Indicators of residential traffic exposure: Modelled NO\textsubscript{X}, traffic proximity, and self-reported exposure in RHINE III

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**Highlights**

- The association between self-reported and modelled traffic exposure was studied in low-pollution Northern European cities.
- Self-reported exposure measures were residential traffic exposure and traffic noise exposure.
- Modelled exposure measures were NO\textsubscript{X} dispersion and proximity to large roads.
- Self-reported traffic exposure and NO\textsubscript{X} exposure and traffic proximity had no or weak correlations.
- Traffic noise exposure had higher agreement with modelled exposure than self-reported traffic exposure in most locations.

**Abstract**

Few studies have investigated associations between self-reported and modelled exposure to traffic pollution. The objective of this study was to examine correlations between self-reported traffic exposure and modelled (a) NO\textsubscript{X} and (b) traffic proximity in seven different northern European cities: Aarhus (Denmark), Bergen (Norway), Gothenburg, Umeå, and Uppsala (Sweden), Reykjavik (Iceland), and Tartu (Estonia). We analysed data from the RHINE III (Respiratory Health in Northern Europe, www.rhine.nu) cohorts of the seven study cities. Traffic proximity (distance to the nearest road with >10,000 vehicles per day) was calculated and vehicle exhaust (NO\textsubscript{X}) was modelled using dispersion models and land-use regression (LUR) data from 2011. Participants were asked a question about self-reported traffic intensity near bedroom window and another about traffic noise exposure at the residence. The data were analysed using rank correlation (Kendall's tau) and inter-rater agreement (Cohen's Kappa) between tertiles of modelled NO\textsubscript{X} and traffic proximity tertile and traffic proximity categories (0–150 metres (m), 150–200 m, >300 m) in each centre. Data on variables of interest were available for 50–99% of study participants per each cohort. Mean modelled NO\textsubscript{X} levels were between 6.5 and 16.0 \textmu g/m\textsuperscript{3}; median traffic...
1. Introduction

Different indicators of traffic-related air pollution and noise exposure have been used in epidemiological studies of the long-term air pollution and noise health effects (latently reviewed by WHO, 2013; Recio et al., 2016). Earlier studies often used ecological comparisons between regions to ascertain health effects (e.g. Ferris and Anderson, 1962; Lunn et al., 1970; Rudnik, 1977; Dockery et al., 1993), whereas later studies have used measured community-wide air pollution levels (e.g. Filleul et al., 2005; Beelen et al., 2008; Krewski et al., 2009; Heinrich et al., 2013; Dehbi et al., 2017). As air pollution monitoring networks are often scarce, different proximity air pollution modelling, interpolation, kriging, and similar approximation techniques have been applied.

Proximity models are the most basic approach to measure the proximity (often the distance) of a study participant’s residence to a pollution source (often a busy road). Geographical Information Systems (GIS) have been widely employed, often to estimate exposure in epidemiological studies (e.g. Brauer et al., 2003; Andersson et al., 2011; Pindus et al., 2015). Among more enhanced models, air pollution dispersion modelling has been frequently applied (e.g. Raaschou-Nielsen et al., 2012; Cesaroni et al., 2013). These models use a deterministic approach to describe physical and chemical processes than affect air pollution and incorporate e.g., traffic data, pollution point sources, and meteorological data to estimate pollution concentrations in grid cells or streets. Dispersion modelling has been combined with personal and regional monitoring of traffic-related pollutants such as NOX, or with regional, urban, and local (street canyon) models in hybrid models which may perform better (Zou et al., 2009). Instead of dispersion models, some recent large epidemiological studies, such as ESCAPE, have applied land-use regression models (LUR), which use measurements of pollution as dependent variables and land-use, traffic, demographic, and geographic characteristics as predictor variables (Beelen et al., 2013). In general, correlations between LUR and dispersion models have been relatively good (De Hoogh et al., 2014). Nevertheless, several epidemiological studies have also used self-reported exposure variables. Self-reported traffic exposure in terms of perceived traffic density close to one’s home address, the presence of many large vehicles (heavy traffic), or traffic congestion (traffic jams), have been associated with several respiratory health outcomes, such as asthma symptoms (Brunekreef et al., 2009; Vlaski et al., 2014), wheezing, rhinitis, and coughing (Kuehni et al., 2006), allergic respiratory complaints (Shirinde et al., 2015), and quality of sleep (Gislason et al., 2016). Noise exposure is associated with cardiovascular outcomes (WHO, 2011) and low birth weigh (Ristovska et al., 2014). Doubts remain regarding the validity of self-reported exposure measures, as they are prone to bias; sensitive individuals might over-report exposure (Oiano et al., 2015; Persson et al., 2007) and personal factors confound the association (Riedel et al., 2014). Several studies have also used annoyance as an indicator of traffic exposure; however, due to high variance between areas, it is not always considered to be the best proxy of air pollution exposure (Jacquemin et al., 2008).

Nevertheless, the agreement between air pollution- and noise annoyance ratings has been shown to be good (Shepherd et al., 2016), and both air pollution and noise annoyance have been related to NOX exposure (Fernández-Somoano et al., 2015).

Few studies have investigated agreement between measured/ modelled and self-reported exposure, and different correlations have been reported (e.g. Cesaroni et al., 2008; Heinrich et al., 2005). In the Respiratory Health in Northern Europe cohort (RHINE, 2017, www.rhine.nu), self-reported air pollution exposure measurements, and GIS and dispersion models of NOX have been shown to be effective markers of local traffic pollution (Madsen et al., 2007). The aim of the current study was to investigate potential correlations between self-reported traffic exposure and both modelled NOX exposure and traffic proximity (distance to a busy road) in the seven RHINE centres, mostly mid-sized cities with low or moderate air pollution levels.

2. Material and methods

2.1. Study population

The study population is from RHINE (www.rhine.nu), with a cohort from seven Northern European cities; Aarhus (Denmark), Bergen (Norway), Gothenburg, Umeå and Uppsala (Sweden), Reykjavik (Iceland), and Tartu (Estonia). These cities all participated in the European Community Respiratory Health Survey (ECRHS, www.ecrhs.org) over 1989–1992, and the study cohorts are described in detail elsewhere (Johannessen et al., 2014). The participating centres in the ECRHS sent out a screening questionnaire to a large random sample of adults born in 1945–1973 (25,000 individuals) and living in the study area. Approximately 10 years later all participants who had answered the screening questionnaire in the Northern European cities were invited to do the RHINE follow-up study questionnaire. RHINE III, the second follow-up survey, was undertaken over 2010–2012 (see Johannessen et al., 2014).

2.2. Self-reported exposure

Self-reported residential traffic exposure was measured using the following two questions in RHINE III:

- “Is your bedroom window towards a nearby street (<20m)?”
  - Possible replies constituted four levels: “No”; “Yes, a street with a little traffic”; “Yes, a street with a moderate level of traffic”; “Yes, a street with a lot of traffic”
• “Can you hear traffic noise in your bedroom?”
  ○ Possible replies constituted four levels: “Not at all”; “A little”; “Much”, “Very much.”

2.3. Assignment and geocoding of addresses

The study participants’ address at the time of the 2nd follow-up survey were obtained from population registries. In Aarhus, addresses were retrieved from the Central Population Registry (CPR) and geocoded using the Danish national address database. In Bergen, addresses were retrieved from the National Registry (Folkeregisteret) and geocoordinates obtained from Statistics Norway. For participants in Umeå and Uppsala, Statistics Sweden (SCB) retrieved home addresses from the population register using National Individual Registration numbers; addresses were matched with GIS coordinates by Lantrådet – the Swedish National Land Survey – for Umeå and Uppsala, and by SCB for Gothenburg. In Reykjavik, participants were sampled from Registers Iceland (Þjóðskrá) and address geocoordinates retrieved from Statistics Iceland. In Tartu, the RHINE III questionnaire was sent out according to each participant’s address in the Estonian Population Register 2011. In addition, the questionnaire asked participants about their actual address, which led to several of the addresses in the Tartu RHINE III cohort being corrected.

2.4. NOX exposure dispersion modelling

In each centre the annual average concentrations of NOX were modelled using data collected in 2009–2011, depending on the centre (Supplementary Table A). All models had high resolution (40–150 m) and used Gaussian dispersion distribution; however, several enhancements were applied. Later the air quality grid cell values were then matched with each participant’s residential geocoordinates for the year of participation.

For Aarhus, the Danish AirGIS exposure modelling system (Jensen et al., 2001, 2009b) was applied based on GIS and air quality models. The system calculates air quality at a street location as the sum of three contributions: (a) local air pollution from street traffic, calculated using the Operational Street Pollution Model (OSPM) from input data on traffic (intensity, vehicle types, travel speed), emission factors for the vehicle fleet based on COPERT emission model, street and building geometry, background air pollution concentrations, and meteorology. The OSPM takes into account the street canyon effect, and photochemistry, to estimate NO2 (Berkowicz, 2000); (b) urban background concentrations, calculated from a simplified area source dispersion formula that takes into account urban vehicle emission density, city dimensions (transport distance), and building heights (initial dispersion height) (Berkowicz et al., 2008); and (c) regional background concentrations, estimated from trends at rural monitoring stations and national vehicle emissions (Jensen, 1998). Input data for the AirGIS system were established from various sources and integrated into the dispersion model. A Danish national GIS road network with traffic data and building footprints including building heights. The system has been applied in numerous epidemiological air pollution studies in Denmark (e.g. Raaschou-Nielsen et al., 2012; Sørensen et al., 2012) and validated in several studies (e.g. Berkowicz et al., 2008; Jensen et al., 2009b; Kakkosimos et al., 2010; Ketzell et al., 2011). For Bergen and Reykjavik, NOx dispersion was described using Gaussian dispersion modelling in the air quality management system AirViro (SMHI, 2015). Bottom-up emission inventories had been compiled based on information acquired from the municipalities. The main road network, as well as emissions from ships, were included. The road and ship emission inventory is based on information from the local municipalities and port statistics from MARINE Traffic and EU Stat. Due to a lack of detailed local information regarding the composition of the vehicle fleet and driving conditions, it was assumed that these conditions were similar to the situation in Uppsala, which allowed the same emission factors to be applied. As in Reykjavik and Bergen, there were no large point sources present within the modelled domain in Uppsala, thus the contribution from point sources was disregarded. Validation of the spatial variations was carried out in both Bergen and Reykjavik, comparing modelled results with available measurements from campaigns with passive samplers. Also it was the first time the model was used in these cities.

For Gothenburg, annual means of NOX — provided by the Environment Administration of the City of Gothenburg — were modelled. The NOx models were made using both historical and current emission databases (EDBs), and calculations done using the Enviman AQPlanner (OPSIS AB, Furulund, Sweden) that consists of a Gaussian model AERMOD (US EPA). There were approximately 6700 sources in the EDBs, most of which were road traffic (line) sources (approximately 5900) and shipping (line) sources (approximately 100). Industry and larger energy and heat producers accounted for approximately 500 point sources; small-scale heating and construction machinery emissions were considered as area sources. The model results were validated with data from air quality monitoring stations in Gothenburg (Molnár et al., 2015).

For Tartu, the Gaussian plume model AEROPOL 5 that enabled the use of point, line, and area sources, taking into account both dry and wet deposition, was applied. Traffic emissions data were based on annual traffic flow measurements and modelling ordered by Tartu City Government, and the CAR-FMI emission coefficients (Karpinnen et al., 2000; Härkönen et al., 2001). Domestic heating emissions and the heating patterns were obtained from earlier studies (e.g. Kaasik et al., 2007); industrial emissions were obtained from a registry of industrial point sources. Rush-hour traffic was measured in 2011 during the evening (16:30–17:30) at 33 sites; at the remaining 910 road segments, traffic modelling using CUBE software was performed. The AEROPOL model has previously been applied to air quality modelling of Tartu for other modelling studies and exercises (e.g. Orru et al., 2008). We then applied the porosity concept of Genikhovich et al. (2002) during post-processing of modelled ground-level concentrations: the area under buildings was excluded from the dispersion volume of each grid cell, thus the concentration was divided to the fraction of porosity, i.e. the proportion of non-built-up area (Kaasik et al., 2014). The street canyon effect was not applied, as continuous building fronts higher than or nearly equal to the width of the street seldom occur in Tartu. The computed annual average concentrations of NO2 were validated against passive sampling results available since 1997 (two-week measuring campaigns, four times a year) and data of an urban monitoring station available since 2007. As the AEROPOL model does not include chemical transformation, NO2 concentrations were derived from modelled NOx concentrations based on the monitoring ratios of Estonian urban areas. Each respondent’s
address was geocoded in ArcGIS, and the centre of each respondent’s house linked to the corresponding grid cell. The concentration in grid cell was used as the exposure level of all respondents in that cell.

For Umeå city and Uppsala, dispersion modelling was performed using the Gaussian model included in the Airviro air quality management system. In both cases, emission factors were determined from the Artemis database of the Swedish Transport Administration (Boultet et al., 2007). The Artemis model includes scenarios of the composition of various types of vehicles and fuels, such as the share of diesel cars, as well as the composition of the vehicle fleet in terms of European emission standards (Euro classification) for different years. For Umeå, other emissions were obtained from the SIAAIR 2005 (based on SMED emissions). For Uppsala, only emissions from road traffic were included in the current study, which was the dominant source of NOx in the region. The modelled annual mean concentrations were validated using NO2 passive sampling results available for a large part of Västerbotten county, which was developed for the ESCAPE study (Cyrys et al., 2012; Beelen et al., 2013). As the Umeå cohort was sampled from the south-eastern part of Västerbotten county a large number of the participants lived outside the area covered by the dispersion model, and were assigned NO2 exposure levels from the LUR model. Land-use regression utilized NOx concentrations as the dependent variable and variables such as traffic, topography, and other geographic variables as independent variables in a multivariate regression model (Gilliland et al., 2005). Since the population density in the county varied considerably, the results were adjusted according to population density.

2.5. Traffic proximity modelling

First, large roads were defined. Large roads constituted those with traffic of >10,000 vehicles per day, except in Umeå, where they constituted those with traffic of >8000 vehicles per day due to no road having more than 10,000 vehicles per day at the time of the study. Traffic intensity (vehicles per day) was either measured in large number of receptor points or modelled based on small number of receptor points (see details in Supplementary Table A). Subsequently in all centres the large roads were mapped in GIS, and distances from the centre of nearest large road to the centre of each participant’s residential geocoordinates as centre of the building calculated.

2.6. Statistical methods

Individuals who had answered basic demographic questions, self-reported exposure items and modelled exposure were included in the study. Within each study centre modelled NOx values were divided into tertiles based on frequency distributions using the 33rd and 67th percentiles (tertiles: lowest, medium, and highest values). Traffic proximity was recalculated into categories based on tertiles (closest, medium, and furthest) and categories (cut-offs: <150 m, 150–300 m, and >300 m). A sensitivity analysis was performed with exposure distribution cutoffs at 50th and 90th percentile. The relationship between the different exposure metrics were first analysed using Kendall’s τ (tau) rank correlation. Rank correlations are used to study bivariate non-linear associations. Kendall’s tau (B) is a non-parametric statistical test of concordances and discordances of pairs of values adjusting for ties. It takes values from −1 to 1 corresponding to “perfect” concordance and discordance, and values near 0 indicate no significant concordance. Kendall’s tau is preferable in cases of a large sample size with many values of the same score (Newson, 2002). Agreement between the self-reported and modelled exposure metrics were compared using Cohen’s Kappa inter-rater agreement test with quadratic weights (Halgren, 2012; Banerjee et al., 1999). The test assesses agreement between alternative methods of categorical assessment, and is interpreted as having poor agreement if the score is <0.2; fair for a score of 0.21–0.40; and very good for a score of 0.81–1.00. Sensitivity analysis was performed for rural areas in Umeå, the only centre where part of the study population was rural. All statistics were performed using R (R Core Development Team, 2015) using the “irr” package (Gamer and Lemon, 2012).

3. Results

3.1. Descriptive statistics

A total of 13,550 individuals participated in RHINE III. Of those, 10,708 (79%) had information about the variables of interest. The most common causes of dropout were that the address could not be geocoded or was outside the dispersion model area. In the individual centres participation varied between 50% and 99%; women were in the majority (53%, range 50%–60% among the centres), and the mean age was 52 years old (range 50–54 years old) (Table 1). Average modelled NOx was 9.8 μg/m³ ranging from 6.5 in Uppsala to 16.0 μg/m³ in Gothenburg. Within-centre range NOx exposure also varied highly: a small range in Aarhus of 8.9–16.1 μg/m³ and a high range in Bergen of 2.1–63.8 μg/m³. Median distance to the nearest large road was from 303 to 10,750 m (cohort median 435 m); maximum distance varied from 2170 to 331,000 m (Table 2).

Self-reported traffic exposure was reported as low (no street within 20 m of participants’ bedroom window) by 56.3% of participants (range 36.8–65.9% among the individual centres). High traffic exposure (moderate or high traffic intensity within 20 m of participants’ bedroom window) was reported by an average of 11.0% of the total cohort (range 7.7–18.7%). Traffic noise exposure was reported to be low (no traffic noise heard from the bedroom) by 45.7% (range 19.2–58.6%) of participants, and high by 6.8% (range 3.6–16.3%) of participants. The highest proportion of participants to report high traffic intensity and traffic noise occurred in Tartu. The lowest proportion of participants to report high traffic exposure in terms of proximity occurred in Aarhus; the lowest proportion of participants to report high traffic noise exposure occurred in Uppsala (Fig. 1).

<table>
<thead>
<tr>
<th>RHINE</th>
<th>Inclusion ratea</th>
<th>%</th>
<th>Female (%)</th>
<th>Mean age (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>RHINE 10,708/13,550 79</td>
<td>53</td>
<td>52</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Aarhus 2269/2299 99</td>
<td>53</td>
<td>50</td>
<td></td>
<td></td>
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<tr>
<td>Bergen 1737/2264 73</td>
<td>50</td>
<td>52</td>
<td></td>
<td></td>
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<tr>
<td>Gothenburg 848/1712 50</td>
<td>53</td>
<td>54</td>
<td></td>
<td></td>
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<tr>
<td>Reykjavik 1681/1942 87</td>
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<td>Tartu 863/1396 63</td>
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<td>50</td>
<td></td>
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<tr>
<td>Umeå 1610/1934 83</td>
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<td>54</td>
<td></td>
<td></td>
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<tr>
<td>Uppsala 1700/1933 88</td>
<td>51</td>
<td>53</td>
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</table>

* Proportion of RHINE III participants who reported the variables of interest in the questionnaires.

Table 1 Characteristics of the study participants.
3.2. Correlations

Self-reported exposure to traffic near the bedroom window had low correlation (Kendall's tau) with modelled NOx (0.161; \( p < 0.05 \)) with modelled NOx and measured traffic proximity (distance to nearest busy road) in the whole cohort; and also in each centre with tau ranging from 0.056 to 0.305 (all \( p < 0.05 \)) - lowest correlation was found in Reykjavik and highest in Uppsala (Table 3). Traffic proximity tertiles and categories (<150 m; 150–300 m; >300 m) correlated weakly with self-reported exposure; correlation coefficients were lower than for modelled NOx at 0.134 and 0.130, respectively (\( p < 0.05 \) for both). Of all correlations, self-reported traffic exposure and modelled NOx was highest in five of seven centres; the exceptions were Bergen and Reykjavik, where traffic proximity tertiles and traffic proximity categories correlated better with NOx, respectively. Self-reported traffic noise levels from each

<table>
<thead>
<tr>
<th>NOx (( \mu g/m^3 ))</th>
<th>Traffic proximity (m)*</th>
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<tbody>
<tr>
<td>min</td>
<td>33rd %ile</td>
</tr>
<tr>
<td>RHINE III</td>
<td>0.5</td>
</tr>
<tr>
<td>Aarhus</td>
<td>8.8</td>
</tr>
<tr>
<td>Bergen</td>
<td>2.1</td>
</tr>
<tr>
<td>Gothenburg</td>
<td>4.1</td>
</tr>
<tr>
<td>Reykjavik</td>
<td>0.5</td>
</tr>
<tr>
<td>Tartu</td>
<td>2.8</td>
</tr>
<tr>
<td>Uméa*</td>
<td>2.8</td>
</tr>
<tr>
<td>Uppsala</td>
<td>3.3</td>
</tr>
</tbody>
</table>

* Large roads had traffic of >10,000 vehicles per day, except in Uméa, where the threshold was 8000.

![Can you hear traffic noise in your bedroom?](image)

![Is your bedroom window towards a nearby street (<20 m)?](image)

Fig. 1. Percentage distribution of self-reported exposure categories.
participant’s bedroom window correlated weakly with modelled NOx and traffic proximity tertiles and categories, among the whole cohort with tauds of 0.157, 0.133, and 0.150, respectively (p < 0.05 for all); these correlations were highest in Uppsala. No exposure metric correlated significantly with self-reported noise exposure in Reykjavík; traffic proximity tertiles did not correlate with self-reported noise exposure in Umeå (0.032; p > 0.05) (Fig. 2, Table 3). Self-reported traffic noise exposure had the highest correlation with NOx in Tartu and Umeå, with traffic proximity tertiles in Bergen and Uppsala, and with traffic proximity categories in Aarhus and Gothenburg. Comparing the correlation coefficients of both subjective outcomes among the centres, traffic noise correlated highest with traffic proximity tertiles (in Bergen) and modelled NOx (in Tartu). The correlation between the two self-reported measures was moderate (Supplementary Table C), and a reanalysis with percentiles cut-off at 50th and 90th percentile showed very similar results (Supplementary Table D). There was no association between modelled NOx levels and the correlation between subjective and objective measures (Supplementary figure A).

### 3.3. Inter-rater agreement test

Agreement between self-reported traffic exposure and the objective exposure metrics among the whole cohort was 0.107, 0.162 and 0.180 for NOx, and traffic proximity tertiles and categories, respectively (Table 4). Agreements varied between 0.069 and 0.274 (Reykjavík and Uppsala); all agreements were statistically significant (p < 0.05). Comparing agreements between self-reported traffic exposure and the objective exposure metrics, NOx had the highest agreement in four of seven centres; Gothenburg, Tartu, Umeå, and Uppsala; traffic proximity tertiles and categories correlated best with self-reported traffic exposure in Bergen and Aarhus, respectively.: NOx and traffic proximity tertiles correlated joint-highest self-reported traffic exposure in Reykjavík.

Agreement between self-reported noise exposure and the objective exposure metrics among the whole cohort was 0.156, 0.148, and 0.168 for NOx, and traffic proximity tertiles and categories, respectively. In Reykjavík there were no statistically significant agreements; traffic proximity tertiles did not significantly agree with self-reported noise for Umeå (Supplementary Table B). Comparing agreements between self-reported noise and the different objective exposures, traffic proximity in categories had the best agreement in four of seven centres (Aarhus, Bergen, Gothenburg, and Uppsala); modelled NOx had better agreement with self-reported noise in Tartu and Umeå (Supplementary Table B). Comparing Cohen’s Kappa values for all outcomes across all centres, the best agreements occurred between self-reported noise exposure and traffic proximity tertiles in Aarhus and Bergen, and between the former and NOx in Tartu.

### 4. Discussion

In the current study, we found weak or no statistically although significant correlations and agreements between self-reported residential traffic exposure and modelled traffic pollution and GIS-derived traffic proximity in seven study centres in Northern European cities, a very diverse settings in terms of an investigation of this type. Self-reported traffic, and traffic noise exposure were reported as none or moderate by the majority of study participants, and reports of high exposure varied in frequency between the centres. The range of modelled NOx varied among the study cities, reflecting in part choices made by the modellers due to differences in the available raw data and different approaches, but also different urban environment characteristics, such as the degree of urbanisation and traffic density, the latter being reflected in the variability in traffic proximity. Modelled NOx correlated better than traffic proximity with self-reported traffic exposure except in Reykjavík and Bergen, where traffic proximity tertiles and category correlated better with self-reported exposure, respectively. However, in terms of traffic noise exposure, traffic proximity measurement correlated best, except in Tartu and Umeå. In our study, agreement and correlations of self-reported traffic exposure and self-reported traffic noise exposure to measured variables were similar, although traffic exposure tended to correlate better with modelled NOx exposure than did noise exposure.

### 4.1. Correlation and agreement between self-reported and modelled exposure in other studies

Previous studies of self-reported traffic exposure and modelled outcomes have been set in less diverse settings, e.g. Cesaroni et al. (2008) found that Rome residents’ self-reported traffic exposure levels were significantly correlated with objectives measures from r = 0.32–0.49. However, exposure (mean NOx was 45 μg/m³) was higher than in the current study, where the mean of NOx was 9.8 μg/m³. It is possible that self-reported exposure correlate better with objective data in areas with larger pollution gradients. Heinrich et al. (2005) compared rates of self-reported high traffic exposure within categories of modelled NOx exposure, and in general found rather low-agreements, although they tended to be higher in urban compared to rural settings. In the current study, correlations and agreements between urban and rural areas were tested with Umeå data, the only centre where part of the study population was rural. We found no significant difference between

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Table 3

<table>
<thead>
<tr>
<th>Traffic near bedroom</th>
<th>Traffic noise</th>
</tr>
</thead>
<tbody>
<tr>
<td>NOx (tertiles) Traffic proximity (tertiles) Traffic proximity (categories)</td>
<td>NOx (tertiles) Traffic proximity (tertiles) Traffic proximity (categories)</td>
</tr>
<tr>
<td>RHINE</td>
<td>Aarhus</td>
</tr>
<tr>
<td>0.161*</td>
<td>0.134*</td>
</tr>
<tr>
<td>Traffic noise (tertiles) Traffic proximity (categories)</td>
<td></td>
</tr>
<tr>
<td>0.125*</td>
<td>0.122*</td>
</tr>
</tbody>
</table>

* “Is your bedroom window near a street (less than 20 m)?”

* “Can you hear traffic noise from your bedroom?”

* Highest tertile, lowest exposure.

* Categories: 0–150 m; 150–300 m; >300 m.

* Weighted mean. *p < 0.05.
Fig. 2. Modeled concentrations of NO\textsubscript{X} within self-reported exposure categories: (a) residential traffic exposure (Is your bedroom window towards a nearby street (<20 m)?) and (b) traffic noise exposure (Can you hear traffic noise in your bedroom?).
urban and rural areas per se. In a study of modelled NO\textsubscript{2} and self-reported odour annoyance, Oiamo et al. (2015) found significant correlations (Spearman’s rho of 0.22), but the correlation between self-reported noise and modelled NO\textsubscript{2} was not significant (Spearman’s rho of 0.22). Correlations and agreements between self-reported and modelled NO\textsubscript{2} was not significant (Spearman’s rho of 0.22), but the correlation between self-reported noise and modelled NO\textsubscript{2} was not significant (Spearman’s rho of 0.22).

4.2. Factors affecting dispersion modelling

Dispersion models provide objective estimates of exposure to large roads or NO\textsubscript{X}, and are becoming standard in modelling exposure. However, models are only as good as the raw data upon which they are based, and while traffic sources were effectively characterized for most of our study centres, the databases varied in age, therefore interpolation was used to compensate for this. The contribution of NO\textsubscript{X} from other pollution sources, such as shipping and industrial sources, were considered in some centres meaning that self-reported traffic or traffic noise exposure underestimate actual exposure, and would provide a lower estimate than modelled NO\textsubscript{X} when there are high contributions from marine traffic or industry. As eventual marine traffic and industry sources were not well characterized in all studied cities, such contributions may have contributed to differences between study centres. In some models, coarser grid resolution was applied outside of urban centres, so individuals who lived outside Uppsala city centre could have high exposure (as they in fact lived in Stockholm) which was less accurately assigned. Also different dispersion models are heterogeneous which might cause regional heterogeneity across the included centres.

The original cohort was selected from urban centres (except Umeå), areas covered by dispersion models (DMs), so that participants were lost during follow-up if they had moved outside the modelled area. This is a potential source of bias, as those who move and those who stay are likely to be different in a number of ways, including lifestyles, respiratory health, and susceptibility to bad air quality (Jie et al., 2013). In Umeå, the original participants were a mixture of urban and rural dwellers, and LUR was used in non-urban areas that the DM did not cover. In a validation study of the Umeå LUR and DM, the two models were highly correlated in terms of NO\textsubscript{2} (Spearman’s rho of r = 0.78; De Hoogh et al., 2014), and although NO\textsubscript{X} was not tested, it would likely also be highly correlated. Correlations and agreements between self-reported and modelled traffic exposure were lowest in Reykjavik. The NO\textsubscript{X} dispersion models of Bergen and Reykjavik were developed during this study, therefore these models had a smaller degree of validation than the other—older—models.

4.3. Factors affecting self-reported exposure

Our results indicate that in some study settings, self-reported traffic noise exposure was a weakly correlated proxy of measured traffic exposure variables. This may have been because it entailed exposure to traffic regardless of factors not covered by the models, such as the floor level of the building or building characteristics, topography, or tunnels. In the current study, self-reporting of exposure was done using two questions, one for proximity of traffic to and the other for traffic noise exposure in their bedroom. Similarly, Heinrich et al. (2005) compared objective and subjective exposure in two centres, however, more questions were used. In one centre three questions were used to assess traffic exposure at one’s home address, including questions about traffic jams and truck traffic on weekdays. In the other centre road type was assessed (low corresponding to side streets with 30 km/h speed limits) and additional questions were asked about traffic intensity and traffic jams. In the study of Persson et al. (2007), ten items were assessed on a six-point scale; in contrast, Cesaroni et al. (2008) used...
only one self-reported question of traffic exposure. Traffic and traffic noise exposure are related, but not exactly the same; however, few studies have studied their correlation and found it moderate (Fecht et al., 2016). In the current study, the two subjective measurements had a moderate level correlation, but it is noteworthy that Reykjavik, which had the lowest correlation and agreement between subjective and objective exposure metrics, also had the lowest correlation between the two self-reported exposure metrics of traffic proximity and noise exposure (Supplementary Table C). It is therefore possible that other local factors, for example landscapes features, building types, and or characteristics of the car fleet, contributed to the poor correlations of that city.

Whereas modelled traffic proximity and NO exhaust provide objective measures of exposure, self-reported exposure has the advantage of reporting exposure where the person actually lives, and is more sensitive to issues that models do not always account for, such as noise exposure, self-reported exposure has the advantage of anxiety levels and indoor noise reporting (Persson et al., 2007). In addition, subjective perception both on a within-region, and within-city level (e.g. relatively speaking busy street in a middle-sized Northern Europe city is very different from a busy street in an international megapolis), and sensitivities has been shown to be affected by character (anxiety levels) and indoor noise reporting (Persson et al. 2007). In addition, self-reported exposure might be affected by health status, e.g. in a study by Heinrich et al. (2005), individuals with hay fever or asthma clearly over-reported traffic exposure in Germany—but not in Dutch—urban areas. But subjective exposure might be still very important in the case of low air pollution levels as a recent study in Estonia (Ortu et al., 2017) showed that perceived pollution and health risks might play an even more essential role in predicting environmentally induced symptoms and diseases than actual exposure. Understanding such modification effects would help the interpretation of future field studies.

5. Conclusion

We observed significant but weak or no correlations between subjective and objective pollution exposure metrics in most cases in a multi-centre cohort study in low in low- or moderate air pollution exposure settings. Previous health effects studies from this cohort have utilized self-reported air pollution and noise pollution measures. Thus, the availability of the current study data offers an opportunity to compare health outcome associations between those metrics, although their correlation and agreement is low. In our study setting within-city exposure ranges were larger than between-city ranges, which is a motivation to conduct multi-centre studies to study air pollution effects in different study settings.

Acknowledgements

- Air pollution modelling was largely financed by the FAS (Swedish Council for Working Life and Social Research) grant 2010–0442. This work was also supported by the NordicWelfAir project funded by NordForsk (project number: 75007). The RHINE Study has over the years received funding from many research foundations. HO’s and MK’s work on the preparation of this article was supported by the Estonian Ministry of Education and Research grants IUT34–17 and IUT20–11, respectively. The authors would also like to acknowledge Thomas Becker and Matthias Ketzel from Aarhus University, and Christian Askler and Mattias Jakobsson from the Swedish Meteorological and Hydrological Institute for their contributions.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.atmosenv.2017.08.015.

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