, Copenhagen.

[2] 2002. Directive 2002/49/EC of the European Parliament and of the Council of 25 June 2002 Relating to the Assessment and Management of Environmental Noise.

[3] Curcuruto, S., R. Silvaggio, R. Amodio, L. De Rinaldis, E. Mazzocchi, F. Sacchetti, and M. Stortini, 2012. HUSH project contribution to environmental noise directive implementation and revision, focusing on noise management and public information tools. in 41st International Congress and Exposition on Noise Control Engineering 2012, INTER-NOISE 2012.

[4] WHO, 2009. Night Noise Guideline for Europe. World Health Organization.

[5] Miedema, H.M.E., W. Passchier-Vermeer, and H. Vos, 2003. Elements for a position paper on night-time transportation noise and sleep disturbance. ISBN-90-6743-981-9. TNO.

[6] Asdrubali, F., S. Schiavoni, F. D'Alessandro, C. Schenone, and I. Pittaluga, 2012. Innovative aspects in noise mapping and dissemination activities within life+ "NADIA" project. in 19th International Congress on Sound and Vibration 2012, ICSV 2012.

[7] Ribeiro, C., Anselme, C., Mietlicki, F., Vincent, B., Da Silva, R., Gaudibert, P., 2013. At the heart of Harmonica project: the Common Noise Index (CNI), in 42nd International Congress and Exposition on Noise Control Engineering, Internoise 2013: Innsbruck. p. 10.

[8] Blatter, K., P. Graw, M. Münch, V. Knoblauch, A. Wirz-Justice, and C. Cajochen, 2006. Gender and age differences in psychomotor vigilance performance under differential sleep pressure conditions. Behavioural Brain Research 168, 312-317.

[9] Frey, S., S. Balu, S. Greusing, N. Rothen, and C. Cajochen, 2009. Consequences of the Timing of Menarche on Female Adolescent Sleep Phase Preference. PLoS ONE 4.

[10] Brink, M., P. Lercher, A. Eisenmann, and C. Schierz, 2008. Influence of slope of rise and event order of aircraft noise events on high resolution actimetry parameters. Somnologie -

Schlafforschung und Schlafmedizin 12, 118-128.
[11] Griefahn, B., A. Schuemer-Kohrs, R. Schuemer, U. Moehler, and P. Mehnert, 2000. Physiological, subjective, and behavioural responses during sleep to noise from rail and road traffic. Noise Health 3, 59-71.
[12] de Coensel, B. and A.L. Brown, 2012. Modeling road traffic noise using distributions for vehicle sound power level, in internoise 2012, New York: New York.
[13] Brown, A.L., 2013. Issues in the measurement of sleep-related noise events in road traffic streams, in Acoustics 2013: Victor Harbor, Australia. p. 1-5.
[14] Finegold, L.S. and e. al., 1994. Community Annoyance and Sleep Disturbance: Updated Criteria for Assessment of the Impacts of General Transportation Noise on People. Noise Control Engineering Journal 42, 25-30.
[15] Miedema, H.M.E. and H. Vos, 1998. Exposure-response relationships for transportation noise. Acoustical Society of America 104, 3432-3445.
[16] Birchler-Pedross, A., C.M. Schröder, M. Münch, V. Knoblauch, K. Blatter, C. Schnitzler-Sack, A. Wirz-Justice, and C. Cajochen, 2009. Subjective well-being is modulated by circadian phase, sleep pressure, age, and gender. Journal of Biological Rhythms 24, 232-242.
[17] Cajochen, C., C. Jud, M. Münch, S. Kobialka, A. Wirz-Justice, and U. Albrecht, 2006. Evening exposure to blue light stimulates the expression of the clock gene PER2 in humans. European Journal of Neuroscience 23, 1082-1086.
[18] Späti, J., M. Münch, K. Blatter, V. Knoblauch, L.A. Jones, and C. Cajochen, 2009. Impact of age, sleep pressure and circadian phase on time-of-day estimates. Behavioural Brain Research 201, 48-52.
[19] Schmidt, C., F. Collette, Y. Leclercq, V. Sterpenich, G. Vandewalle, P. Berthomier, C. Berthomier, C. Phillips, G. Tinguely, A. Darsaud, S. Gais, M. Schabus, M. Desseilles, T.T. Dang-Vu, E. Salmon, E. Balteau, C. Degueldre, A. Luxen, P. Maquet, C. Cajochen, and P. Peigneux, 2009. Homeostatic sleep pressure and responses to sustained attention in the suprachiasmatic area. Science 324, 516-519.
[20] Münch, M., S. Kobialka, R. Steiner, P. Oelhafen, A. Wirz-Justice, and C. Cajochen, 2006. Wavelength-dependent effects of evening light exposure on sleep architecture and sleep EEG power density in men. American Journal of Physiology - Regulatory Integrative and Comparative Physiology 290, R1421-R1428.
[21] Ackermann-Liebrich, U., B. Kuna-Dibbert, N. Probst-Hensch, C. Schindler, D. Dietrich, E. Stutz, L. Bayer-Oglesby, F. Baum, O. Brändli, M. Brutsche, S. Downs, D. Keidel, M. Gerbase, M. Imboden, R. Keller, B. Knöpfli, N. Künzli, L. Nicod, M. Pons, P. Staedele, J.-M. Tschopp, J.-P. Zellweger, and P. Leuenberger, 2005. Follow-up of the Swiss Cohort Study on Air Pollution and Lung Diseases in Adults (SAPALDIA 2) 1991–2003: methods and characterization of participants. Sozial- und Präventivmedizin 50, 245-263.
[22] Downs, S.H., C. Schindler, L.J. Liu, D. Keidel, L. Bayer-Oglesby, M.H. Brutsche, M.W. Gerbase, R. Keller, N. Kunzli, P. Leuenberger, N.M. Probst-Hensch, J.M. Tschopp, J.P. Zellweger, T. Rochat, J. Schwartz, and U. Ackermann-Liebrich, 2007. Reduced exposure to PM10 and attenuated age-related decline in lung function. The New England Journal of Medicine 357, 2338-47.
[23] Bopp, M., A. Spoerri, M. Zwahlen, F. Gutzwiller, F. Paccaud, C. Braun-Fahrländer, A. Rougemont, and M. Egger, 2009. Cohort profile: The Swiss National Cohort - A longitudinal study of 6.8 million people. International Journal of Epidemiology 38, 379-384.
[24] Vienneau, D., N. Probst-Hensch, J. Wunderli, B. Struchen, E. Fischer, and M. Röösli, 2012. Epidemiological Noise Exposure Assessment - A Pilot Study. 00.0066.PZ / L142-2822. Federal Office for Environment.
[25] Mantzos L., C.P., Kouvaritakis N., Zeka-Pschou M., 2003. European Energy and Transport Trends to 2030, O.f.O.P.o.t.E. Communities, Editor. European Commission.
[26] Heutschi, K., 2004. SonRoad: New Swiss road traffic noise model. Acta Acustica United with Acustica 90, 548-554.
[27] Wunderli, J.M., 2012. sonRAIL - from the scientific model to an application in practice, in Euronoise: Prague.
[28] Empa, 2010. FLULA2, Ein Verfahren zur Berechnung und Darstellung der Fluglärmbelastung. Technische Programm-Dokumentation. Version 4. Eidgenössische Materialprüfungs- und Forschungsanstalt (Empa), Abteilung Akustik / Lärmminderung, Dübendorf.
[29] NAO, 1986. Lärmschutz-Verordnung (LSV) vom 15. Dezember 1986 (Stand am 1. August 2010) (Noise Abatement Ordinance (NAO) from 15 December 1986 (Effective 1 August 2010)). SR 814.41.
[30] BAFU, 2009. SonBase - die GIS-Lärmdatenbank der Schweiz. Grundlagen, in Umwelt-Wissen. Bundesamt für Umwelt: Bern. p. 61.
[31] Swisstopo, 2007. VECTOR 25. Das digitale Landschaftsmodell der Schweiz. Bundesamt für Landestopografie, Wabern. 31.
[32] H., C.L.I.K.L.-F.N.P., 2007. Sanierung Strassenlärm. Stand und Perspektiven: Dezember 2006. Bundesamt für Umwelt: Bern. p. 50.
[33] Arendt. Verkehrsaufkommen flächendeckend ermitteln. 2009 [cited 01.04.2014]; Available from: http://arendt.ch/objects/websitedownload/53/download/VM_Flyer_d.pdf.
[34] 2010. Nationales Personenverkehrsmodell des UVEK, Basismodell 2005, B.f. Raumentwicklung, Editor. Bundesamt für Raumentwicklung.
[35] 2010. Nationales Personenverkehrsmodell des UVEK, Referenzzustand 2030, B.f. Raumentwicklung, Editor. Bundesamt für Raumentwicklung.
[36] BAFU, 1987. Computermodell zur Berechnung von Strassenlärm, Teil 1, Bedienungsanleitung zum Computerprogramm StL-86, in Schriftenreihe Umweltschutz. Bundesamt für Umweltschutz: Bern. p. 47.
[37] 2013. Monitoring Eisenbahnlärm. Bundesamt für Verkehr: Bern.
[38] Thron, T. and M. Hecht, 2010. The sonRAIL emission model for railway noise in Switzerland. Acta Acustica United with Acustica 96, 873 - 883.
[39] FOEN, 1990. SEMIBEL: Schweizerisches Emissions- und Immissionsmodell für die Berechnung von Eisenbahnlaerm. ed. S.F.O.f.t. Environment. Swiss Federal Office for the Environment.
[40] Pietrzco, J.S. and R.F. Hofmann, 1988. Prediction of A-weighted aircraft noise based on measured directivity patterns. Applied Acoustics 23, 29-44.
[41] Schäffer, B., R. Bütikofer, S. Plüss, and G. Thomann, 2011. Aircraft noise: Accounting for changes in air traffic with time of

day. Journal of the Acoustical Society of America 129, 185-199.
[42] Bissegger, M., 2013. Noise Management in the light of airport developement, in 42nd International Congress and Exposition on Noise Control Engineering, Internoise 2013: Innsbruck. p. 7.
[43] Andrew, W., L. Junenette, I. Jonathan, S. Melly, and F. Dominici, 2013. Residential exposure to aircraft noise and hospital admissions for cardiovascular diseases: Multi-airport retrospective study. BMJ (Online) 347.
[44] Anna, L., M. Blangiardo, L. Fortunato, S. Floud, K. De Hoogh, D. Fecht, R.E. Ghosh, H.E. Laszlo, C. Pearson, L. Beale, S. Beevers, J. Gulliver, N. Best, S. Richardson, and P. Elliott, 2013. Aircraft noise and cardiovascular disease near Heathrow airport in London: Small area study. BMJ (Online) 347.
[45] Beelen, R., G. Hoek, D. Houthuijs, P.A. Van Den Brandt, R.A. Goldbohm, P. Fischer, L.J. Schouten, B. Armstrong, and B. Brunekreef, 2009. The joint association of air pollution and noise from road traffic with cardiovascular mortality in a cohort study. Occupational and Environmental Medicine 66, 243-250.
[46] Gan, W.Q., H.W. Davies, M. Koehoorn, and M. Brauer, 2012. Association of long-term exposure to community noise and traffic-related air pollution with coronary heart disease mortality. American Journal of Epidemiology 175, 898-906.
[47] Huss, A., A. Spoerri, M. Egger, and M. Röösli, 2010. Aircraft noise, air pollution, and mortality from myocardial infarction. Epidemiology 21, 829-836.
[48] Selander, J., M.E. Nilsson, G. Bluhm, M. Rosenlund, M. Lindqvist, G. Nise, and G. Pershagen, 2009. Long-term exposure to road traffic noise and myocardial infarction. Epidemiology 20, 272-279.
[49] Sřrensen, M., Z.J. Andersen, R.B. Nordsborg, T. Becker, A. TjŘnneland, K. Overvad, and O. Raaschou-Nielsen, 2013. Long-term exposure to road traffic noise and incident diabetes: A cohort study. Environmental Health Perspectives 121, 217-222.
[50] Sřrensen, M., Z.J. Andersen, R.B. Nordsborg, S.S. Jensen, K.G. Lillelund, R. Beelen, E.B. Schmidt, A. Tjřnneland, K. Overvad, and O. Raaschou-Nielsen, 2012. Road traffic noise and incident myocardial infarction: A prospective cohort study. PLoS ONE 7.
[51] Thron, T. and M. Hecht, 2010. The sonRAIL emission model for railway noise in Switzerland. Acta Acustica united with Acustica 96, 873-883.
[52] Schäffer, B., S. Plüss, and G. Thomann, 2014. Estimating the model-specific uncertainty of aircraft noise calculations. Applied Acoustics 84, 58-72.
[53] Kephalopoulos, S., M. Paviotti, F. Anfosso-Lédée, D. Van Maercke, S. Shilton, and N. Jones, 2014. Advances in the development of common noise assessment methods in Europe: The CNOSSOS-EU framework for strategic environmental noise mapping. Science of The Total Environment 482–483, 400-410.

14. Appendix B

This paper does not belong to this thesis. It describes the derivation of the intermittency ratio used in this thesis

Intermittency ratio: A metric reflecting short-term temporal variations of transportation noise exposure

Jean Marc Wunderli[1], Reto Pieren[1], Manuel Habermacher[2], Danielle Vienneau[3,4], Christian Cajochen[5], Nicole Probst-Hensch[3,4], Martin Röösli[3,4] and Mark Brink[6]

[1] Empa, Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Dubendorf, Switzerland.

[2] N-sphere AG, Zürich, Switzerland

[3] Swiss Tropical and Public Health Institute, Basel, Switzerland

[4] University of Basel, Basel, Switzerland

[5] Centre for Chronobiology, Psychiatric Hospital of the University of Basel, Basel, Switzerland

[6] Federal Office for the Environment, Bern, Switzerland

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ORIGINAL ARTICLE

Intermittency ratio: A metric reflecting short-term temporal variations of transportation noise exposure

Jean Marc Wunderli[1], Reto Pieren[1], Manuel Habermacher[2], Danielle Vienneau[3,4], Christian Cajochen[5], Nicole Probst-Hensch[3,4], Martin Röösli[3,4] and Mark Brink[6]

Most environmental epidemiology studies model health effects of noise by regressing on acoustic exposure metrics that are based on the concept of average energetic dose over longer time periods (i.e. the L_{eq} and related measures). Regarding noise effects on health and wellbeing, average measures often cannot satisfactorily predict annoyance and somatic health effects of noise, particularly sleep disturbances. It has been hypothesized that effects of noise can be better explained when also considering the variation of the level over time and the frequency distribution of event-related acoustic measures, such as for example, the maximum sound pressure level. However, it is unclear how this is best parametrized in a metric that is not correlated with the L_{eq}, but takes into account the frequency distribution of events and their emergence from background. In this paper, a calculation method is presented that produces a metric which reflects the intermittency of road, rail and aircraft noise exposure situations. The metric termed intermittency ratio (*IR*) expresses the proportion of the acoustical energy contribution in the total energetic dose that is created by individual noise events above a certain threshold. To calculate the metric, it is shown how to estimate the distribution of maximum pass-by levels from information on geometry (distance and angle), traffic flow (number and speed) and single-event pass-by levels per vehicle category. On the basis of noise maps that simultaneously visualize L_{eq}, as well as *IR*, the differences of both metrics are discussed.

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Keywords: annoyance; cardiovascular effects; emergence; fluctuation; intermittent noise; sleep disturbance

INTRODUCTION

Transportation noise, largely from road, railway and aircraft traffic, is one of the most widespread sources of environmental stress and discomfort in daily life. Health effects of noise may emerge directly via autonomous stress reactions to the physical exposure or indirectly via negative affective states, for example, annoyance. Noise pollution is an important public health factor, with recent burden of disease estimates ranking it the second major environmental health risk after air pollution in Europe.[1] A recent review[2] summarized the current state of knowledge about auditory and non-auditory effects of noise.

Health Effects of Transportation Noise

The most prevalent non-auditory effect in a population exposed to environmental noise is annoyance. It may result from interference with daily activities, rest or sleep, and can be accompanied by negative emotional and behavioral responses such as anger, displeasure, exhaustion and by stress-related symptoms.[3,4] Importantly, humans perceive, evaluate and react to environmental sounds not only during daytime, but also when asleep. Laboratory and field studies have consistently demonstrated that transportation noise induces acute and short-term effects on sleep.[5,6] Acute physiological reactions to noise events during the night include conscious and unconscious awakenings, shifts to lighter sleep stages, cortical and cardiovascular arousals (increases of heart rate and blood pressure) and body movements.[6,7] The probabilities of such reactions are clearly correlated with acoustic characteristics of noise events, especially with the maximum sound pressure level and the slope of rise of the level. In field studies, noise events with L_{Smax} as low as about 35 dB(A) at the ear of the sleeper have been shown to induce electroencephalography awakening reactions in sleeping environments with low background noise.[8–10] Recent evidence as, for example, from the HYENA study[11] suggests that nocturnal noise exposure may be more relevant for the genesis of long-term cardiovascular outcomes than daytime noise exposure, probably because of repeated autonomic arousals (short-living heart rate and blood pressure increases, peripheral vasoconstriction and so on) during sleep that have been shown to habituate to a much lesser degree than cortical arousals.[5,12] A recent Swiss study presented evidence of an adverse effect of railway noise on blood pressure, that was especially associated with night-time exposure.[13] As it is well plausible that an ample proportion of the long-term cardiovascular health impacts of noise are triggered by repeatedly occurring cardiovascular arousals during sleep,[14] it is paramount for a (new) noise metric to amply reflect the acoustic characteristics of the noise events that potentially trigger such arousals.

[1]Empa Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Duebendorf, Switzerland; [2]n-Sphere AG, Zurich, Switzerland; [3]University of Basel, Basel, Switzerland; [4]Swiss Tropical and Public Health Institute, Basel, Switzerland; [5]Psychiatric Hospital of the University of Basel, Center for Chronobiology, Basel, Switzerland and [6]Federal Office for the Environment, Bern, Switzerland. Correspondence: Dr. Jean Marc Wunderli, Empa Laboratory for Acoustics/Noise control, Swiss Federal Laboratories for Materials Science and Technology, Ueberlandstrasse 129, Duebendorf CH-8600, Switzerland.
Tel.: +41 58 765 4748. Fax: +41 58 765 6993.
E-mail: jean-marc.wunderli@empa.ch
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Differences between Constant and Intermittent Noise

Many epidemiological studies, as well as many annoyance surveys consider noise exposure as equivalent continuous levels over longer time periods (e.g., L_{dn}, L_{den}, L_{night} or L_{day} [15,16]). It has been repeatedly shown that such energy-based exposure measures (alone) have limited explanatory power regarding annoyance or disturbance effects—yet essentially better and more sophisticated acoustic metrics to explain noise effects are likewise missing. This has earned the L_{eq} a reputation of being the "best of all the bad noise exposure metrics". However, the application of equivalent levels to predict the impact of noise on sleep has not met with much success.[17] Whereas the probabilities of event-related awakenings and cardiovascular arousals clearly increase with the maximum sound pressure level of noise events, average noise metrics usually fail to predict noise-induced sleep disturbances sufficiently.[6,18,19] One can conclude that depending on whether the noise source is intermittent (such as passing flights or trains) or continuous (such as road traffic from a busy highway), the effects of noise on sleep might be better explained by the number of noise events and their characteristics, than by average noise exposure.[8]

Figure 1 illustrates the effect of different degrees of source intermittency on the course of the sound pressure level for a given average exposure level (L_{eq} = 55 dB(A)) over an 8-h observation period, recorded along a railway double-track line (red) and a highway (blue). Although, in this example, both exposure situations yield the same average level during the observation period, the higher degree of intermittency of individual events, which is characteristic to railway traffic, produces more strongly pronounced fluctuations in the sound pressure level.

It is highly questionable that the two exposure situations shown in Figure 1 result in the same overall effect during any given time period. Although a larger impact on sleep by the exposure from the railway line at night time can be expected for the above mentioned reasons, the presence of calm periods of respite in between loud pass-by events may render the same railway noise more acceptable during daytime. The latter is reflected in the "rail bonus" which is inherent to many railway noise regulations in Europe. The constant noise from the highway on the other hand might pose no threat to a good night's sleep, while being considerably annoying at daytime. It is thus well possible that the established source dependence of reported annoyance (in particular, as reviewed by Miedema et al.[20]) can at least partially be explained by differences in noise variation over time.

It is important to note that the temporal variation characteristics of noise do not just vary between different source categories (road, rail and air), but also within the same source. This is most clearly obvious in road traffic noise, where the temporal structure of occurring noise events shows a high variation between small one-lane city streets with highly intermittent noise up to wide 4-, 6- or even 8-lane highways, which produce a nearly continuous sound exposure with very little fluctuation. Miedema concluded that highway noise produced higher annoyance as compared with main roads at comparable L_{Aeq24h}.[21] In contrast Lercher et al.[22] found higher annoyance prevalences for main-road traffic than for highway traffic noise of the same L_{den} level. The same group also found that the distance between dwellings and the nearest railway track was a level-independent predictor for railway noise annoyance, with higher annoyance ratings in people living < 300 m from the railway track.[23] These findings suggest an influence of the temporal structure of noise exposure on annoyance reactions—and maybe other health outcomes as well—however, the functional relationship is yet unclear.

Metrics for Acoustically Characterizing Intermittent Noise Situations

No matter how one looks at it, it seems essential that large-scale long-term health effect studies, as well as annoyance surveys, should consider more detailed characteristics of the temporal variation of the sound, as well as its emergence instead of only considering averaged exposure levels. In the last decades there have in fact been several proposals to add time-related variables to replace or supplement the L_{eq}. A discussion of alternatives or amendments to the L_{eq} can be found in TSG9[24] and Commission E.[25] Common approaches are the introduction of thresholds and the counting of the number and duration of events (e.g., the Noise and Number Index, which was common during the 1970s and 1980s) or an application of level statistics. The latter are, for example, the basis of derived quantities such as the Traffic Noise Index,[26] the Noise Pollution Level,[27] the Common Noise Index developed within the Harmonica project,[28] the concept of notice-events [29] or fluctuation and emergence, as used by Bockstael et al.[30] However, these metrics have so far not reached a broader application for regulatory purposes either because they highly correlate with the L_{eq} or for reasons of complexity to be implemented in or combined with common calculation models.

Rationale and Goals

Although the association between (night time) noise and health outcomes could be convincingly demonstrated in past research, the causal chain from reactions to individual noise events in the night to the long-term outcomes is not yet fully understood. We hypothesize that for a given exposure level the potential of noise

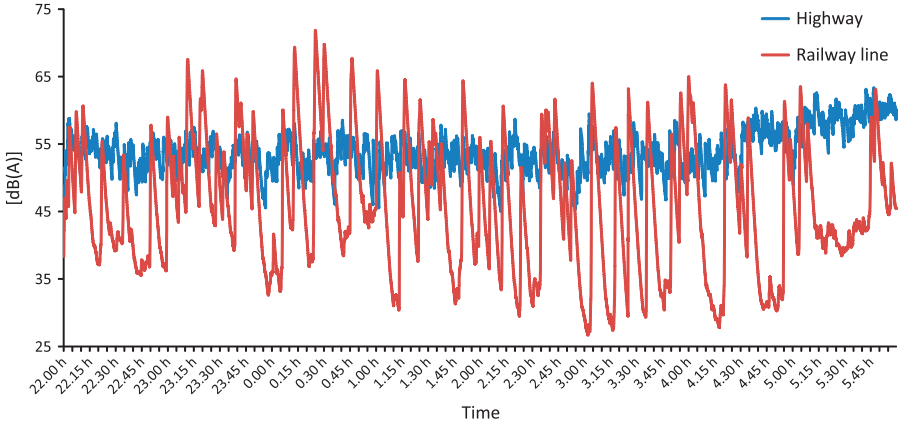

Figure 1. Level-time course of the sound pressure level produced by road traffic on a highway, at 7.5 m distance, and along a railway line with predominantly freight traffic events, at 250 m distance, for a time period of 8 h. Both signals have been recorded along the Swiss north-south transit axis on the Gotthard route (road and rail) between 22 and 06 h and normalized to 55 dB(A) L_{eq}.

to activate pathophysiologically relevant pathways is more pronounced in noise situations that produce relatively few but loud single events with calm periods in between, than constantly emitting noise sources with almost no variation of the sound pressure level over time. If this would hold true, one should observe higher risk estimates for cardiovascular and other health outcomes in exposure situations that emit noise intermittently rather than constantly. Therefore, a measure of intermittency may contribute, at least in part, to elucidate the large proportion of unexplained variance which is usually found in noise effect models that predict effects from average exposure solely. Such a measure could therefore be used as a second predictor metric, complementing energy equivalent average level metrics such as the L_{eq}.

The present work was stimulated by the demands of the *SiRENE* study[31] which investigates transportation noise effects in the Swiss population. One goal of the project is to elucidate the effect of source intermittence on cardiovascular morbidity and mortality in two large-scale epidemiological studies (the *SAPALDIA* cohort and the *SNC* study). We aimed at deriving an exposure metric to quantify the intermittence and put to test the working hypothesis mentioned above. In this paper we thus present a calculation method by which current noise exposure models can be extended with a metric that yields an integral description of the eventfulness (or intermittency) of noise exposure situations, taking into account both number and magnitude of noise events during a certain time period. The metric referred to as intermittency ratio (*IR*) can be derived either directly from acoustic measurements or calculated from traffic and geometry data for any transportation noise source and any time period (including day and night).

METHODS

In the next sections, a calculation method is described by which *IR* can be integrated in current noise exposure calculation models. The method has been implemented in *sonBASE*, the Swiss noise mapping database[32] and *IR* has been calculated for entire Switzerland. The result of this effort is presented in the Results section where noise maps are shown that incorporate the *IR* metric alongside L_{eq}.

IR Basic Principle

Highly intermittent traffic noise exposure situations consist of subsequent pass-bys of vehicles (cars, aircraft, trains and so on) which acoustically stand out from the background (noise) by a certain degree. We define such parts of the level-time course as "noise events". A noise event can be characterized by its maximum level, its sound exposure level, the emergence from background noise, its duration, or by the slope of rise of the level. For an integral characterization of the "eventfulness" of an exposure situation over a longer period of time we introduce the event-based sound pressure level $L_{eq,T,Events}$, which accounts for all sound energy contributions that exceed a given threshold, that is, clearly stand out from background noise. This event-based sound pressure level $L_{eq,T,Events}$ can now be compared with the overall sound pressure level $L_{eq,T,tot}$. The *IR* is defined as the ratio of the event-based sound energy to the overall sound energy.

Hence *IR* is defined as

$$IR \equiv \frac{10^{0.1L_{eq,T,Events}}}{10^{0.1L_{eq,T,tot}}} \cdot 100 = 10^{0.1\left(L_{eq,T,Events} - L_{eq,T,tot}\right)} \cdot 100(\%) \quad (1)$$

$L_{eq,T,tot}$ corresponds to the equivalent continuous sound pressure level of all sound sources involved and is given as

$$L_{eq,T,tot} \equiv 10 \cdot \log_{10}\left(\frac{1}{T}\int_0^T 10^{0.1L(t)}dt\right)[\mathrm{dB}] \quad (2)$$

where $L(t)$ is the continuous sound pressure level at the receiver position.

A single pass-by only contributes to $L_{eq,T,Events}$ if its level exceeds a given threshold K.

$$K \equiv L_{eq,T,tot} + C[\mathrm{dB}] \quad (3)$$

This threshold K is defined relative to the long-term average of the overall sound pressure level $L_{eq,T,tot}$ and an offset C. Thus $L_{eq,T,Events}$ is defined as:

$$L_{eq,T,Events} \equiv 10 \cdot \log_{10}\left(\frac{1}{T}\int_0^T H(L(t)-K)10^{0.1L(t)}dt\right)[\mathrm{dB}] \quad (4)$$

with the Heaviside step function H. The offset C is the only free parameter within the definition of *IR*. On the basis of practical experience on transportation noise situations, C might not be smaller than 0 and not larger than about 10 dB. For low values of C, almost any situation produces a large *IR*, whereas high values of C almost always produce low *IR*, as only in extraordinarily intermittent situations the level rises above the high threshold. This means that such choices of C cause greatly skewed distributions of *IR*. To be able for *IR* to distinguish between situations with different degrees of intermittency, the criterion for setting C was a preferably uniform spread of *IR* across the range of exposure situations as they occur in the real world. The balance between these extreme cases was investigated by numerical simulations of various traffic situations and resulted in $C = 3$ dB.

The integration time T has to be chosen in a way that the partial L_{eq}s do not significantly differ from the total L_{eq}. If this precondition is not fulfilled, for example, because of a highly varying traffic flow, it is recommended to determine partial Intermittency Rates and average them according to the equation given in Appendix section A7.

By definition, *IR* only takes values between 0 and 100% (including 0% and 100%). An *IR* of $>50\%$ means that more than half of the sound dose is caused by "distinct" pass-by events. In situations with only events that clearly emerge from background noise (e.g., a receiver point close by a railway track), *IR* yields values close to 100%. As, for example, Figure 2 shows a railway noise situation where the single events exhibit

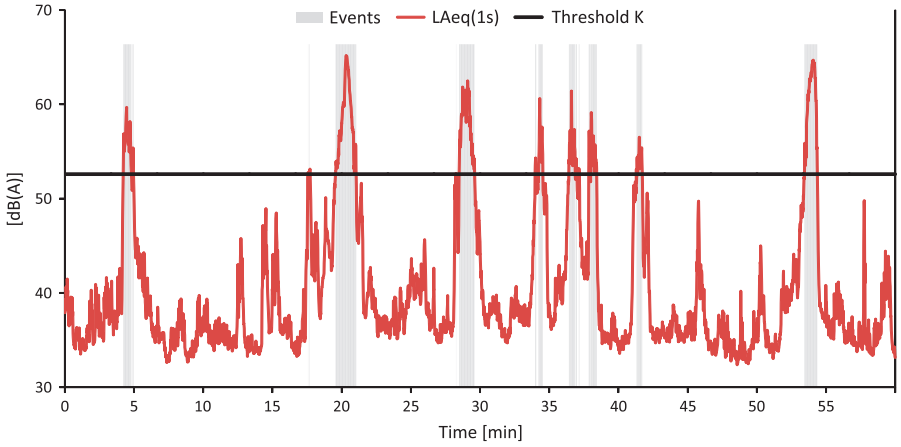

Figure 2. Railway noise recordings at 560 m distance from a freight train railway line during the night. Background noise is dominated by nature sounds and distant road traffic noise. Intermittency ratio (*IR*) = 87%. The portions of the curve marked "gray" are used to calculate $L_{eq,T,Events}$.

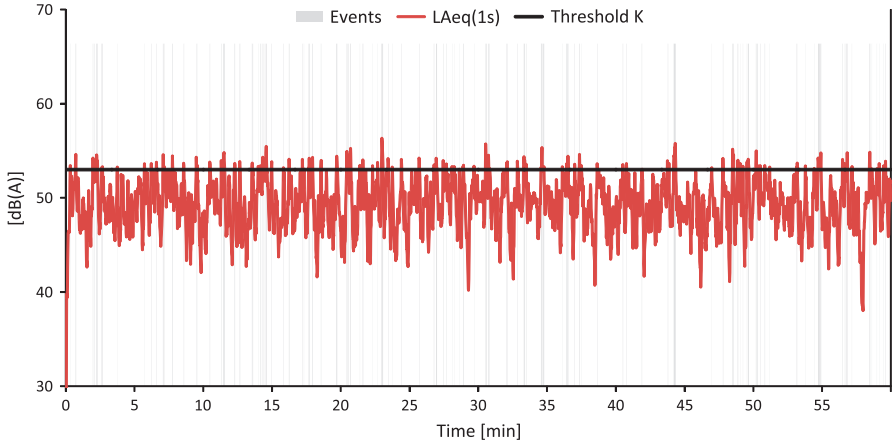

Figure 3. Road traffic noise at 100 m distance from a highway with a speed limit of 120 km/h. A total of 3200 vehicles per hour with 12% heavy traffic. Intermittency ratio (*IR*) = 19%. The portions of the curve marked "gray" are used to calculate $L_{eq,T,Events}$.

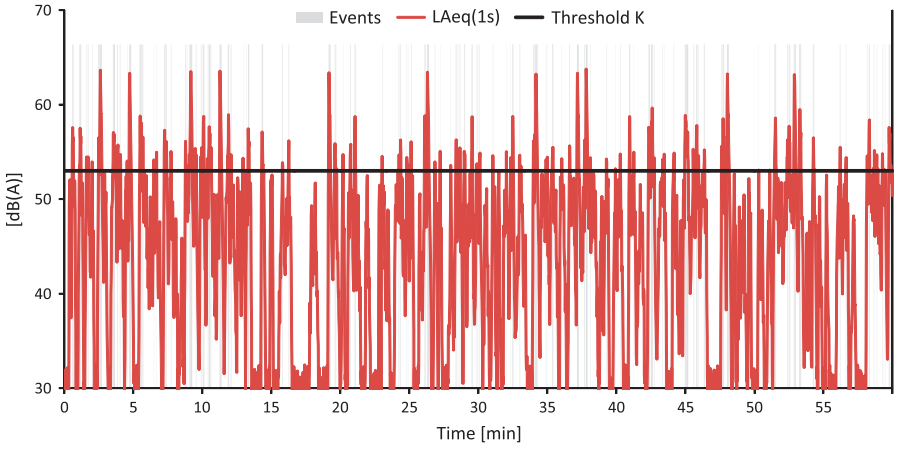

Figure 4. Road traffic noise at 100 m distance from a highway with a speed limit of 120 km/h. A total of 315 vehicles per hour with 12% heavy traffic. Intermittency ratio (*IR*) = 62%. The portions of the curve marked "gray" are used to calculate $L_{eq,T,Events}$.

FAST-weighted maximum levels which are about 20 dB(A) higher than background noise and a high *IR* of 87%. In contrast Figure 3 depicts a situation with intensive road traffic at 100 m distance. Here, the levels are rather constant over time and consequently, the *IR* is low with only 12%.

Air and railway traffic generally exhibit a high *IR*, with the exception of situations with such a high background noise (e.g., noise from other sources) that the events are partially or fully masked. The *IR* for road traffic is primarily determined by the number of pass-by events and by the distance between source and receiver. Figure 4 shows another road traffic situation with the same overall L_{eq} as Figure 3 but a shorter distance to the source and less vehicles per hour, resulting in a significantly higher *IR* of 62%.

Estimation of the *IR* by Calculation

In the examples shown above, the IR was derived based on measured level *vs* time data. However, with only a few exceptions (such as, for example, the aircraft noise model FLULA2[33], transportation noise calculation models that can be used for large-scale calculations such as CNOSSOS,[34] Harmonoise,[35] Nord2000[36] or Doc. 29[37] are designed in a way that only permits equivalent continuous sound pressure levels L_{eq} or derived quantities such as L_{den} as an output, but no information on level variation over time. Therefore, a direct calculation of the IR is not possible in these models. However *IR* can be estimated based on traffic information which is commonly at hand for noise assessment purposes. The corresponding procedure is illustrated in Figure 5.

The first step is to assess the maximum sound pressure level L_{Fmax} of single pass-by events. The estimation of maximum levels as given in equation (5) is based on the assumption that the propagation conditions for the entire source are also representative for the geometry that determines the maximum level. This might not hold in the case of strong changes of shielding effects, that is, if barriers abruptly end along the line source. Therefore in such situations it is recommended to split the source into different sections with uniform propagation conditions.

The maximum level of a single pass-by in a given source–receiver geometry can be written as

$$L_{Fmax} = L_{eq,T,single} + \log_{10}\left(\frac{vT}{D\Phi}\right) + \Delta L_{air}[\mathrm{dB}]. \quad (5)$$

Where D is the shortest distance of the source polygon to the receiver point, v denotes the speed, Φ is the source path aspect angle, that is, the angle that the source path subtends at the receiver, and ΔL_{air} denotes a correction for air absorption. A derivation of equation (5) can be found in Appendix section A1. Equation (5) is valid for point sources. In Appendix section A2 an expression for the maximum level of moving line sources with finite size is given, which should be used in case of trains.

As a next step, for each vehicle category a distribution of maximum levels is derived by adding a source level variance. It is assumed that the maximum levels of single pass-bys are normally distributed. For road traffic an additional variation in level is introduced to account for the temporary overlap of single pass-by events (see Appendix section A3 for details).

As an example, Figure 6 shows calculated maximum level distributions for the traffic situation presented in Figure 4. In Figure 6a sound pressure

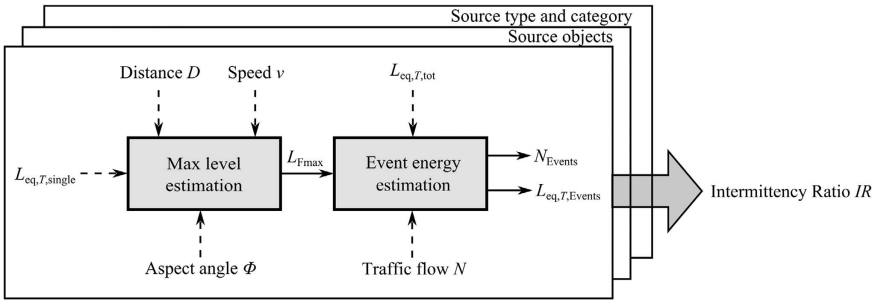

Figure 5. Scheme for the calculation of the intermittency ratio (*IR*).

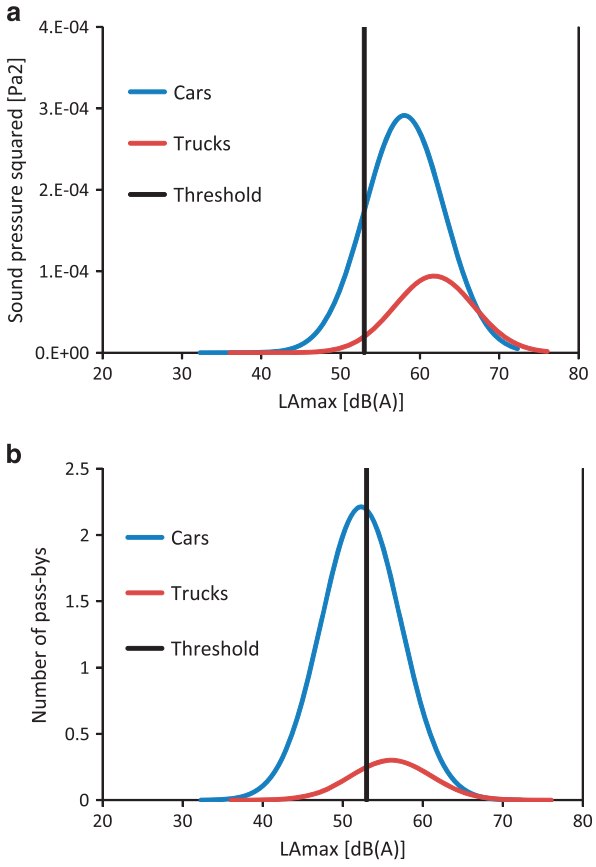

Figure 6. Calculated maximum level distributions for the traffic situation presented in Figure 4, **a**: sound pressure squared, **b**: number of events. Calculation details: 277 cars with 120 km/h and a single-event L_{eq} of 24.0 dB(A) and 38 trucks with 90 km/h and a single-event L_{eq} of 29.0 dB(A). $L_{eq,T,tot} = 50.0$ dB(A), $L_{eq,Events} = 47.8$ dB (A), $IR = 60\%$, $N_{Events} = 151$. Resolution of the *x*-axis: 0.1 dB.

squared of the maximum pass-by levels, and in Figure 6b the number of pass-by events is shown. The vertical black line indicates the threshold *K*. The number of pass-bys *N* above threshold is calculated according to Appendix section A4. *N* represents an estimate of the number of events. As can be seen, the majority of trucks produce maximum levels above the threshold, that is, noise events, but less than half of the passenger cars do so.

Looking at Figure 6a depicting sound intensity, one would assume a much higher *IR* than 60% as only a small part of the level distribution remains below the threshold. However, it has to be considered that only maximum levels are shown. Even for single pass-bys with maximum levels clearly above *K*, a significant part of *L*(*t*) remains below the threshold. This aspect is taken into account by introducing a weighting function as defined in Appendix section A6 that cuts off that part of an event which remains below the threshold.

As a last step, according to equation (1), *IR* is calculated for all traffic noise sources individually, as well as for the sum of them.

RESULTS

Including the *IR* in Noise Mapping

As part of the *SiRENE* study, *IR* was calculated for all dwellings in Switzerland (54,300,000 facade points, assigned to 1,813,000 buildings). As an example, Figures 7 and 8 depict noise maps that incorporate *IR* for an area north of the city of Zurich. Figure 7 shows the road traffic exposure situation during the day. The background color gradient of the map indicates the yearly average for road traffic noise L_{day} ranging from 30 to 70 dB(A). As can be seen, important noise sources are a highway/expressway that passes the map section in the lower right corner, and several major roads. In addition, there are numerous arterial roads with significantly less traffic. Values for *IR* have been calculated at all facade points. The facade points are shown as dots, reflecting the value of *IR* with different colors. Low values of *IR* can not only be found far away from roads but also in vicinity of the highway, despite the very different L_{eq}s. In contrast, high values of *IR* can be found close to smaller roads with thinned-out traffic where single pass-bys of vehicles clearly stand out from background noise. Generally, receiver points on the front-side of buildings exhibit a markedly higher *IR* than on the back-side, where noise is much more diffuse.

The influence of background noise on *IR* can be studied well in Figure 8. The same map section as above is used, but the image shows aircraft noise exposure. Disregarding background noise, aircraft noise is basically always "eventful", however, in presence of an elevated background noise (e.g., through road traffic), aircraft flyover events can be masked, resulting in a reduction of *IR*. In the map section shown in Figure 8, the 16-h-L_{eq} produced by aircraft is almost identical everywhere on the map, at a comparably low level of about 47 dB(A). At receiver positions with elevated road traffic noise, aircraft noise events only cause small *IR*. Elevated levels of *IR* for aircraft noise can only be found far away from major roads where the overall road traffic L_{eq} is low, such as for example, on the left side and on the right upper corner of Figure 8.

Correlation between L_{eq} and *IR*

As most of the common noise metrics like L_{day}, L_{den}, or L_{dn}, to name just a few, are usually highly correlated (e.g., for road traffic noise, *r* is clearly >0.9), the introduction of a complementary (new) metric only makes sense if it is basically uncorrelated with the L_{eq}, as was, for example, pointed out by the authors of the I-INCE report on supplemental noise metrics.[24] Table 1 lists the

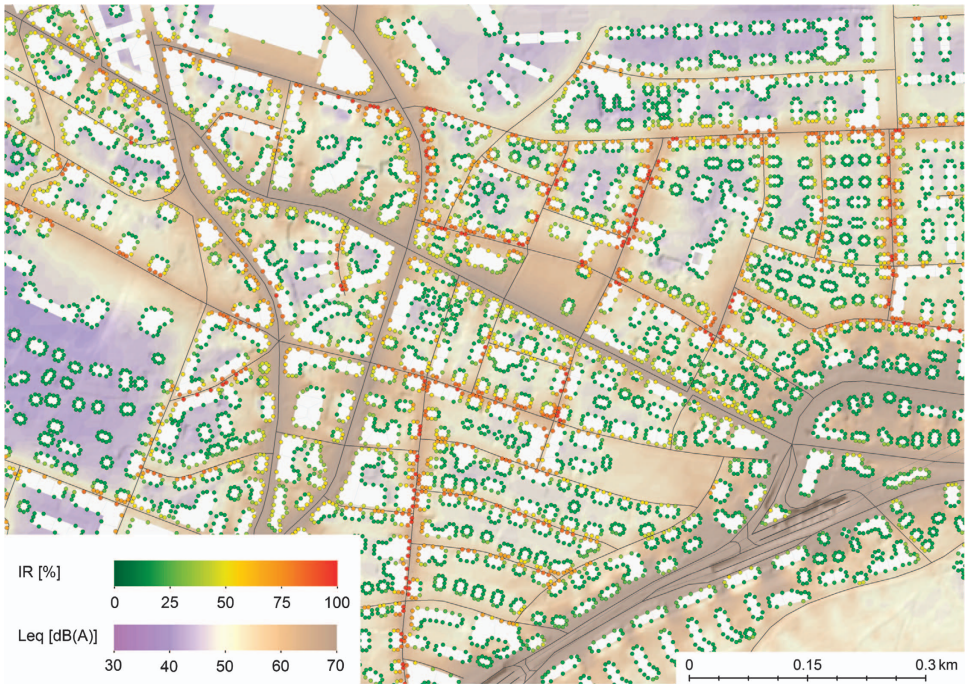

Figure 7. Road traffic noise map showing both L_{eq} and *IR* during daytime (16 h). Intermittency ratio (*IR*) has been calculated for individual facade points on the map, marked as small dots. The map section shows an area north of the city of Zurich in vicinity of Zurich airport.

Figure 8. Aircraft noise map showing both L_{eq} and *IR* during daytime (16 h). Intermittency ratio (*IR*) has been calculated for individual facade points on the map, marked as small dots. The map section shows an area north of the city of Zurich in vicinity of Zurich airport.

correlations between different conventional noise metrics and *IR* for the map section shown in Figures 7 and 8.

Compared with correlations between the most common metrics, the correlations between the L_{eq} and *IR* in the map section above are rather small. One could expect the correlation to be positive though, for quite obvious reasons. For example, facade points pointing towards a road are characterised by a markedly higher L_{eq} as those pointing to a rather calm backyard, where the distance to passing vehicles is longer and hence, both L_{eq} and *IR* are smaller. For aircraft noise, the L_{eq} during day is very similar for

Table 1. Pearson correlation coefficients (rounded to two decimals) between L_{eq}- and *IR*-based measures at facade points from the map section shown in Figures 7 and 8. ($N = 35{,}704$ facade points).

	$L_{day,\ air}$	$L_{night,\ air}$	$L_{day,\ rail}$	$L_{night,\ rail}$	$L_{day,\ road}$	$L_{night,\ road}$
IR day, air	0.28					
IR night, air		0.34				
IR day, rail			0.10			
IR night, rail				0.23		
IR day, road					0.46	
IR night, road						0.44

Abbreviation: IR, intermittancy ratio.

all facade points within the map section, but due to different masking effects from other sources, aircraft *IR* values nearly spawn the whole possible range of values, thus only correlating little with the L_{eq}.

DISCUSSION

Typically, studies on health and annoyance effects of noise rely on acoustic exposure metrics that are based on the concept of an average energetic dose over longer time periods (e.g., $L_{eq,24h}$, L_{dn}, L_{den}, L_{night} or L_{day}). However there are strong indications that these L_{eq}-based quantities are not appropriate to predict noise-induced sleep disturbances and also have their limitations with respect to satisfactorily explaining variance of annoyance or disturbance of activities. Potentially, other metrics can add to, improve, or replace the predictions made using L_{eq}-based measures. Therefore in this paper, we developed the metric *IR*. This metric allows an integral characterization of the eventfulness of a noise exposure situation, independent of the overall energetic dose. The decorrelation from the L_{eq} is an eminent feature of a complimentary noise metric that aims at explaining remaining variance in epidemiological studies, and in a practical example, we could demonstrate that the *IR* metric is essentially uncorrelated with energetic measures.

An advantage of the concept of *IR* is that it can be derived based on acoustic measurements, as well as, from calculations as it has been designed in a way which facilitates subsequent implementation into the most common traffic noise calculation models. The latter allows it to be used in large-scale epidemiological (population-based) studies and other types of observational investigations such as annoyance surveys. However, one has to be aware of the fact that in the procedures described, the estimation of maximum levels is based on several assumptions. An increase of the accuracy of the calculation of *IR* could be achieved by applying calculation models that directly yield maximum levels of single pass-bys—or even better—entire level *vs* time courses. First approaches for such models are at hand (for example see de Coensel et al.).[38]

As part of the calculation of *IR*, the decision whether an "event" accounts for $L_{eq,T,Events}$ (according to equation (4)) or not, is based on a simple criterion, the parameter *C*, for which a value of 3 dB has been assumed. This value has not been set based on any verified psychoacoustic principle but was derived empirically to attain a uniform spread of *IR* across the range of the different exposure situations. The question of how much an event really has to stand out from background noise in order to be termed "event" by normal listeners depends on various other parameters (which were not addressed in the present paper). In fact, for the noticeability of an event, not only the acoustic characteristics of the event compared to the background, but also the attentional, cognitive and emotional situation of the listener is relevant, as was described by de Coensel et al.[29]

In the near future, we expect to be able to increase the explanatory power of our epidemiological models by incorporating *IR* as a predictor alongside with average level measures within the aforementioned *SiRENE* study. The *IR* metric with its current parameter setting of *C* will be further validated using a self-report rating scale that measures the subjective perception of intermittency by residents affected by noise. In order to further investigate if *IR* is an appropriate measure to describe the relevant aspects of the temporal pattern of noise exposure, we would like to encourage others to implement the methodology presented here in their own models.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

ACKNOWLEDGEMENTS

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REFERENCES

1 Hanninen O, Knol AB, Jantunen M, Lim TA, Conrad A, Rappolder M *et al.* Environmental burden of disease in Europe: assessing nine risk factors in six countries. *Environ Health Perspect* 2014; **122**: 439–446.
2 Basner M, Babisch W, Davis A, Brink M, Clark C, Janssen S *et al.* Auditory and non-auditory effects of noise on health. Lancet. 2013; **383**: 1325–1332.
3 Ohrstrom E, Skanberg A, Svensson H, Gidlof-Gunnarsson A. Effects of road traffic noise and the benefit of access to quietness. *J Sound Vibr* 2006; **295**: 40–59.
4 Héritier H, Vienneau D, Frei P, Eze IC, Brink M, Probst-Hensch N *et al.* The association between road traffic noise exposure, annoyance and health-related quality of life (HRQOL). *Int J Environ Res Public Health* 2014; **11**: 12652–12667.
5 Basner M, Muller U, Elmenhorst EM. Single and combined effects of air, road, and rail traffic noise on sleep and recuperation. *Sleep* 2011; **34**: 11–23.
6 Brink M, Lercher P, Eisenmann A, Schierz C. Influence of slope of rise and event order of aircraft noise events on high resolution actimetry parameters. *Somnologie* 2008; **12**: 118–128.
7 Basner M, Griefahn B, Muller U, Plath G, Samel A. An ECG-based algorithm for the automatic identification of autonomic activations associated with cortical arousal. *Sleep* 2007; **30**: 1349–1361.
8 Basner M, Samel A, Isermann U. Aircraft noise effects on sleep: application of the results of a large polysomnographic field study. *J Acoust Soc Am* 2006; **119**: 2772–2784.
9 Brink M, Omlin S, Mueller C, Pieren R, Basner M. An event-related analysis of awakening reactions due to nocturnal church bell noise. *Sci Total Environ* 2011; **409**: 5210–5220.
10 Elmenhorst EM, Pennig S, Rolny V, Quehl J, Mueller U, Maass H *et al.* Examining nocturnal railway noise and aircraft noise in the field: sleep, psychomotor performance, and annoyance. *Sci Total Environ* 2012; **424**: 48–56.
11 Jarup L, Babisch W, Houthuijs D, Pershagen G, Katsouyanni K, Cadum E *et al.* Hypertension and exposure to noise near airports: the HYENA study. *Environ Health Persp* 2008; **116**: 329–333.
12 Muzet A. Environmental noise, sleep and health. *Sleep Med Rev* 2007; **11**: 135–142.
13 Dratva J, Phuleria HC, Foraster M, Gaspoz JM, Keidel D, Kunzli N *et al.* Transportation noise and blood pressure in a population-based sample of adults. *Environ Health Perspect* 2012; **120**: 50–55.
14 Hofman WF, Kumar A, Tulen HM. Cardiac reactivity to traffic noise during sleep in man. *J Sound Vibr* 1995; **179**: 577–589.
15 Directive EU Directive 2002/49/EC of the European Parliament and of the Council of 25 June 2002 relating to the assessment and management of environmental noise. *Off J Eur Communities* 2002; **18**: 12–25.
16 Tenailleau QM, Bernard N, Pujol S, Houot H, Joly D, Mauny F. Assessing residential exposure to urban noise using environmental models: does the size of the local living neighborhood matter?. *J Expos Sci Environ Epidemiol* 2015; **25**: 89–96.
17 Griefahn B, Marks A, Robens S. Noise emitted from road, rail and air traffic and their effects on sleep. *J Sound Vibr* 2006; **295**: 129–140.
18 Griefahn B, Schuemer-Kohrs A, Schuemer R, Moehler U, Mehnert P. Physiological, subjective, and behavioural responses to noise from rail and road traffic. *Noise Health* 2000; **3**: 59–71.

19 Basner M, Isermann U, Samel A. Die Ergebnisse der DLR-Studie und ihre Umsetzung in einer laermmedizinischen Beurteilung für ein Nachtschutzkonzept. *Zeitschrift fuer Laermbekaempfung* 2005; **52**: 109–123.
20 Miedema H, Oudshoorn C. Annoyance from transportation noise: relationships with exposure metrics DNL and DENL and their confidence intervals. *Environ Health Persp* 2001; **109**: 409–416.
21 Miedema H. *Response Functions For Environmental Noise In Residential Areas*. TNO Institute of Preventive Health Care: Leiden, The Netherlands, 1993.
22 Lercher P. A comparison of regional noise-annoyance-curves in alpine areas with the European standard curves. *In:* de Greve B, Botteldooren D, Rüdisser J (eds) 9th International Congress on Noise as a Public Health Problem (ICBEN). 2008. Foxwoods, CT, USA.
23 Lercher P, Botteldooren D. de Greve B, Dekoninck L, Ruedisser J (eds). The effects of noise from combined traffic sources on annoyance: the case of interactions between rail and road noise. *Internoise*. Istanbul, Turkey. 2007.
24 TSG9 Supplemental metrics for day/night average sound level and day/evening/night average sound level. *I-INCE Technical Study Group on Metrics for Environmental Noise Assessment and Control* 2014.
25 Commission E. *Position paper on EU noise indicators*. Belgium, 2000 ISBN 92-828-8953-X.
26 Langdon FJ, Scholes W. The traffic noise index: a method of controlling noise nuisance. *Architects' J* 1968; **147**: 20.
27 Robinson DW. The concept of noise pollution level. *J Occup Environ Med* 1971; **13**: 602.
28 Ribeiro C, Anselme C, Mietlicki F, Vincent B, Da Silva R, Gaudibert P(eds.) *At the heart of Harmonica project: the Common Noise Index (CNI) 6a Giornata di Studio sull'Acustica Ambientale* 2013 Genova, Italy.
29 De Coensel B, Botteldooren D, De Muer T, Berglund B, Nilsson ME, Lercher P. A model for the perception of environmental sound based on notice-events. *J Acoustic Soc Am* 2009; **126**: 656–665.
30 Bockstael A, De Coensel B, Lercher P, Botteldooren Deditors. *Influence of temporal structure of the sonic environment on annoyance 10th International Congress on Noise as a Public Health Problem (ICBEN)* 2011London, UK.
31 Heritier H, Vienneau D, Brink M, Cajochen C, Eze I, Karipidis I *et al.* (eds.) SiRENE: an interdisciplinary study on the health effects of transportation noise exposure: a study protocol. Swiss Public Health Conference; 2014 21–22 August 2014; Olten.
32 BAFU., SonBase: die GIS-Laermdatenbank der Schweiz Grundlagen Bern: Bundesamt fuer Umwelt; 2009. Available from www.umwelt-schweiz.ch/uw-0908-d.
33 Schaeffer B, Buetikofer R, Pluess S, Thomann G. Aircraft noise: accounting for changes in air traffic with time of day. *J Acoust Soc Am* 2011; **129**: 185–199.
34 Kephalopoulos S, Paviotti M, Anfosso-Ledee F, Common Noise Assessment Methods in Europe (CNOSSOS-EU) 2012 Contract No.: EUR 25379 EN.
35 Salomons E, van Maercke D, Defrance J, De Roo F. The harmonoise sound propagation model. *Acta Acustica United with Acustica* 2011; **97**: 62–74.
36 DELTA. Nord2000: Comprehensive Outdoor Sound Propagation Model. DELTA, Danish Electronics, Light and Acoustics 2001.
37 ECAC. *DOC29: Report on Standard Method of Computing Noise Contours around Civil Airports. Volume 2: Technical Guide*3rd edn.European Civil Aviation Conference (ECAC):Neuilly-sur-Seine, France, 2005.
38 de Coensel B, Brown AL, Modeling road traffic noise using distributions for vehicle sound power level. internoise 2012, New York; 19–22 August 2012.
39 Heutschi K. SonRoad: New Swiss road traffic noise model. *Acta Acustica United with Acustica* 2004 May–June **90**: 548–554.
40 Heutschi K. On single event measurements of heavy road vehicles in freely flowing traffic. *Acta Acustica United with Acustica* 2008; **94**: 709–714.
41 Poulikakos LD, Heutschi K, Soltic P. Heavy duty vehicles: impact on the environment and the path to green operation. *Environ Sci Policy* 2013; **33**: 154–161.
42 Heutschi K, Poulikakos LD(eds.)Noise monitoring of trucks*Euronoise* 2012Prague.
43 Heutschi K, Poulikakos LD, Soltic P(eds.) *Environmental impact of heavy vehicles based on noise, axle load and gaseous emissions Conference on Weigh In Motion ICWIM6*. Dallas, TX, USA, 2012.
44 Thron T, Hecht M. The sonRAIL emission model for railway noise in Switzerland. *Acta Acustica United with Acustica* 2010; **96**: 873–883.
45 Wunderli JM(ed)SONRAIL: From the scientific model to an application in practice*Euronoise*. Prague, 2012.
46 Schaeffer B, Zürcher Fluglärmindex ZFI, Berechnungsvorschrift (Zurich Aircraft Noise Index ZFI, Calculation Rule). Version 2 2006.
47 Schaeffer B, Zellmann C, Krebs W, Pluess S, Eggenschwiler K, Buetikofer R *et al.* Sound source data for aircraft noise calculations: state of the art and future challenges. *Euronoise* 2012 589–594.
48 Zellmann C, Wunderli JM, Schaeffer B. sonAIR: data acquisition for a next generation aircraft noise simulation model. *Internoise* 2013 Insbruck.
49 Brown AL, Tomerini D. Distribution of the noise level maxima from the pass-by of vehicles in urban road traffic streams. *Road Transport Res* 2011; **20**: 50–63.

A. APPENDIX

A.1 Estimation of the maximum level of a moving point source with constant speed

Taking geometrical spreading and air absorption into account, the effective value of sound pressure, p_{eff}, of a non-moving monopole point source at distance r can be given as

$$p_{\text{eff}}^2(r) = \frac{W\rho_0 c \cdot 10^{-0.1ar}}{4\pi r^2}. \tag{6}$$

with W the sound power, ρ_0 the density of air, c_0 the speed of sound in air and a the atmospheric attenuation coefficient in dB/m.

If the point source is moving with constant speed $v << c$ in m/s along a straight path at distance D from a receiver, the effective sound pressure can approximated by

$$p_{\text{eff}}^2(t) = \frac{1}{4\pi} \cdot \frac{W\rho_0 c \cdot 10^{-0.1a\sqrt{\left[v\cdot(t-t_p)\right]^2+D^2}}}{\left[v\cdot(t-t_p)\right]^2 + D^2}, \tag{7}$$

with t_p being the time when the maximum level is reached. Note that $t_p - D/c_0$ corresponds to the time when the source is at shortest distance to the receiver. The FAST-weighted maximum level L_{Fmax} is approximated by

$$L_{\text{Fmax}} \approx 10\cdot\log_{10}\left(\frac{p_{eff}^2(t=t_p)}{p_0^2}\right) \tag{8}$$

$$= 10\cdot\log_{10}\left(\frac{W\rho_0 c 10^{-0.1aD}}{4\pi D^2 p_0^2}\right)[\text{dB}]. \tag{9}$$

Applying the FAST-weighting to a signal emitted by a moving source, leads to a systematic underestimation of its actual level. The magnitude of underestimation depends on the speed v and extent of the source L as well as the shortest distance to the receiver D. In a highway situation with 120 km/h and a measurement distance of 7.5 m relative to the center of the lane the error amounts to approximately 0.5 dB for passenger cars.[39] For increasing distances or lower speeds the error decreases rapidly. Therefore we decided to neglect that effect in equation (8).

The equivalent continuous sound pressure level L_{eq} for an integration interval of $t_p - \tau$ to $t_p + \tau$ can be given as

$$\begin{aligned} L_{\text{eq},T,\text{single}} &= 10\cdot\log_{10}\left(\frac{1}{T}\int_{-\tau}^{\tau}\frac{p_{eff}^2(t)}{p_0^2}dt\right) \\ &= 10\cdot\log_{10}\left(\frac{W\rho_0 c}{4\pi T p_0^2}\int_{-\tau}^{\tau}\frac{10^{-0.1a\sqrt{v^2\cdot t^2+D^2}}}{v^2\cdot t^2+D^2}dt\right) \\ &= 10\cdot\log_{10}\left(\frac{W\rho_0 c}{4\pi T p_0^2}\cdot\frac{2}{D^2}\int_0^{\tau}\frac{10^{-0.1aD\sqrt{\left(\frac{vt}{D}\right)^2+1}}}{\left(\frac{vt}{D}\right)^2+1}dt\right) \end{aligned} \tag{10}$$

by substituting $vt/D = \tan\theta$ it can be simplified as

$$\begin{aligned} L_{\mathrm{eq},T,\mathrm{single}} &= 10\cdot\log_{10}\left(\frac{W\rho_0 c}{4\pi T p_0^2}\cdot\frac{2}{D^2}\int_0^{\Phi/2}\frac{10^{-0.1\alpha D\sqrt{1+\tan^2\theta}}}{1+\tan^2\theta}\cdot\frac{D/v}{\cos^2\theta}d\theta\right)\\ &= 10\cdot\log_{10}\left(\frac{W\rho_0 c}{4\pi T p_0^2}\cdot\frac{2}{Dv}\int_0^{\Phi/2}10^{-0.1\alpha D\sqrt{1+\tan^2\theta}}d\theta\right)\\ &= 10\cdot\log_{10}\left(\frac{W\rho_0 c}{4\pi T p_0^2}\cdot\frac{2}{Dv}\int_0^{\Phi/2}10^{-0.1\alpha D\frac{1}{\cos\theta}}d\theta\right). \end{aligned} \tag{11}$$

with Φ as the angle that the source path subtends at the receiver. For a pass-by of infinte length $\Phi = \pi$. By neglecting air absorption ($\alpha = 0$) follows

$$\begin{aligned} L_{\mathrm{eq},T,\mathrm{single}} &= 10\cdot\log_{10}\left(\frac{W\rho_0 c}{4\pi T p_0^2}\cdot\frac{2}{Dv}\cdot 1\Big|_{\theta=0}^{\theta=\Phi/2}\right)\\ &= 10\cdot\log_{10}\left(\frac{W\rho_0 c}{4\pi T p_0^2}\cdot\frac{\Phi}{Dv}\right)[\mathrm{dB}] \end{aligned} \tag{12}$$

Subtracting (9) from (12) an expression for the maximum level of a constantly moving point source in dependence of the L_{eq} can be found

$$L_{\mathrm{Fmax}} = L_{\mathrm{eq},T,\mathrm{single}} + 10\cdot\log_{10}\left(\frac{vT}{D\Phi}\right)[\mathrm{dB}] \tag{13}$$

where

$$\Delta L = 10\cdot\log_{10}\left(\frac{vT}{D\Phi}\right)[\mathrm{dB}]. \tag{14}$$

Subtracting equation (11) from equation (9) it can be shown that the influence of air absorption on ΔL is independent from the speed v, but dependent on the distance and the source aspect angle Φ. Assuming a pass-by of infinite length for the influence of air absorption, ΔL can be approximated as

$$\Delta L \approx 10\cdot\log_{10}\left(\frac{vT}{D\Phi}\right) + \Delta L_{\mathrm{air}}(\alpha D)[\mathrm{dB}]. \tag{15}$$

ΔL_{air} can be given as

$$\begin{aligned} \Delta L_{\mathrm{air}} &= 10\cdot\log_{10}\left(\frac{10^{-0.1\alpha D}}{\frac{2}{\pi}\int_0^{\pi/2}10^{-0.1\alpha D\frac{1}{\cos\vartheta}}d\theta}\right)\\ &= -10\cdot\log_{10}\left(\frac{2}{\pi}\int_0^{\pi/2}10^{0.1\alpha D\left(1-\frac{1}{\cos\vartheta}\right)}d\theta\right)[\mathrm{dB}] \end{aligned} \tag{16}$$

ΔL_{air} is always positive and increases with increasing air absorption αD. For $\alpha D = 1$ dB equation (16) yields $\Delta L_{\mathrm{air}} = 1$ dB, with $\alpha D = 3$ dB follows $\Delta L_{\mathrm{air}} = 2$ dB.

By substituting $\theta = \frac{\pi}{4}\vartheta + \frac{\pi}{4}$ and hence adapting the integration limits the above expression can be efficiently numerically solved

$$\begin{aligned} \Delta L_{\mathrm{air}} &= -10\cdot\log_{10}\left(\frac{2\pi}{\pi 4}\int_{-1}^{1}10^{0.1\alpha D\left(1-\frac{1}{\cos\left(\frac{\pi}{4}\vartheta+\frac{\pi}{4}\right)}\right)}d\vartheta\right)\\ &= -10\cdot\log_{10}\left(\frac{1}{2}\int_{-1}^{1}k(\vartheta)d\vartheta\right) \approx -10\cdot\log_{10}\left(\frac{1}{2}\sum_{i=1}^{n}w_i k(\vartheta_i)\right)[\mathrm{dB}] \end{aligned} \tag{17}$$

Applying a Gauss-Legendre integration of third order ($n = 3$)

$$\begin{aligned} \Delta L_{\mathrm{air}} &\approx -10\cdot\log_{10}\left(\frac{5}{18}k\left(-\sqrt{3/5}\right)+\frac{8}{18}k(0)+\frac{5}{18}k\left(\sqrt{3/5}\right)\right)\\ &\approx -10\cdot\log_{10}\left(\frac{5}{18}10^{0.0159\cdot\lambda}+\frac{8}{18}10^{0.4142\cdot\lambda}+\frac{5}{18}10^{4.6783\cdot\lambda}\right) \end{aligned} \tag{18}$$

with $\lambda = -0.1\alpha D$, the error remains below 0.1 dB for $\alpha D < 7$ dB.

Although the atmospheric attenuation coefficient α is highly frequency dependent, in order to keep the computational cost low, a representative value per noise source type is proposed. Based on the analysis of representative source spectra it is recommended to set $\alpha = 5$ dB/km for road and air traffic.

A.2 Estimation of the maximum level of a moving line source with constant speed

The estimation for single pass-by L_{Fmax} as given in Appendix section A1 is based on the assumption of a moving point source, which can be used for cars and planes, but not for trains. In this section an extension is presented to derive L_{Fmax} for moving line sources of finite length L. It is assumed that the sound power W is equally distributed over an incoherently radiating line source.

The maximum level is reached for the geometrical situation in which the center of the line source is at shortest distance D to the receiver. Integrating over the length of the line source, the sound pressure can be given as

$$p_{\mathrm{eff}}^2(t=t_p) = \int_{-L/2}^{L/2}\frac{\rho_0 cW\cdot 10^{-0.1\alpha\sqrt{s^2+D^2}}}{4\pi L}\frac{}{s^2+D^2}ds \tag{19}$$

Neglecting air absorption ($\alpha = 0$) it can be deduced

$$p_{\mathrm{eff}}^2(t=t_p) = \frac{\rho_0 c}{4\pi}\cdot\frac{2W}{LD^2}\int_0^{L/2}\frac{1}{\left(\frac{s}{D}\right)^2+1}ds. \tag{20}$$

Applying the substitution $s/D = \tan\theta$ it follows

$$\begin{aligned} p_{\mathrm{eff}}^2(t=t_p) &= \frac{\rho_0 c}{4\pi}\cdot\frac{2W}{LD^2}\int_0^{\arctan(L/(2D))}\frac{1}{\tan^2\theta+1}\cdot\frac{D}{\cos^2\theta}d\theta\\ &= \frac{\rho_0 c}{4\pi}\cdot\frac{2W}{LD}\int_0^{\arctan(L/(2D))}1d\theta = \frac{\rho_0 cW\Theta}{4\pi LD} \end{aligned} \tag{21}$$

with $\Theta = 2\arctan(L/(2D))$ as the opening angle of the source. The maximum level of a moving line source can then be given as

$$L_{\mathrm{Fmax}} \approx 10\cdot\log_{10}\left(\frac{\rho_0 cW\Theta}{4\pi LDp_0^2}\right)[\mathrm{dB}]. \tag{22}$$

Neglecting air absorption the L_{eq} of a single pass-by remains as for point sources, see equation (12). ΔL is derived by subtracting (12) from (22):

$$\Delta L = 10\cdot\log_{10}\left(\frac{vT\Theta}{L\Phi}\right) + \Delta L_{\mathrm{air}}[\mathrm{dB}] \tag{23}$$

with

$$\Theta = 2\arctan\frac{L}{2D}. \tag{24}$$

Thereby $\Phi >> \Theta$, meaning that the whole source element, i.e., for example the entire railway track, must be clearly longer than the moving line source, i.e., the train. For $L \ll 2D$, Θ can be approximated by $\frac{L}{D}$, in which case equation (23) takes on the behavior of a point source according to equation (15).

For moving line sources the influence of air absorption is generally slightly smaller than for point sources. Nevertheless the

expression for ΔL_{air} given in equation (18) can be used as first approximation.

As railway noise sources generally exhibit more high frequent components than road and air traffic it is recommended to set a = 7 dB/km for railway traffic.

A.3 Maximum level distribution

It is assumed that the maximum levels of single pass-bys are normally distributed for a given vehicle category and only slightly varying speed.

$$h(L_{max}) = \frac{1}{\sqrt{2\pi\sigma^2}} e^{-\frac{\left(L_{Fmax}-\bar{L}_{Fmax}\right)^2}{2\sigma^2}} \quad (25)$$

with the standard deviation σ in dB and the arithmetic mean $\bar{L}_{max}$. However, sound levels obtained from noise calculations usually represent an energetic mean (here denoted L^E) over a certain time. Given the above assumption, the following relation between the arithmetic and the energetic mean can be deduced

$$\bar{L}_{Fmax} \approx L^E_{Fmax} - 0.115\sigma^2 \quad (26)$$

In real situations, the sound pressure signals of single pass-bys overlap temporarily. By the temporal overlap, the total L_{eq}, $L_{eq,T,tot}$, and—by definition—the threshold K are not affected. However, the intensity above the threshold ($L_{eq,Events}$) increases and thus the Intermittence Ratio. The influence of the overlap effect on *IR* in the case of road traffic noise was investigated by numerical simulations. Measurement data of periodically and stochastically distributed pass-by times were synthesized as the two marginal cases. It could be shown that particularly intermittency rates between 0 and 0.5 are affected. It was found that this effect could be approximated by an artificial increase of the standard deviation of the level by

$$\sigma = \sqrt{\sigma_0^2 + \sigma_{OL}^2} \quad (27)$$

with σ_0 being the single source standard deviation and σ_{OL} an additional standard deviation for the temporal overlap. At maximal overlap the simulations yielded $\sigma_{OL} \approx 4.5$ dB, at minimal overlap however $\sigma_{OL} \approx 2$ dB. As a compromise between both marginal cases σ_{OL} = 3 dB is adopted for road traffic noise. For railway and air traffic noise the temporal overlap is neglected, i.e. σ_{OL} = 0 dB.

The single source standard deviation reflects the variation of the sound power level within a vehicle category at a given speed. For the current application of *IR* within the project *SiRENE* it was concluded that a representative standard deviation σ_0 = 2 dB can be used for all vehicle categories such as passenger cars, heavy and light trucks, different types of trains and aircraft and that the assumption of a normal distribution was valid. For road noise the above conclusions were taken based on measurement data of road traffic monitoring stations collected in the project *Footprint*[40–43] based on a classification system with 10 categories (Swiss10). For railway noise measurement data from the project *sonRAIL* was analysed[44,45] and for aircraft noise we relied on the emission database of *FLULA2*[33,46] and measurements from the *sonAIR* project.[47,48] However it has to be kept in mind that this setting is depending on the local (national) composition of the vehicle fleet and its categorization and does not account for significant variations in the travelling speed such as congestions. For other countries a wider spread might be appropriate, see for example.[49] Therefore the setting of these standard deviations has to be checked and if necessary adjusted to the local conditions.

A.4 Number of events

The number of events exceeding the threshold K can be calculated by

$$N_{Events} = N \cdot \int_K^\infty h(L_{Fmax}) dL_{Fmax} = N \cdot \frac{1}{2}\left[1 + \mathrm{erf}\left(\frac{\bar{L}_{Fmax} - K}{\sqrt{2\sigma^2}}\right)\right] \quad (28)$$

with N the traffic flow intensity, i.e. the number of pass-bys within time T, and with the Gauss error function erf().

A.5 Event-*Leq*

The Event-*Leq* $L_{eq,T,Events}$ is calculated by

$$L_{eq,T,Events} = 10 \cdot \log_{10}\left(N \cdot \int_K^\infty h(L_{Fmax}) w(L_{Fmax}) 10^{0.1 L_{Fmax}} dL_{Fmax}\right) - \Delta L[\mathrm{dB}] \quad (29)$$

where w denotes a weighting function and

$$\Delta L = L_{Fmax} - L_{eq,T,single}[\mathrm{dB}] \quad (30)$$

Equation (29) has to be evaluated for different source types (regarding the parameters $L_{eq,T,single}$, D, v, Φ) separately. On the basis of the partial Event-*Leq*s $L_{eq,T,Events,i}$ the total *IR* is calculated by

$$IR = \frac{\sum_i 10^{0.1 L_{eq,T,Events,i}}}{10^{0.1 L_{eq,T,tot}}} = \sum_i IR_i \quad (31)$$

In doing so, the temporal overlap between different source types is neglected which underestimates the *IR*. As pass-bys of aircraft and trains generally only last a small percentage of time this effect is of minor importance.

A.6 Weighting function

The weighting function $w(x)$ considers the truncation effect due to the finite line source length and makes sure that for a given event only the intensity above the threshold K is integrated. $w(x)$ is formulated as

$$w(x) = f(x,) \cdot g(x, aD), \quad (32)$$

with $x = L_{Fmax} - K$. f corresponds to the analytical solution of a line source neglecting air absorption and g is an empiric correction for the influence of the air absorption. It can be shown that

$$f(x, \Phi) = \frac{2\arccos\left(10^{-0.05x}\right)}{\Phi} \overset{!}{\leq} 1 \quad (33)$$

The influence of the air absorption is approximated under the assumption of an infinitely extended source (i.e. $\Phi = \pi$). Investigations by numerical simulations yielded the empirical model

$$g(x, aD) \approx (1 - b \cdot e^{-cx})^{-1} \quad (34)$$

with parameters b and c for which the following linear regressions were obtained

$$b = 0.1597 \cdot \log_{10}\left(\frac{aD}{1dB}\right) + 0.1887 \quad (35)$$

$$c = 0.0243 \cdot \log_{10}\left(\frac{aD}{1dB}\right) + 0.0596 \quad (36)$$

A.7 Averaging of intermittency rates over different time periods

Averaging of *IR* over *M* time periods *k* is defined as

$$\overline{IR} \equiv \frac{\sum_{k=1}^{M} IR^k 10^{0.1L^k_{eq,T,tot}}}{\sum_{k=1}^{M} 10^{0.1L^k_{eq,T,tot}}}. \quad (37)$$

The Intermittency Ratio *IR* of time period *k* is weighted with the total *Leq* during that period. Hence time periods with higher *Leq* get a higher weight on *IR* than periods with lower *Leq*.

The averaging has to be performed as the final step, after the summation of *IR* over different sound sources as defined in Appendix section A5.