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(道路交通騒音による健康リスク：
リスクコミュニケーションのための健康リスクマップ作成
および電気自動車への移行による健康リスクの低減)

2020年12月

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  (Journal of Japan Society of Civil Engineers, Ser. G (Environmental Research) Vol. 76, No. 5, I_441-I_449, 2020)

- Mapping of transportation noise-induced health risks as an alternative tool for risk communication with local residents
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  (The 28th Japan Society of Civil Engineers Symposium)

Abstract

Environmental noise is a threat to public health. It may cause a myriad of health effects, varying from sleep disturbance to severe outcomes, such as hypertension, ischaemic heart disease, stroke, and diabetes. A health impact estimation of 32 European countries showed that more than 10,000 premature deaths per year are caused by environmental noise exposure.

In an attempt to mitigate environmental noise, the European Union requires the member states to produce noise maps for estimating noise exposure and developing noise mitigation measures to address the noise issues (Directive 2002/49/EC). Thus far, the noise maps, which are geospatial visualisations of sound levels, have been created in cities and around major noise sources in Europe. Decision-makers would be able to estimate the health impacts of noise exposure and make use of them in policymaking.

Nevertheless, estimations of sound level are not useful in communicating risks with the general public because a noise map shows acoustic intensity instead of health effects. Moreover, recent social changes in achieving a low carbon society would transform the acoustic environment, i.e. the transition to electric vehicles. However, the health risk reduction impact of the transition remains unclear.

Thereby, the objectives of the present study were (1) to develop health risk maps from noise maps as an alternative tool for public risk communication; and (2) to analyse the reduction of health risks in a future noise-exposure setting by the transition to electric vehicles.

In both studies, I investigated the health risks due to road traffic noise, which is the major source of environmental noise. As health outcomes, high annoyance, high sleep disturbance, and ischaemic heart disease were selected because their exposure-response functions with the sound levels are shown in the Environmental Noise Guidelines for the European Region issued by the World Health Organization Regional Office for Europe. This guidelines are based on scientific evidence from large-scale epidemiological studies and socio-acoustic surveys. The numerical calculations on the health risks in Sapporo City, Japan were also carried out.

In the first study, I created health risk maps as an alternative tool for enhancing risk communication. The health risk maps were derived from the sound levels and exposure-response relationships with health outcomes. To demonstrate it, I calculated the sound levels using geospatial data of Sapporo City and the common methodological framework for noise mapping in the European Union, i.e. CNOSSOS-EU. In addition, the number of population exposed to road traffic noise and health risks in Sapporo City was estimated by employing the exposure-response functions and the national health statistics and surveys.

The health risk maps visualise the distribution of health risks instead of acoustic intensity. For instance, a percentage of people highly sleep-disturbed of 6.0% was estimated instead of 55 dB of night equivalent sound level. Ten deaths out of 100,000 people due to ischaemic heart disease were estimated instead of 64 dB of day-evening-night equivalent sound level. The health risks were unevenly distributed but are relatively high in areas located in close proximity to the roads while they were negligible in other areas. By using the health risk maps, the general public will be able to realise the significance of the health impacts of noise exposure.

The estimated number of people highly annoyed and highly sleep-disturbed in Sapporo City was 100,773 and 44,674, respectively, in 1.91 million population. In respect to ischaemic heart disease, the estimated number of patients and yearly deaths were 257 and 49, respectively.

To summarise, in the first study, I demonstrated the feasibility of health risk mapping to identify and assess health risks for effective public health risk communication. Contrary to the ordinary noise maps, the health risk maps directly show the quantitative risks in a graphical format and will enable the general public to comprehend the significance of the health risks due to noise exposure. The methods are not limited to road traffic noise and health risks as defined in this study, but are also applicable to other types of traffic noise and health risks; provided that the dose-response relationships are available. The health risk maps would contribute to the knowledge sharing with local communities and raising public awareness; thus, effective for public health risk communication. A significant proportion of people are at risk and efforts to mitigate health risks are necessary.

In the second study, I investigated the health risks reduction by the transition to electric vehicles (EVs) from the internal combustion engine vehicles (ICEVs). Firstly, I examined the relationship between the reduction of the health effects and traffic conditions factors, i.e. the percentage of heavy vehicles and traffic speed, based on CNOSSOS-EU and the exposure-response functions. Given that EVs have no internal combustion engine, the driveline noise was assumed as negligible. To validate the calculation results, I selected two urban areas in Sapporo City with different traffic conditions; high/low percentage of heavy vehicles and low/high traffic speed. The reduction of health risks due to the transition to EVs was estimated. The methods used are identical to the methods in the first study. Additionally, the total mitigation in Sapporo City by the introduction of EVs was also carried out.

The calculation results showed that the higher the percentage of heavy vehicles, and the lower the traffic speed, the more effective the health risks reduction would be. For example, the health risk reduction for highly annoyed at 70 dB of day-evening-night equivalent sound level is 55.0% when traffic speed of 30 km/h and 30% of heavy vehicles in the fleet were assumed; and 66.3% when traffic speed of 20 km/h and 50% of heavy vehicles were assumed.

The results in the two areas are consistent with the calculations, which show that health risk reduction has more impact with higher percentages of heavy vehicles and lower traffic speeds. The area with higher percentages of heavy vehicles and lower traffic speeds contributed to higher risk reductions (30–50%) compared to the area with the opposite traffic conditions (10–30%). Meanwhile, the estimated health risk reduction in the total agglomeration of Sapporo City was approximately 20%.

To conclude, in the second study, I analysed the reduction of health risks by the transition to EVs and reveal the effective traffic conditions. The calculations showed that health risk reduction largely depends on the proportion of heavy vehicles and traffic speed. Higher percentages of heavy vehicles and lower traffic speeds would contribute to an effective health risk reduction.

This thesis presents approaches that would contribute to managing health risks due to road traffic noise. Health risk maps provide considerable insights into health risk conditions and are effective as an alternative tool for risk communication. Several ten percent of health risk reductions were estimated with the shift to EVs; therefore, the widespreading of EVs would be a transformative means in mitigating the health effects of road traffic noise. Hopefully, these contributions will hence serve a role in our eventual transition towards achieving zero noise pollution and a healthier living in the future. Further research is needed to assess the effectiveness of the health risk maps in actual settings and to identify additional factors that could further enhance the health risk reduction by the transition to EVs.

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List of Abbreviations

ASJ RTN-Model	Acoustical Society of Japan Road Traffic Noise Model
BEV	battery electric vehicle
BMI	body mass index
CNG	compressed natural gas
CNOSSOS-EU	Common Noise Assessment Method in Europe
CVD	cardiovascular disease
DALY	disability-adjusted life year
DIS	difficulty initiating sleep
DMS	difficulty maintaining sleep
DNL	day-night level
DRM	Digital Road Map
DW	disability weight
EEA	European Environment Agency
END	Environment Union Directive
ERF	exposure-response function
EU	European Union
EUR-A	WHO epidemiological subregion in Europe
EV	electric vehicle
GIS	geographic information systems

GRADE	Grading of Recommendation Assessment, Development and Evaluation
HEV	hybrid eletric vehicle
ICEV	internal combustion engine vehicle
ICSD	International Classification of Sleep Disorders
HA	high annoyance
HSD	high sleep disturbance
%HSD	percentage of the population "highly sleep-disturbed"
%HA	percentage of population "highly annoyed"
IHD	ischaemic heart disease
NIHL	noise-induced hearing loss
OR	odds risk
PHEV	plug-in hybrid vehicle
PSG	polysomnography
SEL	sound exposure level
YLD	years lived with disability
YLL	years of life lost
RR	relative risk
SWS	slow-wave sleep
WHO	World Health Organization
WHO-EU	World Health Organization Regional Office for Europe
L_{day}	day equivalent sound level
L_{den}	day-evening-night equivalent sound level
L_{evening}	evening equivalent sound level
L_{night}	night time equivalent sound level

Chapter 1

Introduction

1.1 Background

Environmental noise is a growing problem and many people may not be aware of its impacts on their health. It might be tempting to think that environmental noise is not a serious health issue. After all, it has been traditionally dismissed as a natural part of everyday life. This could not be further from the truth. The mounting evidences linking noise to adverse health effects, coupled with proactive legislation, primarily in the European Union (EU), are now driving change (Murphy, E. and King, E. A., 2014).

Clear links between excessive exposure to environmental noise and adverse health effects have been well established. Exposure to prolonged or excessive noise can cause a multitudinous of health effects including annoyance and sleep disturbance, and more serious issues, such as the negative effects on the cardiovascular and metabolic systems, and cognitive impairment in children. The World Health Organization Regional Office for Europe (WHO-EU) has identified the key health outcomes associated with environmental noise based on the available evidences and specific outcomes and also public concern over the health outcomes resulting from noise exposure in its most recent document on environment and health, i.e. the *Environmental Noise Guidelines for the European Region* (WHO-EU, 2018).

Environmental noise, particularly road traffic noise, remains a major environmental problem that is affecting the health and well-being of millions of people in Europe. The latest report on environmental noise in Europe (European Environment Agency, 2020), published early this year, indicates that 20% of Europe's population is exposed to long-term noise levels that are harmful to human health. The percentage corresponds to more than 100 million people. It was also reported that environmental noise contributed to 48,000 new cases of ischaemic heart disease (IHD) a year and 12,000 premature deaths.

In addition, it was estimated that 22 million people suffered from chronic high annoyance (HA) and 6.5 million people suffered from chronic high sleep disturbance (HSD). The number of population exposed to environmental noise is projected to increase due to urban growth and increased mobility demand (European Environment Agency, 2020).

Environmental noise is now recognised as a public health issue that must be addressed in the modern society. Realising the harmful effects of environmental noise on health, the European Commission issued the European Environmental Noise Directive (Directive 2002/49/EC) as a policy instrument to assess and manage environmental noise. The Directive instructs the member states to develop strategic noise maps for all major roads, railways, airports and large agglomerations on a five-year basis.

A noise map is a means of presenting calculated and/or measured noise levels in a representative manner over a particular geographic area (Murphy and King, 2010). It quantifies and visualises noise pollution levels, thus enabling identification of locations that are subject to excessive noise levels. Subsequently, noise action plans can be carried out to protect public health.

However, noise maps are incomprehensible for the non-expert and local communities in communicating risks. This is because noise maps only show sound levels or acoustic intensity instead of the health effects. Communicating risks is important to educate the public so that they can understand the dangers of environmental noise. The biggest threat associated with environmental noise is that the extent of the problem is not recognised; or if environmental noise is even an issue. Hence, there is a need to educate the general public about the harmful effects of environmental noise on health. An involved and informed public is important for developing and enhancing public and political support in the implementation of noise mitigation measures.

Therefore, an alternative tool for risk communication that can contribute to knowledge sharing with the general public and raising public awareness is crucial. The health risks should be presented and quantified in a manner that is easy for the public to understand. One of the effective risk communication tools is graphical material, and a map, particularly, is a potentially powerful means of conveying spatial information (Dransch, D. et al., 2010; Stieb et al., 2019). The usefulness of a map in risk communication is acknowledged in many fields, including in environmental and health science (Dransch, D. et al., 2010).

Moreover, recent development in establishing a low carbon society through the proliferation of electric vehicles (EVs) might transform the acoustic environment. There will be a growing fleet of EVs in the near future, particularly due to the 'EV Initiative' that aims to accelerate the adoption of EVs worldwide; and with some countries setting the target to stop the sales of fossil-fuelled vehicles, e.g. Norway in 2025 and Japan in 2050.

An EV is known as the 'silent' vehicle due to the absence of an internal combustion engine. It has an electric motor, hence, has minimal propulsion noise. There are extensive studies on the sound level reduction due to the shift to EVs, in which 1–4 dB reduction was estimated (Campello-Vicente, H. et al., 2017; Iversen, L.M. et al., 2013; Kaliski, K. et al., 2012; Lelong, J. and Michelet R, 2001; Ögren, M. et al., 2018; Verheijen, E. and Jabben, J., 2010). The future development of electric mobility may reduce the health risks of road traffic noise due to the decrease in sound levels. The quantification of the impact of health risk reduction and the factors in reducing the health risks, i.e. traffic conditions, however, remain unclear.

1.2 Research objectives

Given the background of study and overview of the problem statement, there is an imminent need to address the gaps in the literature and propose a new understanding of the topic to enhance the interdisciplinary nature of environmental research. On the grounds that health risk assessment is vital to protect public health, the objectives of the study are as follows:

- To develop health risk maps from noise maps as an alternative tool for public risk communication; and
- To analyse the reduction of health risks in a future noise-exposure setting by the transition to EVs from the internal combustion engine vehicles (ICEVs).

The first objective to develop health risk maps is executed with the aim to enhance risk communication with the local residents. The sound levels shown in noise maps are not useful because the acoustic intensity is exhibited instead of the health effects. It would be challenging for the public to recognise the health impacts of environmental noise from ordinary noise maps. To increase their understanding, a tool that structured the health risks appropriately and easily understood is necessary. The health risk map with the numerical information and visual representation of the adverse health effects is suggested as an alternative tool for public risk communication.

The second objective to evaluate the effective traffic conditions in reducing health risks by the shift to EVs is executed with the aims to elucidate the contribution of the mitigation approach. Thus far, the effectiveness of health risks mitigation by the transition from conventional vehicles, i.e. ICEVs to EVs and the factors that could influence the effectiveness remains unclear. The findings in past literatures estimated reductions by the adoption of light and heavy EVs; however, the reduction in health risks was not investigated.

The health risks due to road traffic noise, which is the major source of environmental noise were investigated in the first and second studies. The health outcomes with exposure-response functions (ERFs) and sound levels as revealed in the *Environmental Noise Guidelines for the European Region* (WHO-EU, 2018) issued by the WHO-EU, i.e. high annoyance (HA), high sleep disturbance (HSD), and ischaemic heart disease (IHD) were selected and investigated in both studies.

This dissertation consists of five chapters. Chapter 1 explores the concept of research background, research questions and problems, objective of the research, framework summarising the flow of the methodology, and the significance of study.

Chapter 2 presents a systematic review of previous studies, which consolidates the theories and methodologies in numerous research. The literature review aims to present a comprehensive summary of researches, past epidemiological and socio-acoustic studies on the adverse health effects of traffic noise, including the selected health outcomes of this study, which are explained in detail. Other health outcomes, such as negative metabolic effects and cognitive impairment in children are also introduced.

Chapter 2 also explains the methods to estimate sound levels and health risks to give an insight into the current method of strategic noise mapping in Europe. The noise prediction model as employed in this study, i.e. CNOSSOS-EU, is presented to explain the equations that are used in the model for sound level calculations in both studies and calculations of the relationship between health risks and traffic conditions in the second study. For the second study, a concise explanation on EVs and past literature on the EVs' impact in reducing sound level is given.

Chapter 3 covers the relevant research approaches in making a noise map, and employs the ERFs and national health statistics and surveys to convert the noise map into health risk maps as an alternative risk communication tool. The chapter also presents the developed health risk maps and the health risk distribution in Sapporo City and estimates of the number of population in the study area that is exposed to road traffic noise.

Chapter 4 discusses the impact of the shift to EVs in reducing health risks and the effective traffic conditions. The calculation results of the relationship between health risks and traffic conditions are also shown. The calculation results are validated by the estimations of health risk reduction after the transition in a sample of two study areas in Sapporo City. The chapter also estimates the health risk reduction in the total agglomeration of Sapporo City.

Chapter 5 synthesises the overall findings and discusses the contributions to the theory and body of knowledge. Limitations of the study are addressed and several future research directions are also suggested.

Chapter 2

Literature Review

2.1 Adverse health effects of traffic noise exposure

2.1.1 General overview

Rapid urbanisation in the cities, increasing traffic rate, and increased industrial activities are some of the driving changes that cause various forms of pollution including environmental hazards, i.e. environmental noise pollution, which affects humans both physiologically and psychologically. The physiological effects of noise exposure are emphasised in this chapter in accordance with the objectives of this study. This chapter discusses on past literatures regarding adverse health effects of traffic noise exposure (Section 2.2), methods to estimate sound levels and health risks (Section 2.3), and adoption of electric vehicles (EVs) to mitigate noise (Section 2.4) in order to form a conceptual framework to undertake the study. The literature review suggests the essential aspects, i.e. methods, experimental design, and appropriate procedures. It also helps to define the study problem, provides a rationale for conducting the study, and designs the research.

By way of definition, environmental noise refers to any unwanted sound created by human activities that are considered harmful or detrimental to human health and quality of life (Murphy, E. et al., 2009*b*). The World Health Organization (WHO) defines environmental noise as 'noise emitted from all sources except for noise at the industrial workplace' (WHO-EU, 2011), while the European Union (EU) Directive 2002/49/EC on the management of environmental noise (European Union, 2002) defines environmental noise as 'unwanted or harmful outdoor sound created by human activities, including noise from road, rail, airports and from industrial sites'.

Although not necessarily used consistently, the terms community, residential, or domestic noise have been used interchangeably with environmental noise (WHO-EU, 2011).

Environmental noise has traditionally been dismissed as an inevitable fact of life and has not been targeted and controlled to the same extent as other health risks (Murphy, E. and King, E. A., 2014). Over the past few decades, scientists have made noteworthy progress in measuring the impacts of environmental noise on human health. The progress made is largely from the results of experimental and epidemiological studies of its effects on numerous diseases. What was once referred to as a 'forgotten pollutant' is now recognised as an environmental and public health issue that needs to be addressed in the modern society (Murphy, E. and King, E. A., 2014).

Health effects on the auditory system were first recognised in occupational settings (Basner, M. et al., 2014), where long-term exposure to excessive noise was associated with direct injury to the auditory system, resulting in noise-induced hearing loss (NIHL). NIHL can also be triggered by a one-time exposure to an intense impulse sound, i.e. gunfire and fireworks (Basner, M. et al., 2014; Le, T.N. et al., 2017). It is also a risk factor for tinnitus; a change in sound perception or ringing in the ears. Tinnitus can affect quality of life in several ways, leading to sleep disturbance, insomnia, anxiety, depression, or the inability to sustain attention. While hearing loss often coincides with tinnitus, not every case of tinnitus traces back to hearing loss. Likewise, not every person with hearing loss experiences tinnitus. Nevertheless, hearing loss and tinnitus are often reported in combination, suggesting that both symptoms share common pathophysiological pathways (Basner, M. et al., 2014; Le, T.N. et al., 2017; Stansfeld, S. and Matheson, M., 2003).

Simultaneously, the non-auditory effects of environmental noise exposure are increasingly acknowledged owing to the abundant evidence that environmental noise may pose a threat to the health of the general public. While it is generally recognised that environmental noise is a problem, to the extent to which noise has been linked to a range of non-auditory health effects, it has remained the subject of continued discussion.

The *Burden of Disease from Environmental Noise* (WHO-EU, 2011) produced by the World Health Organization Regional Office for Europe (WHO-EU) is one of the most important documents outlining the importance of the non-auditory effects. The document is the first serious attempt to examine the evidence base for the noise-health relationship, and produces estimates (for which there is sufficient evidence) of the extent of the disease burden using the disability-adjusted life years (DALYs) (Murphy, 2017). The burden of disease is expressed in terms of DALYs, which are the sum of potential years of life lost (YLL) due to ill health, disability, or early death and the equivalent years of healthy life lost due to being in a state of poor health or disability (YLD) (Murphy, E. and King, E. A., 2014).

Table 2.1 displays the summary of results from the *Burden of Disease from Environmental Noise* study (WHO-EU, 2011) in DALYs lost due to environmental noise exposure in Europe. The study concludes that the main impacts are annoyance and sleep disturbance, where one in three individuals in Europe is annoyed during the daytime and one in five has disturbed sleep at night purely from traffic noise alone (WHO-EU, 2011). The five noise-induced exposure impacts mentioned in the table refer to the cumulative result in the loss of approximately 1.0–1.6 million DALYs annually. The various forms of transportation are the main sources of environmental noise exposure, with road traffic noise proves to be the main offender, followed by aircraft and railway noise, both in Europe and cities around the world (Murphy, 2017; European Commission Working Group Assessment of Exposure to Noise (WG-AEN), 2007; Welch, D. et al., 2013).

Table 2.1: Annual burden of disease from environmental noise in Europe, adapted from Murphy, E. and King, E. A. (2014)

Noise-induced Exposure	DALYs[a]
Annoyance	587,000[b]
Sleep disturbance	903,000[c]
Cardiovascular diseases	61,000[d]
Tinnitus[e]	22,000[f]
Cognitive impairment in children	45,000[g]

[a] DALYs are the sum of the potential years of life lost due to premature death (YLL) and the equivalent years of 'healthy' life lost by virtue of being in states of poor health or disability (YLD) (WHO-EU, 2011)

[b] lost for inhabitants in towns >50,000 population

[c] lost for EUR-A (WHO epidemiological subregions in Europe comprising Andora, Austria, Belgium, Croatia, Cyprus, the Czech Republic, Denmark, Finland, France, Germany, Greece, Iceland, Ireland, Israel, Italy, Luxembourg, Malta, Monaco, the Netherlands, Norway, Portugal, San Marino, Slovenia, Spain, Sweden, Switzerland and the United Kingdom) inhabitants in towns >50,000 population

[d] for ischaemic heart disease (IHD) in high-income European countries

[e] Tinnitus is defined as the sensation of sound in the absence of an external sound source (WHO-EU, 2011)

[f] EUR-A adult population

[g] EUR-A countries for children aged 7–19 years

Environmental noise, particularly that is caused by transportation means, may lead to adverse health effects through a nexus of pathways, namely direct and indirect effects (Hahad, O.

et al., 2019; Münzel, T. et al., 2018). The direct effect is sleep disturbance and the indirect effects are interference with cognitive and emotional responses and disturbance in daily activities and communication (Hahad, O. et al., 2019; Münzel, T. et al., 2018). Environmental noise is viewed as a significant cause of sleep disturbance (direct effect) and annoyance (indirect effect), which are potential health stressors that can lead to and/or trigger more serious health problems.

Figure 2.1 summarises the modern noise reaction model established by Babisch, W. (2002), explaining the adverse cardiovascular effects of noise exposure. The noise reaction scheme suggests the ‘direct pathway’ and the existence of an ‘indirect pathway’. Direct pathway is determined by the instantaneous interaction of the acoustic nerve with the various structures of the central nervous system. The indirect pathway is the pathway in which disturbance of activities, sleep, and communication causes cognitive and emotional response and annoyance. This is followed by stress responses and chronic stress, which generates risk factors and eventually causes cardiovascular diseases (CVDs). As a result, both direct and indirect pathways can initiate physiological stress reactions, which may result in a number of negative health effects, especially as a result of long-term exposure (Babisch, W., 2002, 2003).

A recent study suggested that it is conceivable that the indirect pathway via the cortical region may not affect health as a stressor (Eriksson, C. et al., 2018). Therefore, annoyance does not play a role in the relationship between noise exposure and cardiovascular disease, as will be the case if the relationship is primarily driven by direct links between noise exposure and subcortical areas (Eriksson, C. et al., 2018). Meanwhile, noise induced physiological responses during sleep are well-documented, principally denoting one pathway is largely unmediated by auditory experience and appraisal (Berglund, B. et al., 1999).

Figure 2.2 illustrates a pyramid of health effects which shows the severity of health and well-being due to long-term environmental noise exposure. The health consequences vary from feelings of discomfort to stress and increased risk of CVDs and ultimately mortality. In a part of a population exposed to high noise levels, stress reactions, sleep-stage changes, and other biological and biophysical effects may occur. In turn, these may lead to health risk factors deterioration, i.e. blood pressure and blood lipids. The subsequent changes may develop clinical symptoms, such as insomnia and CVDs for a relatively small part of the population and consequently increase the rates of premature mortality (European Environment Agency, 2014).

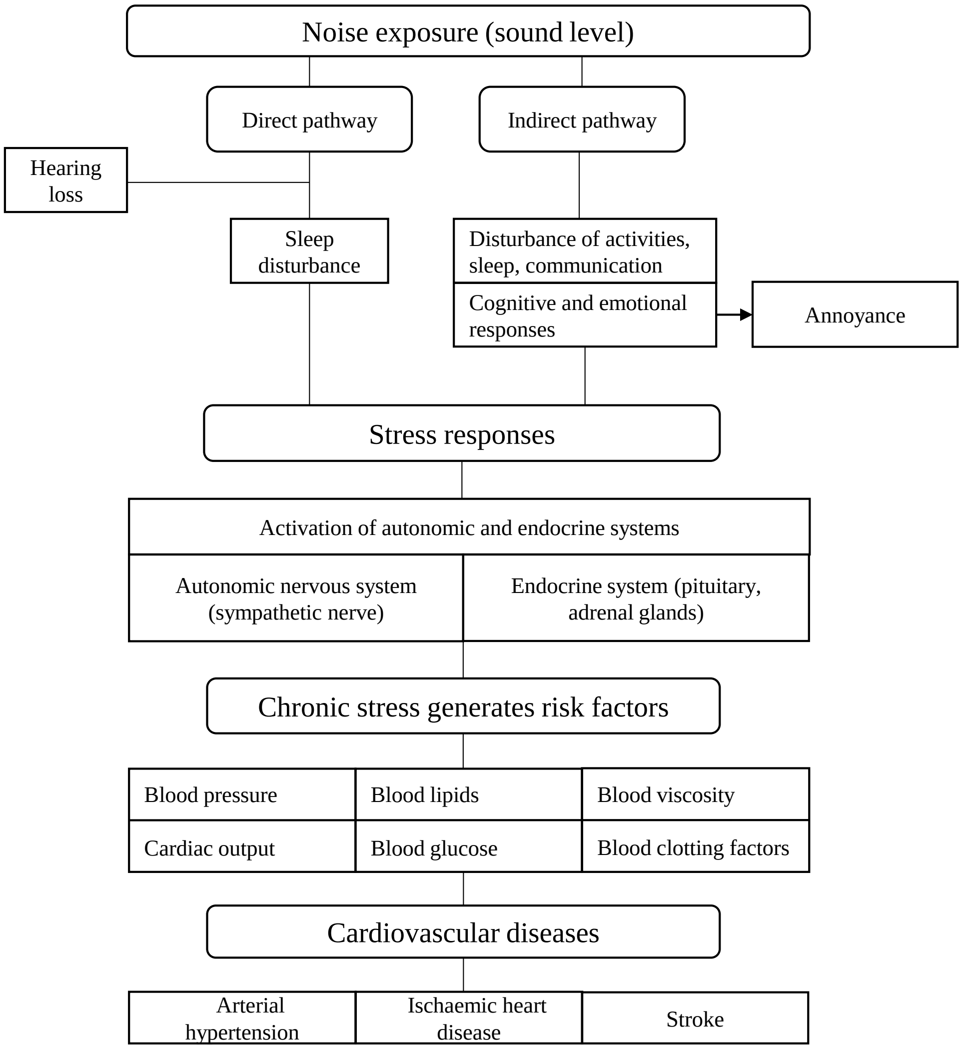

Figure 2.1: Biological pathways to explain adverse health effects due to noise exposure. The direct pathway (left) illustrates subcortical reactions including noise-induced hearing loss and sleep disturbances. The indirect pathway (right) illustrates cortical feedback reactions including the psychological effects of annoyance and a part of sleep disturbances. They may act as stressors, affect the homeostasis, and change physiological parameters, such as blood pressure and blood flow. Chronic changes may lead to severe outcomes, such as ischaemic heart disease (IHD) (Münzel, T. et al., 2018).

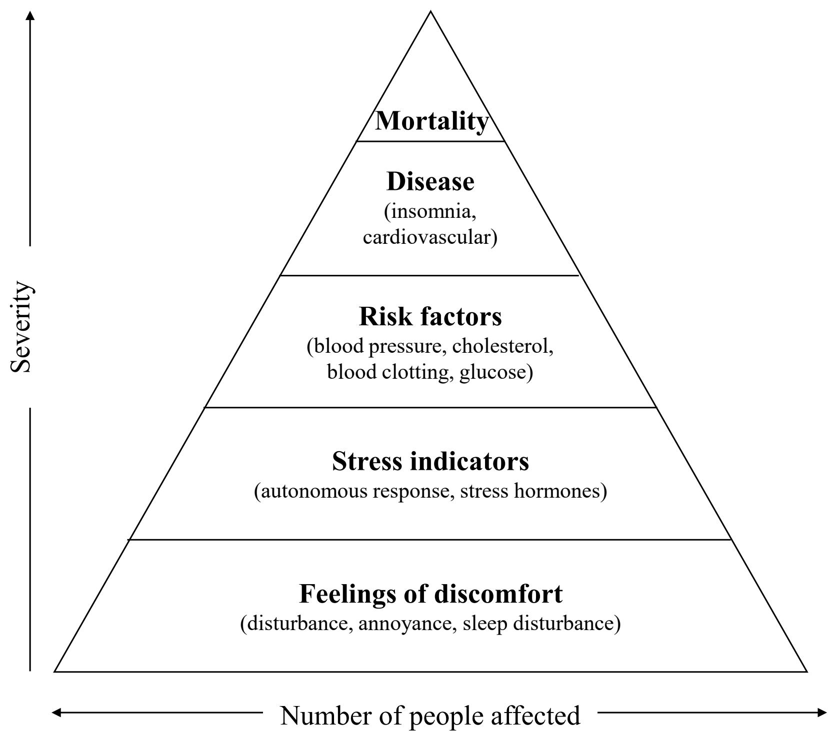

Figure 2.2: Pyramid of health effects of noise, which shows how noise exposure affects health. The bottom base shows the number of people affected. This figure decreases to the top. The height of the pyramid reflects the increase in the severity of the effects. From the bottom to top, the effects are feeling of discomfort (disturbance, annoyance, sleep disturbance), stress indicators (autonomous response, stress hormones), risk factors (blood pressure, cholesterol, blood clotting, glucose), disease (insomnia, cardiovascular), and mortality (Babisch, W., 2002).

2.1.2 Annoyance

Environmental noise is a 'hidden' type of pollutant where the damage is usually long-term and of permanent nature. Thus far, most assessments of environmental noise problem have been based on the annoyance it causes to humans, or the extent to which it disturbs various human activities. Assessment of health outcomes potentially related to noise exposure is limited (WHO-EU, 2011).

Annoyance is defined as 'a feeling of displeasure associated with any agent or condition, known or believed by an individual or group to adversely affect them' (Lindvall, T. and Radford, E.P, 1973). Although annoyance is a 'feeling', the broad definition of health as given by the WHO Constitution, 'a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity' embraces the concept of well-being and, thereby, renders noise impacts, such as population annoyance, interference with communication, and impaired task performance as 'health' issues (Berglund, B. et al., 1999). Therefore, a high level of annoyance caused by environmental noise should be considered as one of the environmental health burdens (WHO-EU, 2011) and taken into account when estimating the health effects of noise.

Noise annoyance is typically associated with the indirect reaction chain in humans that is closely related to the initiation of emotional stress. Studies have shown that individuals annoyed by noise tend to experience a series of negative emotions, including anger, disappointment, unhappiness, withdrawal, distraction, anxiety, exhaustion, and even depression (WHO-EU, 2011). The principal reaction schema in Figure 2.1 simplifies the cause-effect chain of noise and annoyance through the emotional and cognitive perception of the sound.

In the context of environmental noise, annoyance is often observed due to transportation via aircraft, road, and rail, and partially in industrial neighbourhoods (Guski, R. et al., 2017). The exposure-response functions (ERFs) relating a noise indicator to a health outcome can be used for health impact assessments and political decision-making against environmental noise. A number of studies have been conducted to establish relationships to show which annoyance level is associated with a given noise exposure level. The synthesis curves for road traffic, railway, and aircraft noise were presented by Miedema, H.M.E. and Vos, H. (1998) in which attempts were made to find the 95% confidence intervals around the exposure-response curves, taking into account the variation between individuals and studies. These curves were based on the previous studies analysed by Schultz, T.H.J. (1978) and Fidell, S. et al. (1991) for which day-night level (*DNL*) and percentage of highly annoyed persons (%HA) meeting certain requirements could be derived, augmented with a number of additional studies (Miedema, H.M.E. and Vos, H., 1998). The synthesis by Miedema, H.M.E. and Vos, H. (1998) was more comprehensive than the previous ones.

In a later study (Miedema, H.M.E. and Oudshoorn, C.G., 2001), the methods used to establish the confidence intervals were improved where the relationship model between exposure and annoyance is more sophisticated and better suited for the data. The model provides a firmer basis for the relationships and their confidence intervals. The ERFs for both severe annoyance and annoyance were derived for road traffic, railway, and aircraft noise, with higher annoyance for aircraft noise than for road traffic or railway noise at the same exposure level. These curves have been recommended for use in the European Union (EU) legislation on noise (Commission of the European Communities, 2002).

The derived ERFs from the epidemiological study (Miedema, H.M.E. and Oudshoorn, C.G., 2001) were employed for the environmental burden of disease and exposed population estimations in numerous studies. The WHO (WHO-EU, 2011) reported that one in three individuals is annoyed during the daytime, while 57 million people (12% of the population) in 25 EU countries are annoyed by road traffic noise; approximately 24 million (42%) of those are thought to be severely annoyed (den Boer, L.C. and Schroten, A., 2007). A research in a highly-populated city in Korea estimated that approximately 10% of the total population (n = 1,471,944) experiences high-level annoyance due to road traffic noise (Park, T. et al., 2018). Another study estimated that railway noise causes annoyance in about 5.5 million people (1% of the European population), of whom 2 million are severely annoyed (den Boer, L.C. and Schroten, A., 2007). As for aircraft noise, it was estimated that over 2.6 million people in Europe were exposed to aircraft noise levels above 55 dB(A) in the year 2007, rising to almost 3.3 million in 2015. Approximately 15% of them were estimated to be highly annoyed (Janssen, S.A. et al., 2011).

A subsequent study updated the ERF for annoyance due to aircraft noise based on studies between 1991 and 2006 from seven European studies conducted in Switzerland, Germany, and the Netherlands (Janssen, S.A. and Vos, H., 2009). The updated ERF for aircraft noise (Janssen, S.A. and Vos, H., 2009) and ERFs for road traffic and railway noise (Miedema, H.M.E. and Oudshoorn, C.G., 2001) were employed in an assessment carried out for 33 countries in the Europe. The assessment reported that in 2012, around 27.6 million adults living in agglomerations or near major sources with noise levels equal or above 55 dB L_{den} (day-evening-night equivalent sound level) may be considered as being annoyed by noise from road traffic, railways, aircrafts or industry; 12.8 million of them were severely annoyed (Blanes, N. et al., 2016).

Most studies in European countries indicated that railway noise causes less annoyance at the same average sound level than other transportation noise sources (Fields, J.M. and Walker, J.G., 1982; Knall, V. and Schuemer, R., 1983; Miedema, H.M.E. and Oudshoorn, C.G., 2001; Miedema, H.M.E. and Vos, H., 1998; Möhler, U., 1988). The reflection of the findings, where at the same noise level, railway noise is evaluated to be less annoying than road traffic noise, was defined as "railway bonus" in the noise regulations of some European countries (Ma, H. and Yano, T., 2005). In Germany, a bonus of 5 dB(A) has been set by the German noise regulations, i.e. railway noise is assumed to be 5 dB(A) louder than road traffic noise to achieve the same level of annoyance (Schreckenberg, D. et al., 1999). Similarly, ISO 1996-1 recommends a railway noise bonus of between 3 and 6 dB(A) in railway noise assessments (ISO 1996-1:2016, 2016).

However, there are some studies that produced different results. Nearly four decades ago, a past study in the United Kingdom compared three surveys including railway and road traffic noises and suggested that the railway noise was not always less annoying than road traffic noise (Berry, B.F., 1983). Moreover, the regression lines for aircraft and road traffic noise at the same annoyance degree seemed to be almost parallel with a 10 dB(A) difference and implied higher annoyance from aircraft noise.

Previous post-1990s Japanese social surveys and simulated laboratory studies (Igarashi, J., 1992; Kaku, J. and Yamada, I., 1996; Morihara, T. et al., 2004; Yano, T. et al., 1997) also showed that railway bonus does not exist and that railway noise annoyance is nearly the same as or slightly higher than road traffic noise annoyance. Another Japanese study determined the annoyance responses to different transportation noises (road, railway, and aircraft traffic) and concluded that railway noise appeared to be the most prominent noise source in the overall annoyance, especially at moderate and low noise levels (Kurra, S. et al., 1999).

In view of population exposure, studies have shown that road traffic noise is responsible for causing the greatest levels of annoyance. More people are affected by road traffic noise than railway noise because the number of exposed populations is much larger. The results from a study on the basis of survey data in the Netherlands reported that road traffic noise is responsible for the largest number of highly annoyed people by noise during sleep, and railway is responsible for the least number of highly annoyed people among the three transportation noises (WHO-EU, 2009). Some countries in the WHO European Region conducted large-scale national surveys on noise annoyance and indicated road traffic noise as the most important source of annoyance (WHO-EU, 2018).

Aircraft noise can also be a substantial source of annoyance and affect a much smaller proportion of the population compared to road traffic noise. It is often cited as a reason against airport expansion and is one of the most common complaints raised by residents living in the vicinity of airports (Murphy, E. and King, E. A., 2014). There are several studies that reported higher percentages of people highly annoyed for aircraft noise compared to road traffic noise. A recent study in Switzerland found that aircraft noise annoyance elicited the highest percentages of highly annoyed persons at the same sound level, followed by railway noise, and finally road traffic noise (Brink, M. et al., 2019). Meanwhile, general (not specific for nighttime) annoyance data from Germany and the United Kingdom give an indication that similar numbers of people are affected (WHO-EU, 2009).

The WHO-EU has presented a methodology for estimating the prevalence of noise annoyance by combining the existing noise exposure data with the exposure-response relationships for noise annoyance that were determined in previous studies (WHO-EU, 2011). In the most recent guidelines, the WHO-EU identified a massive number of studies that proves the association between road traffic noise and annoyance and were used to model the ERFs of the relationship between L_{den} and percentage of the population "highly annoyed", %HA for road traffic, railway, and aircraft noise (WHO-EU, 2018). %HA is the prevalence indicator used for annoyance in a population.

The guidelines are underpinned by peer-reviewed systematic reviews of the pertinent literature in order to incorporate the significant research since the publication of the *Night Noise Guidelines for Europe* in 2009 (WHO-EU, 2009). The WHO-EU review on environmental noise and annoyance (Guski, R. et al., 2017) found evidence of correlations between noise levels and annoyance. The strength of the evidence is seen in the large total sample size encompassing the included studies and the main limitations are due to the variance in the definition of noise levels and %HA (Guski, R. et al., 2017).

The systematic literature search in 20 data bases identified 1,700 studies which resulted in 62 studies after screening, of which 57 were used for quantitative meta-analyses. Of these, a total of 17 road traffic noise studies were identified to model ERFs of the relationship between L_{den} and %HA. These incorporated data from 34,112 study participants. On the association between railway noise and annoyance, a total of ten studies with ERFs were included in the analyses, which incorporated individual data from 10,970 participants. For the relationship between aircraft noise and annoyance, in total, 12 aircraft noise studies were identified to model ERFs of the relationship between L_{den} and %HA. These include data from 17,094 study participants (WHO-EU, 2018).

The derived ERFs are based on the regression equations from the systematic review (Guski, R. et al., 2017) (Equation 2.1 to Equation 2.3):

$$\%\text{HA}_{\text{road}} = 78.9270 - 3.1162 \times L_{\text{den}} + 0.0342 \times L_{\text{den}}^2 \tag{2.1}$$

$$\%\text{HA}_{\text{railway}} = 38.1596 - 2.05538 \times L_{\text{den}} + 0.0285 \times L_{\text{den}}^2 \tag{2.2}$$

$$\%\text{HA}_{\text{aircraft}} = -50.9693 + 1.0168 \times L_{\text{den}} + 0.0072 \times L_{\text{den}}^2 \tag{2.3}$$

where, the range is $40\,\text{dB} \leq L_{\text{den}} \leq 80\,\text{dB}$ for road traffic and railway noise, and $40\,\text{dB} \leq L_{\text{den}} \leq 70\,\text{dB}$ for aircraft noise. At the same equivalent noise level, annoyance is the highest for aircraft noise, followed by railway and road traffic noise (Figure 2.3).

The WHO-EU review found moderate to high quality evidence for statistically significant correlations between noise levels and annoyance raw scores with respect to aircraft, road, railway, and noise source combinations and for the increase of %HA, expressed in terms of odds risk (OR) with a 10 dB increase in levels of aircraft, road, and railway noise (Guski, R. et al., 2017). The analysis of newer surveys (2000–2014) on annoyance due to traffic noise shows statistically significant correlations between noise levels and annoyance scores with moderate strength of the relationship.

The quality of evidence ratings is based on the Grading of Recommendations Assessment, Development, and Evaluation (GRADE). The levels of quality of evidence of the GRADE system are: 1. High – further research is very unlikely to change the confidence in the estimate of effect, 2. Moderate – further research is likely to have an important impact on our confidence in the estimate of effect and may change the estimate, 3. Low – further research is very likely to have an important impact on the confidence in the estimate of effect and is likely to change the estimate, and 4. Very Low – Any estimate of effect is very uncertain.

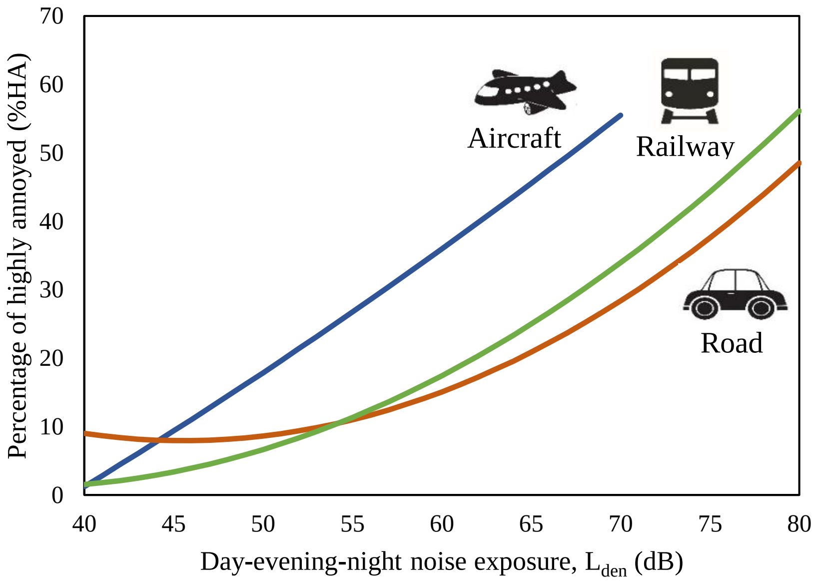

Figure 2.3: Exposure-response relationship of percentage of highly annoyed people (%HA) due to aircraft, road, and rail traffic noises, expressed in day-evening-night equivalent sound level, L_{den} (dB). Adapted from the *Environmental Noise Guidelines* (Guski, R. et al., 2017; WHO-EU, 2018).

2.1.3 Sleep disturbance

Uninterrupted sleep is recognised to be a prerequisite for a healthy person's physiological and mental functioning (Berglund, B. et al., 1999). A reduction in sleep quality is associated with a number of secondary impacts which are typically felt the day after a disturbance has occurred. The after-effects may be evident in an individual including fatigue, low work capacity, reduced cognitive performance, changes in daytime behaviour, and also mood changes and associated negative emotions (Murphy, E. et al., 2009*a*).

Sleep disturbance is considered to be one of the most pronounced and significant health effects of environmental noise. This is attributed to epidemiological studies proving that habitual short sleep (<6 h per night) is associated with obesity, diabetes, hypertension, CVDs, and all-cause mortality, stressing the importance of undisturbed sleep of sufficient length for health in general and cardiovascular health specifically (Münzel, T. et al., 2014).

Sleep disturbance is a broad term describing any sleeping problems, including difficulty of falling asleep, waking often during the night, and decreased quantity or quality of sleep (Zaharna, M. and Guilleminault, C., 2010). There is substantial evidence that excessive environmental noise disturbs sleep. The International Classification of Sleep Disorders (ICSD) classifies 'environmental sleep disorder' as a component of "other sleep disorder" as it is rarely diagnosed in clinical practice and there is controversy regarding it as a true clinical diagnosis (American Academy of Sleep Medicine, 2005). Environmental sleep disorder is defined as a sleep disturbance that is caused by a disturbing environmental factor, which disrupts sleep and leads to a complaint of either insomnia or daytime fatigue or somnolence (Thorpy, M.J., 2012).

As a part of living, and along with being awake, sleep forms an inherent biological rhythm (Cooper, R., 1994). Sleep is an important modulator of hormonal release, glucose regulation, and cardiovascular function (Halperin, D., 2014; Tasali, E. et al., 2008; Van Cauter, E. et al., 2008). The initiation of the slow-wave sleep (SWS), i.e. the deeper sleep stage, in particular, is associated with decreased heart rate and blood pressure, sympathetic nervous activity, and cerebral glucose utilisation when compared with wakefulness (Halperin, D., 2014; Van Cauter, E. et al., 2008).

Excessive environmental noise disturbs sleep in the form of arousals and awakenings and in reducing the amount of time an individual spends in the deep-sleep stages (Murphy, 2017; WHO-EU, 2011). Studies have shown that these various forms of disturbance cause short-term consequences of sleep disturbance, which can be divided into immediate primary (cortical arousals and awakenings, sleep stage change, and autonomic cardiovascular arousal) and 'next-day' secondary effects (fatigue, drowsiness, and reduced performance) (den Boer, L.C. and Schroten, A., 2007; Halperin, D., 2014; Medic, G. et al., 2017; Van Cauter, E. et al., 2008; WHO-EU, 2011).

If transportation noise exposure continues for a prolonged period of time, a chronic noise-induced sleep disturbance may occur. At a level that is severe enough, sleep disturbance can lead to sleep deprivation, which is generally associated with deleterious effects on the physical and mental health of an individual (WHO-EU, 2011). The related long-term health problems include hypertension, heart disease, and cognitive impairment in children (den Boer, L.C. and Schroten, A., 2007; Halperin, D., 2014; Medic, G. et al., 2017; Van Cauter, E. et al., 2008; WHO-EU, 2011).

Objectively, sleep disturbance can be quantified by the number and duration of nocturnal awakenings, the number of sleep stage changes and modifications in their amounts (Murphy, E. and King, E. A., 2014). Electrophysiologically, sleep disturbances can be measured using the so-called polysomnography (PSG) and subjectively, with self-reporting using survey questionnaires distributed to subjects on the morning after a night's sleep (Murphy, E. and King, E. A., 2014; WHO-EU, 2011). In epidemiological studies, self-reported sleep disturbance is the most easily measurable outcome indicator because physiological measurements are costly and difficult to carry out on large samples and may themselves influence sleep (WHO-EU, 2011).

However, subjective and objective assessments of sleep may be substantially discrepant due to sleep misperception and measurement effects, the latter of which may change the quality and quantity of a person's usual sleep (Silva, G.E. et al., 2007). Consistency between subjective and objective sleep measures has also been reported in two studies using small samples from populations with specific somatic or psychiatric disorders (Baekeland, F. and Hoy, P., 1971; Carskadon, M.A. et al., 1976), and thus may not reflect the overall estimates for a community population (Silva, G.E. et al., 2007). Nevertheless, self-reported sleep disturbance may have validity in its own right by reflecting the impact on sleep as perceived by the subject over a longer period of time (WHO-EU, 2011).

Qualitative and quantitative sleep disturbance due to noise from air, railway, and road traffic has been shown in laboratory settings and field studies (Basner, M. et al., 2011; Marks, A. and Griefahn, B., 2007; Ohrstrom, E., 1999). Dissatisfaction with sleep quantity can be expressed as a complaint of insufficient sleep, and dissatisfaction with the quality of sleep is expressed in several ways, i.e. complaint of difficulty initiating sleep (DIS), difficulty maintaining sleep (DMS), and nocturnal awakening with difficulty or inability resuming sleep or non-restorative sleep (Ohayon, M., 1996). However, the results from laboratory settings are not useful in assessing the quantitative aspects of sleep disturbance.

An early review showed that stronger effects were observed in ERFs derived from laboratory studies compared to field investigations at the same noise event descriptors of maximum sound level (L_{max}) or sound exposure level (*SEL*) (Pearsons, K.S. et al., 1990). These differences can be explained by habituation to nighttime noise of subjects in field studies in contrast to the unusual exposure of subjects in laboratory studies. Several studies have proposed the ERFs, which give the probability of noise-induced awakening as a function of L_{max} and *SEL*, by taking the observation by Pearsons, K.S. et al. (1990) into consideration (Finegold, L.S. and Elias, B., 2002; Passchier-Vermeer, W., 1994; Pearsons, K.S. et al., 1990).

Subsequent study using the nightime equivalent sound level (L_{night}) noise indicator proposed by the European Union (2002) presented synthesis curves for self-reported sleep disturbance from aircraft, road traffic, and railway noise (Miedema, H.M.E. et al., 2002). These curves were based on pooled data from 12 field studies. The WHO-EU has presented a methodology for estimating sleep disturbance by integrating the existing noise exposure data with the exposure-response relationships for sleep disturbance that were determined by Miedema, H.M.E. et al. (2002) (WHO-EU, 2011). The WHO-EU considered sleep disturbance to be the major health effect of environmental noise, where an estimated 903,000 DALYs were lost from sleep disturbance in the population living in town of > 50,000 inhabitants as a result of environmental noise exposure in the EU (WHO-EU, 2011).

A more extensive analysis based partly on the same data as Miedema, H.M.E. et al. (2002) and pooled data from 24 field studies yielded similar curves and took into account the variation between individuals and studies (Miedema, H.M.E. and Vos, H., 2007). The exposure-response relations for road traffic and railway noise from this study and aircraft noise from Janssen, S.A. and Vos, H. (2009) were applied in an assessment in 32 countries in Europe (Blanes, N. et al., 2016). The assessment estimated that 13.1 million adults have sleep disturbance due to nighttime noise levels $\geq$ 50 dB L_{night} from road traffic, railway, aircraft or industry. Of these, 6.1 million are highly sleep-disturbed. Approximately 85% of the burden of annoyance and sleep disturbance is related to road traffic noise (Blanes, N. et al., 2016).

In the most recent guidelines, the WHO-EU identified multiple studies that were used to model the ERFs of the relationship between L_{night} and percentage of the population "highly sleep-disturbed", %HSD, for road traffic, railway, and aircraft noise (WHO-EU, 2018). %HSD is the prevalence indicator used for sleep disturbance in a population. Seventy four studies predominately conducted between 2000 and 2015 were included in the review. A meta-analysis of surveys linking road, rail, and aircraft noise exposure to self-reports of sleep disturbance was conducted.

For road traffic noise and self-reported sleep outcomes (awakenings from sleep, the process of falling asleep and sleep disturbance), 12 studies were identified that included a total of 20,120 participants; these were cross-sectional studies, conducted in healthy adults. On the association between railway noise and sleep disturbance, a total of five studies were included in analyses, which incorporated individual data from 7,133 participants. In total, six aircraft noise studies were identified, involving 6,371 study participants (WHO-EU, 2018).

The calculations are based on the regression equations derived from the systematic review on environmental noise and effects on sleep (Halperin, D., 2014) (Equation 2.4 to Equation 2.6):

$$\%\text{HSD}_{\text{road}} = 19.4312 - 0.9336 \times L_{\text{night}} + 0.0126 \times L_{\text{night}}^2 \tag{2.4}$$

$$\%\text{HSD}_{\text{railway}} = 67.5406 - 3.1852 \times L_{\text{night}} + 0.0391 \times L_{\text{night}}^2 \tag{2.5}$$

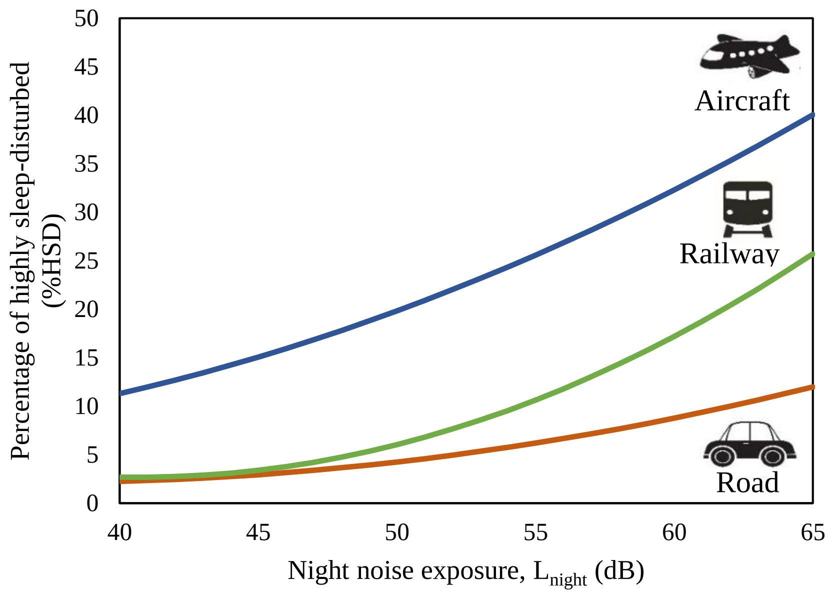

Figure 2.4: Exposure-response relationship of percentage of highly sleep-disturbed people (%HSD) due to aircraft, road, and rail traffic noises, expressed in night noise exposure, L_{night} (dB). Adapted from *Environmental Noise Guidelines* (Halperin, D., 2014; WHO-EU, 2018).

$$\%\text{HSD}_{\text{aircraft}} = 16.7885 - 0.9293 \times L_{\text{night}} + 0.0198 \times L_{\text{night}}^2 \tag{2.6}$$

where, the range is $40\,\text{dB} \leq L_{\text{night}} \leq 65\,\text{dB}$ for road traffic, railway and aircraft. The curves of the ERFs are shown in Figure 2.4. Aircraft noise is associated with more sleep disturbances than the road traffic and railway noise at equivalent noise levels.

The WHO-EU systematic review found that the odds ratio for the %HSD for a 10 dB increase in L_{night} was significant for aircraft, road, and railway noise when the question referred to noise, but non-significant when the question did not refer to noise. The quality of the evidence was rated moderate for cortical awakenings and self-reported sleep disturbance (for questions that referred to noise) induced by traffic noise according to the GRADE criteria.

2.1.4 Cardiovascular diseases

Cardiovascular diseases (CVDs) are a group of disorders of the heart and blood vessels, including arterial hypertension, stroke, and ischaemic heart disease (IHD) (WHO, 2020*a*). As explained in the noise/stress reaction model (Figure 2.1), the biological mechanisms linking noise to CVDs are thought to be the arousal of the endocrine and autonomic nervous systems, which may generate the risk factors of CVDs.

CVDs are the leading causes of mortality in developed and developing countries; more people die annually from CVDs than from any other cause, contributing to more than one-third of the global mortality (Eriksson, C. et al., 2018; Mathers, C.D. et al., 2003; WHO, 2014). In 2015, the WHO estimated that 17.7 million people died from CVDs, representing 30% of all global deaths. An estimated 7.4 million of those deaths were due to coronary heart disease and 6.7 million were due to stroke (WHO Africa, 2020). It was forecasted that by 2030, almost 23.6 million people will die from CVDs, mainly from heart disease and stroke (Eriksson, C. et al., 2018).

In 1994, the International Committee of the Dutch Health Council estimated that the observed threshold for IHD and hypertension corresponded to an L_{dn} (day-night equivalent sound level) value of 70 dB for environmental noise exposure (Health Council of the Netherlands: Committee on Noise and Health, 1994). Later on, the WHO-EU states that the relationship between noise exposure and cardiovascular risks occurs at 65–70 dB $L_{\mathrm{eq,24hr}}$ (continuous equivalent noise level over 24 hours) and that the association is stronger for IHD than hypertension (Berglund, B. et al., 1999). However, the associations from the epidemiological studies are weak. In 2010, the European Environment Agency (EEA) concluded that IHD and hypertension begin to occur or start to rise above background at 60 dB and 50 dB L_{den}, respectively (European Environment, 2010). The lowering of the thresholds is the results from reasonably well-established ERFs which are derived from epidemiological studies.

Studies on the relationship between environmental noise and CVDs often use L_{day} (day equivalent sound level) as the noise indicator. However, both the Dutch Health Council and WHO-EU have made statements on the effects of nighttime noise exposure on the cardiovascular system. Both organisations drew a similar conclusion that although a relationship between nighttime noise exposure and an increased risk of hypertension and CVDs is likely, there is limited, indirect evidence of a causal relationship between exposure to nighttime noise and high blood pressure, cardiac disease (Health Council of the Netherlands: Committee on Noise and Health, 1994), hypertension, and myocardial infarction (WHO-EU, 2009). The WHO-EU also recommends a general threshold of 55 dB L_{night} and an optimal target of 40 dB (WHO-EU, 2009).

Epidemiological studies on the relationship between transportation noise (particularly road traffic and aircraft noise) and CVDs have been carried out on adults and children, focusing on mean blood pressure, hypertension, and IHD as cardiovascular end-points. New evidences on the relationship have accumulated in recent years. The epidemiological studies have provided positive evidence that traffic noise exposure is linked to the above-mentioned CVDs (Babisch, W., 2003, 2014; Belojević, G.A. et al., 2008; Eriksson, C. et al., 2018; Münzel, T. et al., 2018; Münzel, T. et al., 2014; Selander, J. et al., 2009; Van Kempen, E. et al., 2018; WHO-EU, 2009). However, there are very few longitudinal (cohort and case-control) studies available on the cardiovascular impact of transportation noise, with the exception for road traffic noise and IHD. The plausibility of an association, however, requires further and enhanced analysis. In addition, most studies on environmental noise and CVDs focus on IHD and hypertension, with only a few recent studies on stroke. The epidemiological studies mainly report on road and air traffic noise, and there are limited studies on the impacts of railway noise on CVDs.

The WHO-EU estimated the annual burden of disease due to CVDs to be 61,000 years for IHD in high-income European countries (WHO-EU, 2011). Based on the 2012 noise data of 33 countries in Europe, the exposure to environmental noise was reported to contribute to 1.5 million prevalent cases of hypertension among adults. In addition, the total number of hospital admissions for CVDs that are related to noise exposure is estimated to be almost 72,000 cases per year (Blanes, N. et al., 2016). An updated assessment estimated 48,000 new cases of IHD per year and 12,000 premature mortality due to IHD in the European territory as a result of long-term exposure to environmental noise (European Environment Agency, 2020).

2.1.4.1 Ischaemic heart disease

Epidemiological evidences that suggest traffic noise increases the risk of IHD including myocardial infarction (WHO-EU, 2011) have increased significantly. The most comprehensive evidences are available for road traffic noise and IHD. Due to the insufficient studies examining the relationship between aircraft noise and cardiovascular risks, the evidence linking road traffic noise and IHD is stronger than that for aircraft noise (WHO-EU, 2011).

Ischaemic heart diseases include angina, acute myocardial infarction, subsequent myocardial infarctions and complications of infarctions, other acute forms of IHD and chronic IHD (WHO-EU, 2011). A meta-analysis (Babisch, W., 2006, 2008) of 61 epidemiological studies that were conducted from the mid-1990s to 2005 was carried out to assess the relationship between transportation noise and myocardial infarction. The study subjects were adults and children. The studies conducted were either objective or subjective assessment and were mostly of the cross-sectional type (descriptive studies) but observational studies such as case-control and cohort studies (analytical studies) were also available. Most of the studies referred to road traffic noise or commercial aircraft noise, and only a few to military aircraft noise (WHO-EU, 2011).

A total of seven studies; five analytical (Babisch, W. et al., 2005, 2003, 1999, 1994) and two descriptive (Babisch, W. et al., 1993) studies, were selected for deriving the ERF for the association between road traffic noise and myocardial infarction based on the criteria set for the inclusion in the analysis process. The ERF reflects the incidence of myocardial infarction as a function of road traffic noise, measured as $L_{\text{day,16hr}}$ (day equivalent sound level). Myocardial infarction was considered for the meta-analysis because it was the most commonly assessed outcome in the epidemiological studies. The noise impact on myocardial infarction may have been easier to detect because misclassification in the diagnosis of myocardial infarction is less likely than for all IHDs (WHO-EU, 2011).

Although the WHO report (WHO-EU, 2011) is relatively recent, it does not include estimates from the latest epidemiological noise research. Another meta-analysis (Vienneau, D. et al., 2015) was conducted, which included new studies that were selected on the basis of having adjusted for air pollution levels. This is important given that noise and air pollution exposures are often highly correlated, both deriving from traffic sources, and are associated with CVDs (Davies, H.W. et al., 2009). The study identified ten studies on road and aircraft noise exposure conducted since the mid-1990s, providing a total of 12 risk estimates.

The latest meta-analysis providing the ERFs between transportation noise and IHD is the *Environmental Noise Guidelines* (WHO-EU, 2018). A systematic review on the effects of environmental noise exposure on the cardio-metabolic systems (Van Kempen, E. et al., 2018) was conducted as input for the guidelines to update the current state of evidence and assess its quality. 600 references relating to studies on effects of noise from road, rail and air traffic, and wind turbines on the cardio-metabolic system, published between January 2000 and August 2015 were identified. Only 61 studies included information enabling estimation of exposure response relationships. These studies were used for meta-analyses, and assessments of the quality of evidence using the GRADE.

On the relationship between railway noise and the incidence of or mortality from IHD, no evidence was available. Four cross-sectional studies were identified, however, that assessed the prevalence of IHD in a total of 13,241 participants. The overall risk was not statistically significantly increased, with inconsistency across studies (WHO-EU, 2018).

An increase in road traffic noise was associated with the significant increases in the prevalence of IHD, and the incidence of IHD, with the relationship between road traffic noise and the incidence of IHD being the most robust (Van Kempen, E. et al., 2018). For the relationship between road traffic noise and the incidence of IHD, a total of three cohort and four case-control studies were reviewed, which involved a total of 67,224 participants. Furthermore, additional evidence was available from eight cross-sectional studies that investigated the relationship between road traffic noise and prevalence of IHD. These studies involved a total of 25,682 participants. Mortality from IHD was also investigated in one case-control and two cohort studies, which involved 532,268 participants (WHO-EU, 2018).

The WHO-EU review reported a statistically significant association for the incidence of IHD with a relative risk (RR) of 1.08 per 10 dB L_{den} (Equation 2.7) (Van Kempen, E. et al., 2018). In addition, a visualisation of the shape of the association between road traffic noise and the incidence of IHD indicated that the risk of IHD increases from above 50 dB L_{den}, which is consistent with the findings by Vienneau, D. et al. (2015).

No cohort or case-control studies on the relationship between aircraft noise and IHD are available. However, two ecological studies were identified that provide information on the relationship between aircraft noise and incidence (hospital admission) of IHD. These involved a total of 9,619,082 participants. Two cross-sectional studies were identified that assessed the prevalence of IHD in people living in cities located around airports in Europe, which involved 14,098 participants. With regard to the relationship between aircraft noise and mortality due to IHD, one cohort study (4,580,311 participants living in Switzerland) and two ecological studies (3,897,645 participants in the Netherlands and the United Kingdom) were identified (WHO-EU, 2018).

For aircraft noise and incidence of IHD, the review reported a statistically significant association with a RR of 1.09 per 10 dB L_{den} (Equation 2.8) based on the two ecological studies (Van Kempen, E. et al., 2018). Compared with noise from road and aircraft traffic, only a few studies investigated the impact of noise from rail traffic. The association between rail traffic noise and the prevalence of IHD was found to be non-significant.

The RRs of IHD due to road traffic and aircraft noise were estimated by the following equations (Equation 2.7 and Equation 2.8), across a noise range of 40 dB $\leq L_{\text{den}} \leq$ 80 dB (Van Kempen, E. et al., 2018; WHO-EU, 2018).

$$\text{RR}_{\text{road}} = 1.08^{(L_{\text{den}}-53)/10} \tag{2.7}$$

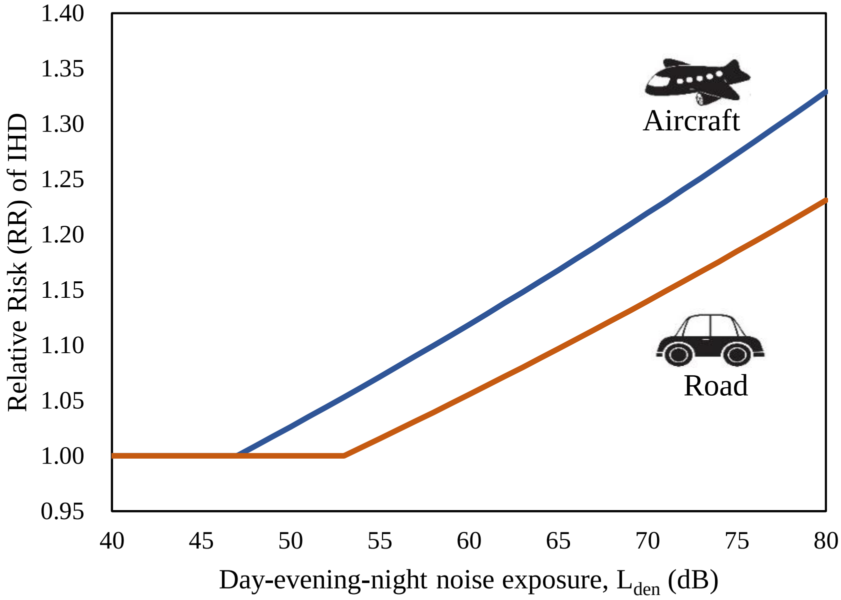

Figure 2.5: Exposure-response relationship of the relative risk (RR) of ischaemic heart disease (IHD) due to aircraft and road traffic noises, expressed in day-evening-night noise exposure, L_{den} (dB). Adapted from *Environmental Noise Guidelines* (Van Kempen, E. et al., 2018; WHO-EU, 2018).

$$\text{RR}_{\text{aircraft}} = 1.09^{(L_{\text{den}}-47)/10} \tag{2.8}$$

The curves of the ERFs are presented in Figure 2.5. Aircraft noise is associated with higher increase of IHD than road traffic noise at equivalent noise levels. However, it should be noted that the quality of evidence supporting the association between aircraft noise and IHD was rated as low, while for road traffic, it was rated as moderate according to the GRADE criteria.

The main divergence from the GRADE criteria was that the initial level of certainty was rated “high” for cohort and case-control studies, “low” for cross-sectional studies, and “very low” for ecological studies (Van Kempen, E. et al., 2018). For aircraft noise, most studies on the impact of aircraft noise were of ecological and cross-sectional design, which were considered as weak to deduce the causality.

2.1.4.2 Hypertension

Hypertension, also known as high blood pressure, is a serious medical condition in which the blood vessels have persistently elevated pressure. It is a common condition and an important risk indicator for IHD and other CVDs. Therefore, the risk increase for hypertension can be transformed into a risk increase for CVDs (WHO-EU, 2018).

Hypertension is a major cause of premature death worldwide. It is estimated that 1.13 billion people worldwide have hypertension, largely in low and middle-income countries. In 2015, one in four men and one in five women had hypertension (WHO, 2020*c*). CVDs were responsible for approximately 17 million deaths a year, and of these, 9.4 million were complications of hypertension (WHO, 2013). At least 45% and 51% of deaths due to CVDs and strokes, respectively, could be attributed to hypertension (WHO, 2013).

The relationship between traffic noise exposure and hypertension has been demonstrated in a large number of cross-sectional and longitudinal studies (Barregard, L. et al., 2009; Bluhm, G.L. et al., 2007; Davies, H. and Van Kamp, I., 2012; Sørensen, M. et al., 2011; Stansfeld, S. and Crombie, R., 2011). Most of these studies were of cross-sectional design and based on aircraft and road traffic noise, where the evidence base for aircraft noise was considerably stronger than the one for road traffic noise. However, the results of these studies were inconsistent.

The WHO review found a positive association between the aircraft, road traffic, and railway noise and the prevalence of hypertension (Van Kempen, E. et al., 2018). A RR of 1.05 per 10 dB L_{den} was estimated for each of the traffic noise. However, the quality of the evidence was rated as very low according to the GRADE criteria because of the high risk of bias in the cross-sectional studies. The response rate in the cross-sectional studies was lower than 60% and hypertension was ascertained by means of self-report only (Van Kempen, E. et al., 2018). There was also an apparent discrepancy between the cohort studies on the impact of traffic noise on hypertension, where no increased risks of hypertension were found. Overall, the review indicated that any estimate of effect between traffic noise exposure and hypertension is very uncertain (Van Kempen, E. et al., 2018).

The *Environmental Noise Guidelines* identified one cohort study (32,635 participants) on the relationship between road traffic noise and incidence of hypertension. In addition, 26 cross-sectional studies were identified that looked at the association between road traffic noise and prevalence of hypertension (154,398 participants). A non-significant RR of 0.97 per 10 dB L_{den} increase in noise levels was found (Lindvall, T. and Radford, E.P, 1973) based on the cohort study. This indicates that road traffic noise does not contribute to the increased risk of hypertension. The evidence was rated as low quality because of the risks of bias and the availability of only one study.

One cohort study was identified that assessed the relationship between aircraft noise and hypertension in people living in Sweden (4,712 participants). Furthermore, nine cross-sectional studies assessed the prevalence of hypertension in 60,121 participants. A nonstatistically significant effect size of RR was found with inconsistency across studies.

Similarly, there was also no association found between railway noise with hypertension based on one cohort study which assessed the incidence among people living in Denmark, involving 7,249 participants and five cross-sectional studies assessing the prevalence of hypertension in 15,850 participants.

2.1.4.3 Stroke

Stroke is the sudden death of some brain cells due to the lack of oxygen when the blood flow to the brain is lost by blockage or rupture of an artery to the brain (Johnson, W. et al., 2016). Stroke is classified as both cerebrovascular and cardiovascular disease, as it is a disruption in blood supply to the brain which causes the neurological abnormalities. The WHO defines stroke as 'rapidly developing clinical signs of focal (or global) disturbance of cerebral function, with symptoms lasting 24 hours or longer or leading to death, with no apparent cause other than of vascular origin' (Truelsen, T. et al., 2000).

The link between heart disease and stroke is significant. Several types of heart disease are risk factors for stroke. Likewise, stroke is a risk factor for coronary heart disease. Annually, 15 million people worldwide suffer a stroke. Of these, five million die and another five million are left permanently disabled (MacKay, J. and Mensah, G.A., 2004). Cerebrovascular diseases (strokes) were estimated to account for 5.5 million deaths worldwide in 2001, equivalent to 9.6% of all deaths (Truelsen, T. et al., 2000).

There are relatively few studies available that investigate the impact of traffic noise on stroke compared with the number of studies on IHD and hypertension. Regarding the impact of road traffic noise on the incidence of stroke, based on one cohort study (51,485 participants), there was a statistically significant RR of 1.14 per 10 dB L_{den} (Sørensen, M. et al., 2011). In the two cross-sectional studies (14,098 participants) on the associations between road traffic noise and the prevalence of stroke and three cohort studies which investigated the relationship between road traffic noise and mortality due to stroke (581,517 participants), no increased risks of stroke were observed (Van Kempen, E. et al., 2018). The evidences between exposure to road traffic noise and incidence of stroke, and mortality due to stroke were rated as moderate quality. For the impact on the prevalence of stroke, the evidence was rated as very low quality.

No cohort or case-control studies on the relationship between aircraft noise and incidence (hospital admission) of stroke were available, but the associations between aircraft noise and an increase in the incidence of stroke were found in the two ecological studies that were conducted in cities around airports in the United Kingdom and United States of America, involving 9,619,082 participants (RR of 1.05 per 10 dB L_{den}). In the two cross-sectional studies that assessed the prevalence of stroke in 14,098 participants, an RR of 1.02 per 10 dB L_{den} was found. On the relationship between aircraft noise and mortality due to stroke, one cohort study (4,580,311 participants living in Switzerland) and two ecological studies (3,897,645 participants in the Netherlands and the United Kingdom) were identified, where an RR of 0.99 and 1.07 per 10 dB L_{den} were found, respectively (WHO-EU, 2018).

However, the WHO systematic review (Van Kempen, E. et al., 2018) concluded that none of the associations was statistically significant. No association between air traffic noise exposure and mortality due to stroke was observed in the evaluated cohort study. The evidences for the association between the exposure to aircraft noise and prevalence and incidence of stroke, and mortality due to stroke, were rated as very low quality (Van Kempen, E. et al., 2018).

As for IHD, no evidence was available on the relationship between railway noise and incidence of or mortality from stroke. However, one cross-sectional study was identified that assessed the prevalence of stroke in 9,365 participants. The overall risk was not statistically significantly increased, with RR = 1.07 per 10 dB L_{den} increase. The evidence was rated very low quality.

2.1.5 Cognitive impairment in children

According to WHO, the case definition of noise-related cognitive impairment is the reduction in cognitive ability in school-age children that occurs while the noise exposure persists and will persist for some time after the cessation of the noise exposure (WHO-EU, 2011). The number of evidence for the effects of environmental noise on the well-being and learning of children in the last decade has increased significantly. The WHO estimates that each year 45,000 DALYs are lost due to cognitive impairment in children (WHO-EU, 2011).

Children are often positioned to be in the 'vulnerable' or 'susceptible' groups where environmental noise has more significant health impacts relative to the rest of the population (Stansfeld, S. and Clark, C., 2015; Van Kamp, I. and Davies, H., 2013). A review on the studies investigating the effects of noise on vulnerable groups shows that although children are less vulnerable for annoyance and awakenings due to noise, they are more vulnerable for physiological effects during sleep and for cognitive effects of noise than adults (Van Kamp, I. and Davies, H., 2013). They are more at risk because of the less-developed coping strategies, and are in a sensitive developmental period, which is more indicative of a life phase rather than age effect (Van Kamp, I. and Davies, H., 2013). The tasks affected are those involving central processing and language comprehension, such as reading, attention, problem solving, and memory (Cohen, S. et al., 1980; Evans, G.W. and Lepore, S.J., 1993; Stansfeld, S. and Clark, C., 2015). Exposure during critical periods of learning at school could potentially impair development and have a lifelong effect on educational attainment (WHO-EU, 2011).

There are also studies that suggest the effects of noise in children may not only be a direct effect of exposure, but also the result of a decrease in sleep quality caused by nighttime noise exposure (van Kamp, I. et al., 2015; WHO-EU, 2009). In the *Night Noise Guidelines*, it has been suggested that nighttime exposure levels above 40 dB may severely affect vulnerable groups (WHO-EU, 2009). Environmental sleep disorder (of which sleep disturbance is an example) may result in secondary deficits, including in concentration, attention, and cognitive performance (WHO-EU, 2009). There is a gap of knowledge on the impacts of nighttime noise on the cognitive impairment in children as most studies focused on the effects during daytime at school.

The WHO-EU systematic review (Clark, C. and Paunovic, K., 2018) assesses the quality of evidence on the effect of environmental noise and cognition. Quantitative non-experimental studies of the association between environmental noise exposure on child and adult cognitive performance published up to June 2015 were reviewed: no limit was placed on the start date for the search.

A total of 34 from 1,032 papers were identified to be included in the narrative systematic review, all of which were of child populations. 82% of the papers were of cross-sectional design, with fewer studies of longitudinal or intervention design. A range of cognitive outcomes were examined. The quality of the evidence across the studies for each individual noise source and cognitive outcome was assessed using an adaptation of GRADE methodology (Clark, C. and Paunovic, K., 2018).

The review identified two papers that reported the results of the cross-sectional road traffic and aircraft noise exposure and children's cognition and health (RANCH) study, which examined exposure–effect relationships. The study of over 2000 children aged 9–10 years, attending 89 schools around three major airports in the Netherlands, Spain, and the United Kingdom did not find an exposure–effect relationship between road traffic noise exposure at primary school and children's reading comprehension (WHO-EU, 2018).

Few studies have investigated other health outcome measures related to cognition. Evidence rated low quality was available for an association between road traffic noise and cognitive impairment assessed through standardised tests. There was evidence rated very low quality for an association between road traffic noise and long-term memory. No studies examined effects on short-term memory. There was evidence rated very low quality, however, that road traffic noise does not have a considerable effect on children's attention. Further, there was evidence rated low quality that road traffic noise does not have a substantial effect on executive function (working memory), with studies consistently reporting no association (WHO-EU, 2018).

Studies of railway noise on children's reading and oral comprehension were lacking. Nevertheless, other measures of cognition yielded evidence rated very low quality for an association between railway noise and children with poorer performance on standardized assessment tests. Evidence for the association between railway noise and children having poorer long-term memory was rated very low quality. No studies examined effects on short-term memory. There was no clear relation between railway noise and attention in children and this evidence was rated very low quality (WHO-EU, 2018).

Evidence rated moderate quality was available for an association between aircraft noise and reading and oral comprehension, assessed by standardized tests. This is based on a narrative review of 14 studies that examined aircraft noise exposure effects on reading and oral comprehension. Of these studies, ten were cross-sectional, and only four had a longitudinal and/or intervention design. Most of the studies (10 of 14) demonstrated a statistically significant association or at least demonstrated a trend between higher aircraft noise exposure and poorer reading comprehension (WHO-EU, 2018).

The review concluded that evidence across studies is of low quality. This is due to the limitation by the low number of studies, in particular road traffic and railway noise (Clark, C. and Paunovic, K., 2018). However, this does not necessarily means that there are no effects; rather, that more robust and a greater number of studies are required (Clark, C. and Paunovic, K., 2018; WHO-EU, 2018).

2.1.6 Metabolic effects

Exposure to environmental noise affects the physiological, metabolic, and immunological functions. Long-term noise exposure increases the risk of metabolic outcomes, including obesity and type 2 diabetes (Münzel, T. et al., 2018). The suggested mechanisms include an effect of noise-induced stress and disturbance of sleep on appetite regulation, changes in the glucose regulation, insulin levels, and insulin sensitivity (Münzel, T. et al., 2018; Sørensen, M. et al., 2013).

2.1.6.1 Obesity

The WHO defines overweight and obesity as abnormal or excessive fat accumulation that may impair health (WHO, 2020*d*). Obesity occurs when the body's energy balance is positive (i.e. when energy intake exceeds energy expenditure) (Hill, J.O. et al., 2013). The worldwide prevalence of obesity nearly tripled between 1975 and 2016 (WHO, 2020*d*). In 2016, more than 1.9 billion adults (18 years and older) were estimated to be overweight. Of these, over 650 million were obese. Moreover, at least 3.4 million deaths per year worldwide, 4% of YLL and at least 4% of DALYs all around the world may be attributed to overweight or obesity (Ng, M. et al., 2014).

The WHO-EU systematic review (Van Kempen, E. et al., 2018) evaluated four studies that investigated the impact of aircraft and road traffic noise on obesity markers, e.g. Body Mass Index (BMI) and waist circumference. All the studies showed that an increase in traffic noise was associated with an increase in obesity markers, although, according to one study, this was present only in certain subgroups (Van Kempen, E. et al., 2018). For road traffic noise, the review reported non-significant combined estimates of 0.03 kg/m^2 per 10 dB L_{den} for BMI and 0.17 cm per 10 dB L_{den} for waist circumference. In the cohort study on the association of aircraft noise and obesity markers, an increase of 10 dB L_{den} was associated with a significant increase of 3.46 cm in waist circumference and 0.14 kg/m^2 for BMI during 8 to 10 years of follow-up.

Although the findings suggest that traffic noise may be associated with obesity markers, however, the WHO systematic review rated the quality of the evidence as low and indicated a need for further research (Van Kempen, E. et al., 2018).

2.1.6.2 Diabetes

According to WHO, diabetes is a chronic disease that occurs either when the pancreas does not produce enough insulin (type 1) or when the body could not use the insulin it produces effectively (type 2). Type 2 diabetes is often milder than type 1 and results from the body's ineffective use of insulin and is largely the result of excess body weight and physical inactivity (WHO, 2020*b*). The number of people with diabetes rose from 108 million in 1980 to 422 million in 2014. In 2016, an estimated 1.6 million deaths were directly caused by diabetes.

The WHO systematic review (Van Kempen, E. et al., 2018) evaluated four studies on the association between traffic noise and the risk of type 2 diabetes. For air traffic noise, the review found a non-significant RR of 1.01 per 10 dB L_{den} for the association of air traffic noise and the prevalence of diabetes. Moreover, no increased risk of the incidence of diabetes was found. For road traffic noise, the two evaluated cross-sectional studies showed a non-significant trend on the prevalence of diabetes and a RR of 1.08 per 10 dB L_{den} was estimated. The studies on the association of rail traffic noise and the prevalence and incidence of diabetes showed no increased risk of diabetes (Van Kempen, E. et al., 2018).

According to the GRADE criteria, the quality of the evidence supporting the association between traffic noise and diabetes was rated as low and indicated a need for further research (Van Kempen, E. et al., 2018).

2.2 Noise policies, guidelines recommendations, and standards

2.2.1 Noise policy recommended by WHO (1999)

The *Guidelines for Community Noise*, which was published in 1999 by WHO was thought as the initial key document linking noise exposure to public health concerns (Murphy, E. and King, E. A., 2014). This document was instrumental in identifying noise pollution as a significant public health problem. According to the document, 40% and 30% of the population of EU countries was exposed to road traffic noise exceeding 55 dB during daytime and nighttime, respectively.

This document, together with the *6th Environmental Action Programme of the European Community* and academic research on noise and health relationships have been instrumental in the development of a legislative framework for the management of environmental noise in Europe (Murphy, E. and King, E. A., 2014). In response to a growing evidence associating excessive noise pollution with health effects, at the EU level, Directive 2002/49/EC, also known as the Environmental Noise Directive (END) was approved (European Union, 2002).

2.2.2 Noise policy recommended by WHO Europe (2009, 2011, 2018)

There have been several guidelines since the END on environmental noise and health, i.e. the *Night Noise Guidelines for Europe* in 2009, which reviews the health effects of exposure to nighttime noise, examines dose-effect relations, and presents interim and ultimate guidelines values for exposure (WHO-EU, 2009) and the *Burden of Disease from Environmental Noise* in 2011, which summarises the evidence on the relationship between environmental noise and health effects (WHO-EU, 2011).

The *Night Noise Guidelines for Europe* recommended a nonbinding limit value of 40 dB $L_{\text{night, outside}}$ to prevent from being exposed to the harmful effects of environmental noise pollution. The document also suggested an interim value of 55 dB $L_{\text{night, outside}}$ if there are authorities that are not able to adopt the initial 40 dB $L_{\text{night, outside}}$ limit value. The 55 dB $L_{\text{night, outside}}$ can be implemented until it becomes more feasible for the authorities to adopt the recommended limit value (WHO-EU, 2009).

The *Burden of Disease from Environmental Noise* quantified for the first time the nature and extent of the disease burden from environmental noise exposure across the EU. The document highlights the seriousness of noise pollution as a public health problem and that, noise exposure is actually rising in Europe and worldwide, as opposed to the trend for other environmental stressors (e.g. second hand smoke, dioxins and benzene), which are declining (WHO-EU, 2011).

The methodology in the document consist of calculating the burden of disease on the basis of the exposure-response relationship (derived from existing epidemiological studies or metaanalysis of published results), exposure distribution, population-attributable fraction, background prevalence of disease and disability weights (DWs) of the outcome (Murphy, E. and King, E. A., 2014). The incidence or prevalence of the health outcome in a population can be obtained by the national health statistics or surveys of the population. The population-attributable fraction is the proportion of disease in the population that is estimated to be caused by environmental noise. DW factors were used to reflect the severity of the disease on a scale from 0 (representing perfect health) to 1 (representing most imperfect health, i.e. death) (Murphy, E. and King, E. A., 2014; WHO-EU, 2011). The summary of the study is shown in Table 2.1 (see Page 8).

The most recent guidelines are the *Environmental Noise Guidelines for the European Region* (WHO-EU, 2018). These guidelines are developed at the request of Member States at the Fifth Ministerial Conference on Environment and Health in Parma, Italy, in March 2010, based on the growing understanding of the health impacts of exposure to environmental noise. These WHO guidelines – the first of their kind globally – lay out recommendations for protecting human health from exposure to environmental noise originating from various sources.

The *Environmental Noise Guidelines for the European Region* provides guidance on protecting human health from harmful exposure to environmental noise by setting health-based recommendations on average environmental noise exposure of five relevant sources of environmental noise (road traffic noise, railway noise, aircraft noise, wind turbine noise and leisure noise) (WHO-EU, 2018). The guidelines are based on the scientific evidence of health effects and an assessment of achievable noise levels. The robust public health advice provided by the document is essential to drive policy action that will protect communities from the adverse effects of noise (WHO-EU, 2018).

The current guidelines differ from the older ones, recommending levels of exposure unlike those previously outlined (especially by the *Night Noise Guidelines for Europe*). The document also provides stronger evidence of cardiovascular and metabolic effects of environmental noise compared to previous WHO guidelines.

The guidelines recommend the following thresholds for the transportation noise as the noise above the suggested levels is associated with adverse health effects (WHO-EU, 2018):

- Road traffic noise – 53 dB L_{den}, 45 dB L_{night}
- Railway noise – 54 dB L_{den}, 44 dB L_{night}
- Aircraft noise – 45 dB L_{den}, 40 dB L_{night}

2.2.3 Noise policy in Japan

In terms of environmental noise researches, policy, and legislation, the EU is regarded as the world leader. Be that as it may, noise is a worldwide problem. The motorisation rate has evolved and the gradual increase in the proportion of people living in the urban areas is a growing concern everywhere, including in Japan.

Based on the data from 1,877 continuous monitoring points, in 1993–1997, the percentage of areas that do not comply with the Japanese Environmental Quality Standards for road traffic noise has risen to more than 87% (Ministry of the Environment, 2019). Meanwhile, motorisation has greatly progressed and the number of motor vehicles owned has increased at almost the same rate and may lead to an increase in noise levels. A statistics published in April 2019 revealed that the total number of motor vehicles owned in Japan is approximately 61.8 million, a 1.7 million increase in a five-year gap, and a giant 60 million leap from its rapid industrial development in the decade from 1965 (Automobile Inspection & Registration Information Association, 2019). In local news, noise pollution is rising as one of the top citizen complaints to the Environment Dispute Coordination Commission, a government organisation that oversees environmental disputes (Terzuolo, C., 2019).

The Japanese government had enacted two basic laws, i.e. the Basic Law for Environmental Pollution Control, enacted in 1967 to combat the serious industrial pollution, which overrode Japan in the period of rapid economic growth in the late 1950's and 1960's and the Nature Conservation Law, enacted in 1972 to stop the destruction of outstanding features of the natural environment. These laws became insufficient in dealing with the new global environmental and urban pollution issues. To address these issues, the Basic Environment Law was enacted to replace the preceding laws in 1968 (amended in 1993) (Ministry of the Environment, 2020*d*).

Regarding the environmental noise problem, the national government enacted the Noise Regulation Law in 1968 (amended in 1993) to regulate the noises from factories and construction over considerable ranges, and set the maximum permissible levels of motor vehicle noise (Ministry of the Environment, 2020*d*). Besides the regulation law, in accordance with the provisions of the Basic Environment Law, the Environmental Quality Standards were legislated for road traffic, aircraft, and Shinkansen Superexpress railway noise (Ministry of the Environment, 2020*d*). Other standards for transportation noise are the Guideline of Noise Measures for Conventional Railways (non-Shinkansen lines) and the Guideline for the Preservation of Living Environment around Small Airfields (Ministry of the Environment, 2020*e*).

The standard values provided in the Environmental Quality Standards for the space adjacent to a road carrying arterial traffic are 73 dB L_{den} and 65 dB L_{night} (Ministry of the Environment, 2020*b*). The L_{den} was translated from the daytime standard values using the derived conversion rule by Brink, M. et al. (2018). For aircraft noise, the standards are established for two area categories, Category I refers to areas used exclusively for residential purpose and Category II refers to other areas where the normal living conditions shall be preserved. The standard values are 70 dB and 75 dB (in WECPNL=Weighted Equivalent Continuous Perceived Noise

Level), for Category I and Category II, respectively (Ministry of the Environment, 2020*a*). Meanwhile, for the Shinkansen Superexpress railway noise, the standards are established for two area categories as well, Category I refers to areas used mainly for residential purpose and Category II refers to other areas, including commercial and industrial areas, where the normal living conditions shall be preserved. The standard values are 70 dB and 75 dB, for Category I and Category II, respectively (Ministry of the Environment, 2020*c*).

Nonetheless, the standard values are 30 years outdated, as opposed to the multiple guidelines in the EU that had been continuously updated. Furthermore, unlike the universal L_{den} and L_{night} noise indicators proposed by the EU, the specified noise indicators in Japan are different for each standard. In general, depending on the noise sources and relevant legislation in a country, the noise indicators can take different forms. However, the problems with the noise indicators might arise when comparison between studies is to be achieved.

2.2.4 Risk communication with strategic noise maps according to the END

The END is the main EU instrument to identify noise pollution levels and trigger necessary actions both at the member states and at the EU levels. Overall, the END focuses on four core areas; (1) strategic noise mapping, (2) population exposure estimation, (3) noise action planning and (4) dissemination of results (European Union, 2002).

The END aims to 'define a common approach intended to avoid, prevent or reduce on a prioritized basis the harmful effects, including annoyance, due to exposure on environmental noise' (European Union, 2002). The END states several actions that need to be progressively implemented by the member states (European Union, 2002; Guarinoni, M. et al., 2012):

- Monitoring of environmental noise – Member States must develop strategic noise maps in order to estimate the level of population and/or building exposure to environmental noise in priority areas in their jurisdications;

- Managing environmental noise issues – on the basis of the developed strategic noise maps, Member States must adopt action plans containing measures designed to address noise issues, including noise prevention/reduction and preserving sound quality where it is deemed to be good;

- Public information and consultation – strategic noise maps, action plans, and relevant information about noise exposure, its effects and measures considered to address environmental noise issues should be made available to the public or developed in consultation with the public;

- Development of a long-term EU strategy – with a view to reduce noise emitted by the major sources (in particular road and rail vehicles and infrastructure, aircraft, outdoor, and industrial equipment and mobile machinery), the EU and Member States should cooperate in order to provide a framework for EU policies addressing environmental noise issues.

The END requires the Member States to develop strategic noise maps for all major roads, railways, airports, and agglomeration on a five-year basis. In the first phase, the strategic noise maps were recommended for all agglomerations with more than 250,000 inhabitants, major roads with more than 6 million vehicle passages a year, railways with more than 60,000 train passages a year, and major airports with more than 50,000 movements a year within the territories. The second phase of noise mapping saw a reduction in the thresholds; major roads were defined as roads with more than three million vehicles, major railways were defined as railways with more than 30,000 trains, and agglomerations were described as areas with more than 100,000 inhabitants, while the criteria for airports remained unchanged (Van Kamp, I. and Davies, H., 2013).

Noise prediction is gaining importance, along with the strategies to manage and reduce noise exposure. Until a few years ago the normal procedure was to act on the existing sources. At present, proactive planning is the way forward. The use of predictive models allows the estimation of noise exposure levels in the study areas, as well as allowing the development of scenarios in order to find the best solution. Strategic noise mapping will be used to provide citizens with information on noise exposure and as a basis for the END action plans to prevent and reduce environmental noise, particularly where exposure levels can harm human health (Alférez, J.R. et al., 2013).

For environmental noise research, mapping is an extremely important part of quantifying and visualising noise pollution levels to monitor the noise effects on the environment (de Kluijver, H. and Stoter, J., 2003). A noise map is a graphical representation of sound level distribution in an area, for a defined period. Defined broadly, noise mapping is the presentation of data on one of the following aspects: a noise situation in terms of the noise indicators, i.e. L_{den} and L_{night}, the exceeding of a limit value, the estimated number of dwellings that are exposed to specific values of a noise indicator, and the estimated number of people exposed to noise (Alférez, J.R. et al., 2013).

L_{night} is the yearly average noise indicator for night-time A-weighted equivalent sound pressure level, while L_{den} is the yearly average noise indicator for day-evening-night-time A-weighted equivalent sound pressure level and is given by the following equation:

$$L_{\text{den}} = 10 \times \log\frac{1}{24}(12 \times 10^{L_{\text{day}}/10} + 4 \times 10^{L_{\text{evening}}+5/10} + 8 \times 10^{L_{\text{night}}+10/10}) \tag{2.9}$$

The day period is 12 hours, evening is four hours, and night is eight hours. The day, evening, and night time periods might differ depending on countries, however, the periods are generally taken to be from 0700–1900, 1900–2300, and 2300–0700, respectively. The weighting factors in the above equation are designed to account for the increase in annoyance at different periods throughout the entire day, hence, the addition of 10 to the value for L_{night} and 5 to the value of L_{evening} (Murphy, E. and King, E. A., 2014).

Figure 2.6 presents a schematic of the noise mapping process. It comprises data collection, noise calculation, validation and mapping, estimation of population exposed, noise action planning, and public dissemination. In general, the main data required for noise mapping are the annual traffic flow for individual sources, building height and geometry information, and local meteorological and topographical (ground elevation) information. For road traffic noise, the data are related to the sound power characteristics of each vehicle type, traffic speed, and road surface.

The calculations for noise levels were carried out using the commercial software programmes, which contain the noise prediction models. There are a number of noise prediction models available for different types of noise sources with different specifications to predict noise levels at specific receiver points. Table 2.2 shows the national/available standards in various countries, inside and outside the EU. Each calculation method was originally developed to the specific conditions or legislation as applied in each nation long before the END was established (Murphy, E. and King, E. A., 2014).

In Japan, making noise maps using the ASJ RTN-Model is not possible because the reflected sounds on buildings and the sound power level at each frequency band are not considered in the model. Meanwhile, in Europe, due to the heterogeneity between each model, it can be problematic to obtain comparable results across the EU (Kephalopoulos, S. et al., 2014; European Environment Agency, 2014). Therefore, the END undertook the development to establish a common noise assessment methods (the Common NOise aSSessment mEthOdS in Europe–CNOSSOS-EU) (European Union, 2015) that represents a harmonised and coherent approach

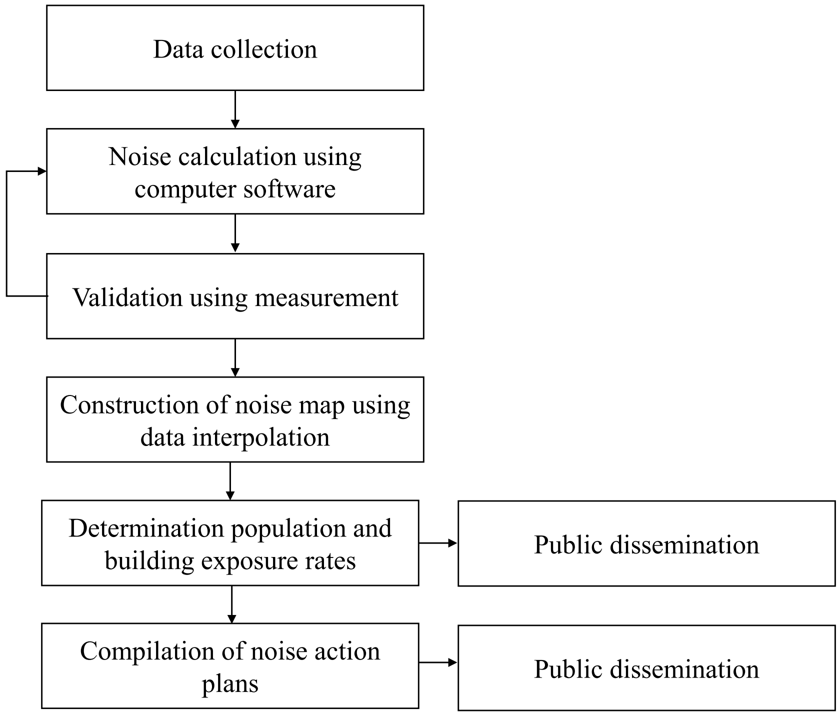

Figure 2.6: Schematic of the noise mapping process, redrawn from (Murphy, E. and King, E. A., 2014)

Table 2.2: Principal models in various countries (Murphy, E. and King, E. A., 2014; Steele, C., 2001)

Principal model	Government users
RLS 90	Germany
ASJ RTN-Model 1993	Japan
NMPB 2008	France, Europe
CoRTN	UK, Australia, Hong Kong, New Zealand
STL-86	Switzerland
RVS 3.02	Austria
01 dB MITHRA	France, Belgium
FHWA (TNM)	USA

to assess noise levels from the main noise sources (road, rail and aircraft traffic, and industrial noise) (Kephalopoulos, S. et al., 2012). The END requires its Member States to use the CNOSSOS-EU methods from 31 December 2018 onwards (European Union, 2015). This new methodology is a welcome development because it offers a solution to the inconsistent noise mapping undertaken in the past. Furthermore, comparability of the results can be achieved, which will facilitate the implementation of common noise policies.

In the assessment of population exposure to noise, the number of inhabitants of a residential building is an intermediate parameter for the estimation of the exposure to noise and can be estimated either on the basis of dwelling units, or on the average dwelling floor space per inhabitant (European Union, 2015). The assessment of population exposure to noise is based on receiver point levels at 4 m above the ground in front of building façades of residential buildings (European Union, 2015).

The results from the noise maps are precursors to adopt noise action plans so that the exposure to excessive noise can be reduced. The information on the noise action plans and the assessment of population exposure to noise should be disseminated to the general public. The key issue concerning the dissemination of information to the public relates primarily to the method of dissemination. The methods used at present are mainly online availability of strategic noise maps and associated noise action plan (Murphy, E. and King, E. A., 2014). It is worth considering other innovative strategies to help raise environmental noise awareness.

One of the strategies to raise environmental awareness is by communicating the risks to the public. Risk communication is important to raise public awareness of the health effects associated with environmental noise and may also serve in gaining public support for appropriate control measures. The WHO defines risk communication as a two-way exchange of real-time information, advice, and opinions between experts and people facing threats to their health, economic, or social well-being (WHO-EU, 2020). One of the key messages regarding risk communication is that the information needs to be appropriately structured and understood by the general public. Information quality, transparency, simplicity, and coherence of the message are the essential elements for effective risk communication (WHO-EU, 2013).

Strategic noise maps may serve multiple purposes. Nevertheless, the estimations of sound level from strategic noise maps are not useful in communicating risks with the general public because the health effects of environmental noise are not explicitly shown. Noise map only shows acoustic intensity instead of health effects. Therefore, explaining the risks using noise maps will be challenging for effective risk communication. The health risks are 'hidden' as the residents are unfamiliar with the relationships between sound levels and health risks. Local residents will therefore find noise maps incomprehensible as they are unable to recognise the quantitative health impacts from an ordinary noise map. An alternative tool (i.e. maps) that displays the health risks transparently will be valuable for enhancing risk communication.

The use of maps to convey information on the environmental burden and health risks is increasing (Dransch, D. et al., 2010; Lahr, J. and Kooistra, L., 2010; Stieb et al., 2019), e.g. mortality risk of radiation exposure (Moen, J.E.T and Ale, B.J.M, 1998) and asthma and cancer risk in relation to air pollutants (Hammond, D. et al., 2011; Severtson, D.J. and Myers, J.D., 2013). In the environmental noise field, thus far, two studies that illustrated health risks in maps were found. First, a study in a densely-populated city in Korea with the aim at providing an overview of the health impacts of road traffic noise had depicted a map showing the population estimates of individual buildings with annoyance and sleep disturbance (Park, T. et al., 2018). Second, a study which addressed the limitation of health data using statistical downscaling approach for making health risk maps had generated a risk map of IHD mortality due to road traffic noise in Melbourne, Australia (Hanigan, I.C. et al., 2019).

These studies show that strategic noise maps can be utilised to make health risk maps by combining road traffic noise exposures and population health outcomes data. The risk maps give visual representations with numerical information of specific adverse health risks that might affect the community.

2.3 Methodological framework of the CNOSSOS-EU

2.3.1 Noise emission model

The CNOSSOS-EU is recommended in Europe for environmental noise prediction. In regard to road traffic, it includes vehicle noise emission models implicitly referring to internal combustion vehicles (Kephalopoulos, S. et al., 2012, 2014). In the CNOSSOS-EU framework, the vehicles are grouped into five separate categories:

- Category 1: Light motor vehicles (passenger cars, delivery vans 3.5 tons)
- Category 2: Medium heavy vehicles (medium heavy vehicles, delivery vans > 3.5 tons, buses, etc. with two axles and twin tyre mounting on rear axle)

- Category 3: Heavy vehicles (heavy duty vehicles, touring cars, buses, with three or more axles)
- Category 4: Powered two-wheelers (mopeds, motorcycles)
- Category 5: Open category (future needs)

The noise emission of a traffic flow is represented by a source line characterised by its directional sound power. The sum of the sound emission, or sound power of a single vehicle, determined by the vehicle flow, is defined by Equation 2.10 (Kephalopoulos, S. et al., 2012, 2014):

$$L_{\mathrm{W',eq,line,i,m}} = L_{\mathrm{W,i,m}}(v_{\mathrm{m}}) + 10 \times \log(\frac{Q_{\mathrm{m}}}{1000 \times v_{\mathrm{m}}}) \tag{2.10}$$

where $L_{\mathrm{W',eq,line,i,m}}$ is the sound power per metre per frequency band, Q_{m} is the steady traffic flow of vehicles of category *m* per hour, and v_{m} is the average speed (km/h) and set as the maximum legal speed of the vehicle category.

The CNOSSOS-EU emission model consists of two main noise sources: rolling noise due to the tyre/road interaction, and propulsion noise produced by the driveline of the vehicle. For Category 1, 2 and 3, the total sound power corresponds to the energetic sum of the noises (Equation 2.11):

$$L_{\mathrm{W,i,m}}(v_{\mathrm{m}}) = 10 \times \log(10^{L_{\mathrm{WR,i,m}}(v_{\mathrm{m}})/10} + 10^{L_{\mathrm{WP,i,m}}(v_{\mathrm{m}})/10}) \tag{2.11}$$

where $L_{\mathrm{WR,i,m}}$ and $L_{\mathrm{WP,i,m}}$ is the sound power level for rolling noise and propulsion noise, respectively. The rolling noise sound power level for each octave frequency band *i* for a vehicle of class *m* = 1, 2, or 3 is defined in Equation 2.12 and Equation 2.13:

$$L_{\mathrm{WR,i,m}} = A_{\mathrm{R,i,m}} + B_{\mathrm{R,i,m}} \times \log(\frac{v_{\mathrm{m}}}{v_{\mathrm{ref}}}) + \Delta L_{\mathrm{WR,i,m}}(v_{\mathrm{m}}) \tag{2.12}$$

$$\Delta L_{\mathrm{WR,i,m}} v_{\mathrm{m}} = \Delta L_{\mathrm{WR,road,i,m}} v_{\mathrm{m}} + \Delta L_{\mathrm{studdedtyres,i,m=1}} v_{\mathrm{m}} + \Delta L_{\mathrm{WR,acc,i,m}} + \Delta L_{\mathrm{W,temp}}(\tau) \tag{2.13}$$

The coefficients $A_{\mathrm{R,i,m}}$and $B_{\mathrm{R,i,m}}$ are given in 125 Hz to 4 kHz octave bands for each vehicle category and for a reference speed v_{ref} = 70 km/h. $\Delta L_{\mathrm{WR,road,i,m}}$ is the correction for the effect on rolling noise of a road surface with acoustic properties different from the virtual reference surface. The type of road surface significantly influences the noise emission of a vehicle and this correction coefficient should be applied when a road surface different from the reference is considered (Kephalopoulos, S. et al., 2012, 2014).

$\Delta L_{\text{studdedtyres,i,m=1}} v_{\text{m}}$ is the correction coefficient for the proportion of light vehicles equipped with studded tyres, which must be taken into account when a significant number of light vehicles in the traffic flow use studded tyres during several months every year. $\Delta L_{\text{WR,acc,i,m}}$ accounts for the effect on rolling noise of a crossing with traffic lights or a roundabout. In the development of strategic noise maps, the effect of acceleration and deceleration may be neglected because the uncertainty on the estimation of acceleration of the traffic can be higher than the effect on noise. $\Delta L_{\text{W,temp}}(\tau)$ is the correction term for the average temperature different from the reference temperature $\tau_{\text{ref}} = 20°\text{C}$. Rolling noise decreases when the air temperature increases (Kephalopoulos, S. et al., 2012, 2014).

The propulsion noise emission produced by the vehicle's driveline includes engine, exhaust, gears, air intake, etc. The propulsion noise sound power level in each octave frequency band *i* for a vehicle of class *m* is defined in Equation 2.14 and Equation 2.15:

$$L_{\text{WP,i,m}} = A_{\text{P,i,m}} + B_{\text{P,i,m}} \times \frac{(v_{\text{m}} - v_{\text{ref}})}{v_{\text{ref}}} + \Delta L_{\text{WP,i,m}}(v_{\text{m}}) \tag{2.14}$$

$$\Delta L_{\text{WP,i,m}}(v_{\text{m}}) = \Delta L_{\text{WP,road,i,m}}(v_{\text{m}}) + \Delta L_{\text{WP,acc,i,m}} + \Delta L_{\text{WP,grad,i,m}}(v_{\text{m}}) \tag{2.15}$$

The coefficients $A_{\text{P,i,m}}$ and $B_{\text{P,i,m}}$ are given in the octave bands for each vehicle category and for a reference speed, v_{ref} = 70 km/h. $\Delta L_{\text{WP,i,m}}(v_{\text{m}})$ is the sum of the correction coefficients to be applied on the propulsion noise for conditions that are different from the reference conditions. $\Delta L_{\text{WP,road,i,m}}$ is the correction for the effect on propulsion noise of the type of road surface, $\Delta L_{\text{WP,acc,i,m}}$ and $\Delta L_{\text{WP,grad,i,m}}$ account for the deviations related to the driving conditions, where the former is the correction of the effect of acceleration and deceleration of vehicles and the latter is the effect of road gradient on the propulsion noise. Road gradient affects the vehicle speed and thus, the rolling and propulsion noise emission of the vehicle. It also affects both the engine load and speed via the choice of gear and therefore, the propulsion noise of the vehicle (Kephalopoulos, S. et al., 2012, 2014).

2.3.2 Noise propagation model

The CNOSSOS-EU specifies a method for calculating the attenuation of noise during its outdoor propagation. Knowing the characteristics of the source, this method helps to determine the equivalent continuous sound pressure level at a receiver point which applies to industrial infrastructures and land transport infrastructure.

The calculations of a receiver ***R*** in the noise propagation model of CNOSSOS-EU are made according to the following steps (Kephalopoulos, S. et al., 2012, 2014):

- breakdown of the noise sources into point sources, if not already expressed as point sources;
- determination of the directional sound power per frequency band of each source;
- calculation of the probability of occurrence of favourable conditions for each direction source S_{i} to receiver ***R*** (S_{i},***R***);
- search for propagation paths between each source and receiver: direct, reflected, and/or diffracted paths;
- on each propagation path:
  – calculation of the attenuation in favourable conditions;
  – calculation of the attenuation in homogeneous conditions;
  – calculation of the occurrence in favourable conditions;
  – calculation of the long-term sound level for each path;
- accumulation of the long-term sound levels for each path, therefore, allowing the total sound level to be calculated at the receiver point.

It should be noted that only the attenuations due to the ground effect (A_{ground}) and diffraction (A_{dif}) are affected by meteorological conditions.

For a point source *S* of directional sound power $L_{\mathrm{W,0,dir}}$ and for a given frequency band, the equivalent continuous sound pressure level at a receiver point *R* (depending on the atmospheric conditions) can be obtained according to the equations (Equation 2.16 and Equation 2.17) given below (Kephalopoulos, S. et al., 2012, 2014).

$$X = \begin{cases} F, \text{ for favourable conditions} \\ H, \text{ for homogenous conditions} \end{cases} \tag{2.16}$$

$$L_{\mathrm{X}} = L_{W,0,\mathrm{dir}} - A_{\mathrm{X}} \tag{2.17}$$

The term A_{X} represents the total attenuation along the propagation path, and is broken down as follows (Equation 2.18):

$$A_{\mathrm{X}} = A_{\mathrm{div}} + A_{\mathrm{atm}} + A_{\mathrm{boundary,X}} \tag{2.18}$$

where A_{div} is the attenuation due to geometrical divergence; A_{atm} is the attenuation due to atmospheric absorption; $A_{\mathrm{boundary,X}}$ is the attenuation due to the boundary of the propagation medium in favourable/homogenous conditions. It may contain the following terms: $A_{\mathrm{ground,X}}$ which is the attenuation due to the ground in favourable/homogenous conditions and $A_{\mathrm{dif,X}}$ which is the attenuation due to diffraction in favourable/homogenous conditions.

For a given path and frequency band, the following two scenarios are possible:

- either $A_{\mathrm{ground,X}}$ ($A_{\mathrm{dif,X}}$ = 0 dB) is calculated with no diffraction and $A_{\mathrm{boundary,X}}$ = $A_{\mathrm{ground,X}}$,
- or $A_{\mathrm{dif,X}}$ ($A_{\mathrm{ground,X}}$ = 0 dB) is calculated. The ground effect is taken into account in the equation itself. This therefore gives $A_{\mathrm{boundary,X}}$ = $A_{\mathrm{dif,X}}$.

The equations for obtaining the long-term sound level at point *R* for a path (*S,R*) (L_{LT}), for all paths ($L_{\mathrm{tot,LT}}$), and in decibels A (dBA) ($L_{\mathrm{Aeq,LT}}$) are Equation 2.19, Equation 2.20, and Equation 2.21, respectively:

$$L_{\mathrm{LT}} = 10 \times \log(p \times 10^{L_{\mathrm{F}}/10} + (1-p) \times 10^{L_{\mathrm{H}}/10}) \tag{2.19}$$

$$L_{\mathrm{tot,LT}} = 10 \times \log\left(\sum_n 10^{L_{\mathrm{n,LT}}/10}\right) \tag{2.20}$$

$$L_{\mathrm{Aeq,LT}} = 10 \times \log \sum_i 10^{(L_{\mathrm{tot,LT,i}} + AWC_{\mathrm{f,i}})/10} \tag{2.21}$$

where, *p* is the mean occurrence of favourable conditions in percentages, *n* is the index of paths between *S* and *R*, *i* is the index of the frequency band, and *AWC* is the A-weighting correction according to the international standard IEC 61672:2003 (Kephalopoulos, S. et al., 2012, 2014).

2.4 Adoption of electric vehicles as a noise mitigation approach

2.4.1 Noise mitigation measures

The noise action plans refer to plans that are designed to manage noise issues and effects, including noise reduction, if necessary. The plans aim to prevent and reduce environmental noise where necessary and particularly where exposure levels can induce harmful effects on human health, and preserve environmental noise quality where it is good (European Union, 2015). To reduce environmental noise, one can consider to control at the noise sources, reduce noise at the transmission path, or implement protection at the receiver end.

Reducing noise at its source is often the primary consideration and considered to be the most effective measure for noise control and regulation. Some of the source-based mitigation measures are legislation, low-noise road/rail surfaces and maintenance, traffic management (reductions in traffic speeds and traffic volume, particularly volume of heavy vehicles), low-noise tyres and low-noise vehicles and also driver behaviour. The biggest source of environmental noise is road traffic (Blanes, N. et al., 2016), more than the rail and aircraft sources combined together (Murphy, E. and King, E. A., 2014). This section deals with the noise mitigation approaches at the source for road traffic noise using engineering noise control measures.

The main sources of road noise are rolling noise and propulsion noise. Rolling noise is the interaction between the vehicle tyre and road surface that generates noise, whereas propulsion noise is the vehicle engine and transmission noise. Low-noise road surfaces provide an optimal solution for reducing rolling noise. The most effective road surfaces for reducing traffic noise pollution are porous and thin-layer asphalt (Murphy, E. and King, E. A., 2014). The noise reduction effects differ depending on the void content and pavement group. Over 6 dB reductions can be achieved using the most absorptive and open porous surfaces (Kropp, W. et al., 2007).

Another way of reducing rolling noise is by replacing tyres with quieter alternatives. Low-noise tyres could reduce noise emissions by approximately 3 dB (Kropp, W. et al., 2007). Adding a porous tread (the texture of the rubber exterior that contacts the ground) to the tyre design could reduce noise emissions by 5 dB (Science Communication Unit, UWE, Bristol, 2017). The benefits of low noise tyres are amplified when applied on low-noise road surfaces (Science Communication Unit, UWE, Bristol, 2017).

In respect of traffic management, there are several studies that suggest banning private cars in certain areas (King, E.A. et al., 2011; Nieuwenhuijsen, M.J. and Khreis, H., 2016). There is also an increasing number of cities that are planning to become partly private car free, mainly focussing on the reduction of private car use in the city centres. Reduction of noise can also be achieved by reducing the speed limits. A study found that 10% and 20% speed reductions resulted in 2.0% and 3.7% reductions, respectively, in exposure above 40 dB L_{night} (Murphy, E. and King, E.A., 2011). However, reductions achieved by implementing speed reductions are much less than those for travel demand reduction (Murphy, E. and King, E.A., 2011).

Other suggested traffic management measures are 'car-free days', which restrict the number of parking spaces in the city centre and investing in cycling infrastructure and public transport, one-way streets, restricting access to heavy vehicles in residential areas (Nieuwenhuijsen, M.J. and Khreis, H., 2016), and traffic calming measures, such as speed bumps (Murphy, E. and King, E. A., 2014).

The majority of vehicles on the road today are conventional vehicles that are powered by internal combustion engines. The internal combustion engine (ICE) generates noises when fuel is burned. Apart from that, noise is also generated from the exhaust, air intake, fans, and auxiliary equipment. Low-noise vehicles, i.e. electric vehicles (EVs), do not have an engine, instead, they have an electric motor and a controller. Therefore, electric and hybrid motor vehicles with electric drive have minimal engine noise and are often regarded as the silent vehicles. At least for speeds lower than 50 km/h, which is the general speed limit in most urban areas, engine is the dominant noise source. Therefore, electrification of vehicles will be an efficient way of reducing noise level in urban areas (Garcia, J.J. et al., 2013).

In the next few decades, it is predicted that there will be a growing fleet of electric and hybrid EVs. Among the heavy vehicles for urban use, i.e. busses and delivery trucks, hybrid EVs are gaining popularity (Garcia, J.J. et al., 2013). These changes can have a major influence on the traffic noise development. The findings from a past literature indicate that there is a potential for noise reduction by replacing the internal combustion engine vehicles (ICEVs) with EVs, however, it is uncertain as to how large the potential is (Iversen, L.M. et al., 2013). The adoption of EVs may be a transformative approach in reducing traffic noise and health risks due to excessive noise exposure. There is a need to verify the effectiveness of EVs in road traffic noise and health risk reduction.

2.4.2 Introduction to electric vehicles

Climate change issues, urban air pollution and the reliance on fossil fuel have become the driving forces for the EVs technology in the automotive industry. EV is a type of vehicle that has an electric motor and is powered by a large traction battery pack or fuel cell for propulsion instead of the ICE. The main benefit of EVs over other modes of transport is their environmentally friendly properties, by which they produce lower greenhouse gases and air pollutants emissions, thus contributing in climate change and air pollution mitigation.

EVs are initially designed and promoted for air pollution control, and are gaining growing interest in recent years. Currently, the volume of EVs is increasing on the European road networks. The adoption of EVs is actively supported by the EU, with several European governments promoting various subsidies and incentives to encourage EV ownership. Outside the EU, Norway, the leading country in EV usage has decided on a national goal to end fossil-fuelled car sales by 2025. Meanwhile, Japan aims to have all new passenger cars be electric by 2050. Looking at the EV market shares in 2018, Norway leads with 46%, while Japan is at 1% (International Energy Agency (IEA), 2019).

There are three main types of EVs, namely pure EVs, also known as battery electric vehicles (BEVs), plug-in hybrid electric vehicles (PHEVs), and hybrid electric vehicles (HEVs). These EVs differ from each other based on how the electricity for vehicle propulsion is generated and stored (Poullikkas, A., 2015; Sovacool, B.K. and Hirsh, R.F., 2009). Any form of EV contains a battery to store power and an electric motor for propulsion except for the hydrogen fuel-cell EVs. The major difference between the three main types of EVs is that for the pure EVs and PHEVs, the battery can be externally recharged, while the HEVs generate energy through the braking system to recharge the battery (Poullikkas, A., 2015).

Pure EVs are all-electric vehicles that are fully powered by batteries, with no secondary source of propulsion, meaning that the vehicles emit no emission from the exhaust and do not contain the typical liquid fuel components, such as the fuel pump, fuel line, or fuel tank. Pure EVs have large capacity battery as they rely solely on electricity. PHEVs are different from the HEVs because PHEVs have a larger grid-chargeable battery and a smaller ICE (Poullikkas, A., 2015; Sovacool, B.K. and Hirsh, R.F., 2009). The electric motor of PHEVs is more powerful than that of HEVs to support the all-electric drive at higher speeds.

HEVs are internal combustion engine vehicles (ICEVs) fitted with onboard battery and electric motor. The battery and electric motor are typically smaller than that of the pure EVs and

PHEVs. In terms of travelling range, HEVs offer higher distance compared to BEVs and PHEVs due to the existence of the ICE. HEVs are fuel-efficient because they employ electric-drive technologies to boost efficiency. The vehicle continues to charge itself and switches from the fuel source to electricity on slow speeds, thereby saving fuel (Alternative Fuels Data Center, 2020).

2.4.3 Impacts of electric vehicles in reducing sound level

To reduce the health effects of road traffic noise, a shift from the ICEVs to EVs may be a transformative approach. While EVs are gaining growing interest in recent years due to their environmentally friendly properties in reducing carbon emissions and energy consumption, their quietness is not recognised to mitigate the health risks due to road traffic noise.

EVs can almost entirely eliminate engine noise, hence, the noise from an EV is virtually equivalent to its rolling noise, i.e. the noise from the tires. Moreover, the use of electric motors in vehicles will substantially reduce noise in urban areas, particularly at low speeds.

A research in the United States indicates that even if all ICEVs are replaced with EVs, they will only reduce the average sound level by 1 dB during the day and 2.5 dB at night (Kaliski, K. et al., 2012). Another study (Lelong, J. and Michelet R, 2001) shows that replacing ICEV passenger cars with EVs give very little effects at peak hours and noon. This lack of reduction is due to the fact that during these hours, more heavy vehicles, i.e. busses and trucks occupy the streets.

In the Netherlands, predictions were performed where 90% of passenger cars and 80% of heavy trucks were substituted with EVs. The results showed that the overall noise reduction was within 3–4 dB and the annoyance effect reduction was more than 30% (Verheijen, E. and Jabben, J., 2010). The study was then extended to 95% of EVs passenger cars and a few heavy vehicles, and as a result, a significant noise level reduction was obtained on urban roads with speed below 30 km/h (Iversen, L.M. et al., 2013). In Spain, a fleet of EVs were introduced into the urban traffic flow and it was concluded that the noise levels using 100% EVs passenger cars would reduce by 2 dB and would improve the acoustic environment for 10% of the citizens (Campello-Vicente, H. et al., 2017).

In a recent study conducted in a city in Sweden (Ögren, M. et al., 2018), several noise reduction measures, including the application of EVs were investigated. The scenarios included speed limit reduction, traffic flow reduction, introduction of EVs, and introduction of low-noise tyres and low-noise road surfaces. For the introduction of EVs, the scenario assumed 25% of light vehicles being electric. The results showed that the introduction of EVs would only have a minor impact on noise exposure, unless they are combined with low-noise tyres or low-noise road surfaces.

A study comparing five hybrid busses with two conventional busses using diesel engine found a maximum of 12 dB reduction when the hybrid busses were driven in the all-electric mode (Biermann, J.W. and Ruschmeyer, S., 2012). Another study comparing one hybrid distribution truck, either in hybrid or electric mode, found that the electric mode benefit exceeded 8 dB at low speed and reduced at higher speeds. At 50 km/h, the difference was only 1 dB (Pallas, M.A. et al., 2012).

A recent study in eight Swedish municipalities (Borén, S. et al., 2016) compared three electric busses, three diesel busses, and three compressed natural gas (CNG) busses. The average values of the acceleration noise for the electric busses were 9.2 dB and 9.9 dB lower than diesel and CNG busses from 0 km/h, respectively. However, from 56 km/h, the values were 5.7 dB and 8.4 dB lower than diesel and CNG busses, respectively.

Overall, with the adoption of light and heavy EVs, past literatures estimated approximately 1–4 dB and 10 dB reductions, respectively. This strengthens the fact that the adoption of EVs contributes to the noise reduction and has a major impact on the noise reduction at lower speeds. However, at higher speeds, the impact reduces. Kaliski, K. et al. (2012) suggested that the level of impact will be a function of the proportion of non-electric heavy vehicles, average speed and proportion of time spent idling, accelerating, and at cruise. The non-electric heavy vehicles will dominate the overall traffic noise level, regardless of the proportion of EVs (Kaliski, K. et al., 2012). Therefore, it can be assumed that in urban areas, the noise level reduction will heavily depend on the proportion of heavy EVs and low speeds. However, further study is required to verify the hypotheses.

A reduction in the noise levels is expected to reduce the health effects of noise. To date, little attention has been given to the health effect reduction of electrically powered vehicles on national roads.

Chapter 3

Mapping of transportation noise-induced health risks as an alternative tool for risk communication with local residents

3.1 Introduction

Noise or unwanted sound is pervasive in day-to-day life and has always been a significant environmental problem for man. Documented problems associated with urban environment noise date back to ancient times. In ancient Rome, chariots were banned from the streets at night to prevent disrupted sleep and annoyance they caused to the residents (Goines, L. and Hagler, L., 2007). A few centuries later, noise mitigation strategies were carried out by some cities in Medieval Europe by banning carriages at night or covering the stone streets with straw in order to reduce noise and ensure peaceful sleep for the residents (Goines, L. and Hagler, L., 2007).

In modern society, population growth, urbanisation, and technological developments contribute to the severity of environmental noise, resulting in an increase in noise levels. In the past, environmental noise has traditionally been dismissed as an unavoidable fact of life because people have a certain predisposition to tolerate noise in their lives.

In 1910, Robert Koch, a Nobel prize-winning microbiologist once said that “One day man will have to fight noise as fiercely as cholera and pest” (Münzel, T. et al., 2014). His prediction came true. Environmental noise is now recognised as an environmental and public health issue that needs to be addressed in the modern society. This mainly owes to the development of studies in estimating population-level exposures to environmental noise that permit large-scale epidemiological studies. The findings indicate that environmental noise is responsible for a range of health effects. Be it in the United States (Hammer, M.S. et al., 2013), Europe (Belojević, G.A. et al., 2008; Lercher, P. et al., 2011; Selander, J. et al., 2009; Van Kempen, E. and Babisch, W., 2012), South America (Paiva, K.M. et al., 2019), and Asia (Ma, J. et al., 2018; Yoshida, T. et al., 1997), these studies have concluded a range of adverse health outcomes due to noise exposure. In fact, there is a burgeoning body of research that links noise to adverse health effects in addition to proactive legislation, primarily in the European Union (EU) (Murphy, E. and King, E. A., 2014).

In the European countries, it is reported that:

- About 40% of the population is exposed to road traffic noise levels exceeding 55 dB L_{day} and more than 30% of the population is exposed to levels exceeding 55 dB L_{night} (Berglund, B. et al., 1999)
- One in three individuals is annoyed during daytime and one in five has disturbed sleep at night because of traffic noise (WHO-EU, 2011)
- Long-term exposure to environmental noise is estimated to cause 12,000 premature deaths and contribute to 48,000 new cases of ischaemic heart disease (IHD) per year in the European territory (European Environment Agency, 2020)
- An estimation of 22 million people suffer chronic high annoyance (HA), 6.5 million people suffer chronic high sleep disturbance (HSD), and as a result of aircraft noise, 12,500 schoolchildren are estimated to suffer learning impairment in school (European Environment Agency, 2020)
- Based on the assessment threshold specified by the Environmental Noise Directive (END) of the European Union (EU), at least 100 million people in the EU are affected by road traffic noise, and in western Europe, at least 1.6 million healthy years of life are lost as a result of road traffic noise (WHO-EU, 2018)

It has also been proven that road traffic, being the most widespread source of environmental noise, is the noise source that generally raises more issues relating to annoyance and sleep disturbance.

Estimates of the number of population exposed to noise levels above the recommended guideline value in major cities in the EU are produced regularly since 2006 in accordance to the END. Other countries, e.g. South Korea, Hong Kong and the US also follow suit by producing noise maps and exposed population estimation. In Japan, however, the fields of environmental noise and public health are not as extensively studied as in the European countries. In regard to road traffic noise, although the Japanese road traffic noise prediction model, ASJ RTN-Model is available, city noise maps and population estimates are not thoroughly investigated. Nonetheless, there are a handful of studies that explored the relationship between noise and health risks and estimated the number of population that suffers from non-auditory adverse effects of road traffic noise in Japan.

A socio-acoustic survey that was carried out in 1997 among 3,600 adult Japanese women living in eight urban residential areas resulted in a crude prevalence rate of insomnia of 11.2% due to nighttime road traffic noise. The study suggested that road traffic noise raises the sound level in bedrooms in such zones, and consequently the prevalence rate of insomnia among the residents, and that noise-induced insomnia is an important public health problem, at least in highly urbanised areas (Kageyama, T. et al., 1997).

In the same year, another questionnaire-based study was performed in a small area near a main road in Tokyo to elucidate the relationship between road traffic noise and adverse effects of noise (Yoshida, T. et al., 1997). 366 women living on both sides of the road were analysed. The study suggested that noise may be related to the health status of inhabitants living in areas with heavy road traffic. The effects were depression, fatigue, and irritability above a threshold of 70 dB $L_{\mathrm{eq(24h)}}$. However, the analysis was unadjusted for age or social class.

An estimation of health risks due to road traffic noise in Japan carried out in 2012 approximated that 0.9 million people were highly sleep-disturbed and the number of estimated prevalence and mortality for cardiovascular diseases was 4,209 and 399, respectively (Matsui, T., 2012).

To date, most research has been focused on the direct cause-effect relationships between transportation noise and health outcomes. The strongest evidence base on causal-effect relationships between noise and health has recently been released by the World Health Organization Regional Office for Europe (WHO-EU) in the form of a guidance document, i.e. the *Environmental Noise Guidelines for the European region* (WHO-EU, 2018). For the adverse health effects of road traffic noise, the WHO-EU guidelines had given priority to the incidence of ischaemic heart disease (IHD) and hypertension, prevalence of highly annoyed population, reading skills and oral comprehension in children, and sleep disturbance (WHO-EU, 2018).

In 2002, the European Commission issued the Environmental Noise Directive (2002/49/EC), also known as the END, as a policy instrument. The Directive aims to avoid, prevent, or reduce on a prioritised basis, the harmful effects, including annoyance due to exposure to environmental noise, by the following actions (European Union, 2002):

- the determination of exposure to environmental noise, through noise mapping, by the common methods of assessment;
- ensuring that information on environmental noise and its effects is made available to the public; and
- adoption of action plans based upon the noise mapping results.

In accordance with Article 6.2 of the END, the European Commission undertook the development of the Common NOise aSSessment methOds (CNOSSOS-EU) that represents a harmonised and coherent approach to assess noise levels from the main noise sources (road, rail, aircraft traffic, and industrial noise) (Kephalopoulos, S. et al., 2014).

At present, noise maps are used for the pre-evaluation of acoustic plans to mitigate the effects of noise on residents (Di, H. et al., 2018; Gozalo, G.R. et al., 2016; Murphy, E. and King, E.A., 2011; Paschalidou, A.K. et al., 2019; Vogiatzis, K. and Remy, N., 2014; Wang, H. et al., 2018). It is an extremely important part of the process of quantifying and visualising noise pollution levels (de Kluijver, H. and Stoter, J., 2003). The END requires the noise maps to be not only made available to the public but also disseminated with the freedom of access to information. Information presented to the general public is required to be 'clear, comprehensible, and accessible'.

Risk communication is important to educate the general public about the dangers associated with environmental noise. However, the sound level estimation is not useful in risk communication with the public because a noise map typically exhibits acoustic intensity instead of the potential health effects. Therefore, explaining the health risks to the residents using noise maps for effective risk communication is challenging since the residents are not familiar with the relationships between sound levels and health risks. Subsequently, local residents will find noise maps incomprehensible as they are unable to recognise the quantitative health impacts from an ordinary noise map. They can only compare the map around their houses with the standard noise level. Therefore, it is not beneficial to communicate the risks of noise to the community.

To heighten the awareness of the general public on the adverse health effects of noise exposure, information about the health risks needs to be appropriately structured to be understandable to the public (WHO-EU, 2013, 2020). The risk communication tool should also be comprehensible and can be interpreted in a meaningful way by the public.

An alternative tool that displays the health risks transparently will be valuable for enhancing risk communication, i.e. graphical material, particularly, maps. Maps are seen as a potentially powerful means of conveying spatial information visually (Dransch, D. et al., 2010; Stieb et al., 2019). Displaying health risks by means of mapping may be effective to present information and communicate findings as they are often more memorable because of the colour and shape (Health and Environment Linkages Policy Series (HELI), 2020).

The aim of this study is to develop an alternative tool for enhancing risk communication by visualising the health risks due to environmental noise in health risk maps translated from ordinary noise maps. The risk maps will exhibit numerical information and provide visual representations of specific adverse health risks that might affect an entire community, thereby serving a variety of risk communication purposes.

In this study, I developed road traffic noise maps and health risk maps of Sapporo City (area 1,121 km^2) in Japan. Firstly, I developed the road traffic noise maps of Sapporo City, Japan. The road traffic noise maps are the basis for generating health risk maps associated with excessive noise pollution, which was the main purpose of our study. I applied the exposure–response functions (ERFs) established by the WHO-EU and the national health statistics and surveys in Japan to convert the sound levels into health risks, and, finally, obtained the geospatial distribution of the health risks: health risk maps. High annoyance (HA), high sleep disturbance (HSD), and ischaemic heart disease (IHD) were the observable health risks. Furthermore, I estimated the number of population exposed to road traffic noise and the negative health outcomes to reveal the overall health effects in the city.

Contrary to the sound level shown in ordinary noise maps, the health risk maps visualised the percentages of people annoyed (%HA) and highly sleep-disturbed (%HSD) and revealed the prevalence and mortality rates for IHD. The health risk maps will help the public realise the significance of the noise exposure impacts on general health. Moreover, the health risk maps can be utilised to demonstrate the potential risk reduction in future noise mitigation strategies. Here, I proposes the development of risk maps instead of noise maps for a convenient understanding of the adverse noise impacts on a community's health.

3.2 Materials and methods

3.2.1 Study area

The study area was Sapporo City, which is located in the northernmost island of Hokkaido in Japan. It is the fifth largest city in Japan, with a total area of 1,121 km^2 and the total population stood at 1.969 million in 2019 (City of Sapporo, 2019). It is characterised by heterogeneous environments of mixed residential and commercial areas with well-planned transportation networks. Figure 3.1 shows the road network of this study. Those expressways and trunk roads, of which the traffic volume information was available, were considered. The total length of the road is 712.5 km, with an average traffic volume of 0.5–19.8 million vehicles per year.

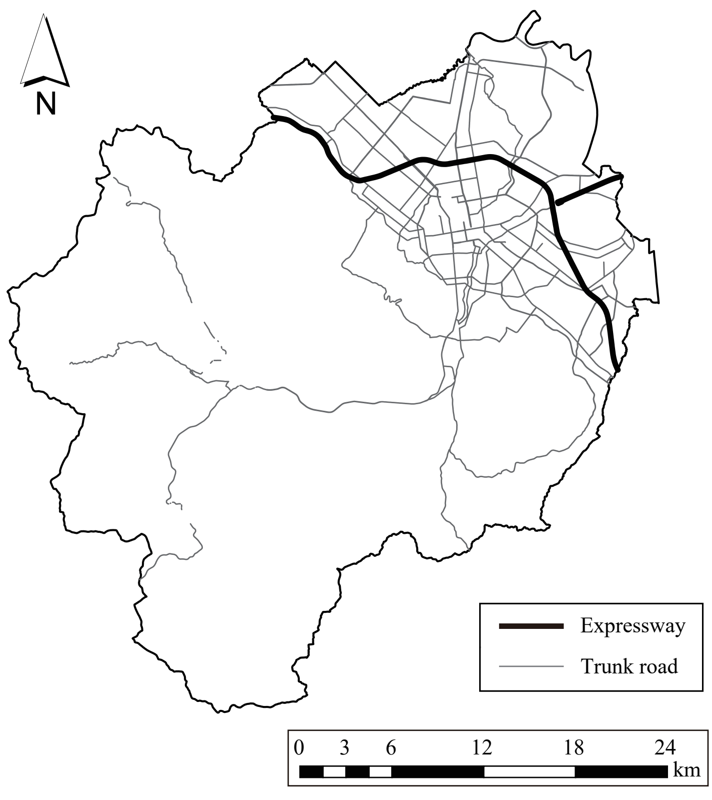

Figure 3.1: Road network in Sapporo City that is considered in the present study. The grey lines are trunk roads and solid black lines are expressways.

3.2.2 Sound level calculation at façades

The first step in calculating sound level is determining the noise prediction method. Most nations use their national calculation methods, if available. However, each national calculation method was originally developed to meet the specific conditions or legislation as applied in each nation, long before the END was issued (Murphy, E. and King, E. A., 2014). While most methods have been revised in recent years, they are based on methods that were developed prior to the advent of the personal computer and data-logging sound level meters (Murphy, E. and King, E. A., 2014). Furthermore, no calculation method was ever developed with the intention of producing noise maps on a national scale.

In Japan, the Acoustical Society of Japan (ASJ) developed the ASJ RTN-Model for the prediction and evaluation of road traffic noise in 1993, and the model is updated every five years, with the latest update in 2018 (Sakamoto, S., 2020). However, the ASJ RTN-Model does not consider the reflected sound on buildings and noise barrier walls. This will cause inaccurate estimations in sound level. Moreover, the frequency characteristics of the sound source are not considered. Therefore, the effect of the sound-absorbing ground will be overestimated when the ASJ model is used (Fukazawa, 2015).

In the Europe, the END requires its Member States to use the common methodological framework for strategic noise mapping, i.e. the Common NOise aSSessments methOdS in the EU (CNOSSOS-EU) noise propagation method from 31 December 2018 onwards (European Union, 2015). A field study on sound power level of Japanese road vehicles compared the spectral characteristics between the average measurement data with three road traffic noise prediction models, i.e. the ASJ RTN-Model 2013, Fukushima model, and CNOSSOS-EU.

The results of the study found that the CNOSSOS-EU model agreed well with the measurement results for all vehicle categories. Although the CNOSSOS-EU model is based on European vehicles, the results showed that the shape of the frequency characteristics of noise emission of the Japanese vehicles was almost the same as the European vehicles (Yonemura et al., 2016). Futhermore, ASJ RTN-Model are meant for estimating noise exposure and developing noise mitigation measures while CNOSSOS-EU is developed for strategic environmental noise mapping. Considering the abovementioned situations, in this study, the CNOSSOS-EU framework was employed as the noise model for estimating the sound power level and sound propagation.

Acquiring accurate calculation levels at the receiver points requires several data that must be pre-processed in the geographical information systems (GIS) software, which allows users to manage and analyse the geographic data (ESRI, 2020). The datasets are traffic network (roads geometry), traffic data (speed, etc.), buildings (shapes and heights), and ground elevation data. The GIS software used in this study was the ArcGIS 10.4.2, ESRI Japan, Tokyo.

The building geodata, including the heights were obtained from Zmap–AREAII, Zenrin, Tokyo. The elevation data were obtained from the Geospatial Information Authority of Japan, Ministry of Land, Infrastructure, Transport and Tourism, which is the national organisation that conducts basic survey and mapping. It also has open geospatial data that can be accessed and used. The ground elevation data made available by the organisation were the 5 m and 10 m grid topographic base maps. Since the 5 m grid topographic base map has a limited range, the 10 m grid topographic base map was used in this study (Geospatial Information Authority of Japan (GSI), Ministry of Land, Infrastructure, Transport and Tourism, 2014).

The geometrical information (height and width) of the selected roads was obtained from the 2015 Digital Road Map Database, issued by the Japan Digital Road Map Association (DRM) (Japan Digital Road Map Association, 2015). The digital road network data include the elevation of the roads from the ground level. The road traffic datasets and characteristics of expressways and trunk roads were obtained from the 2010 Road Traffic Census, issued by the Road Bureau, Ministry of Land, Infrastructure, Transport and Tourism of Japan (Road Bureau, Ministry of Land, Infrastructure, Transport and Tourism, 2010).

The CNOSSOS-EU has five classes of vehicles (Kephalopoulos, S. et al., 2012). However, only two categories (light and heavy vehicles) are available in the Road Traffic Census in Japan. I classified the two categories into Category 1 and 3, respectively. The information of the road characteristics, i.e. the lane and central reservation width and information relating to traffic flows, i.e. one way/two way lane, the number of vehicles per hour during the day, evening, and night, the percentages of heavy vehicles during the day, evening, and night, and the average daily traffic, were also extracted from the Road Traffic Census.

The effects of the correction coefficients were also taken into consideration. The road surface was set as the reference road surface according to the CNOSSOS-EU and the temperature of the road surface was set to 10 °C based on the information on the annual average temperature of Sapporo City (Japan Meteorological Agency, 2010). Based on the typical vehicle speed limits in Japan, I assumed the average speeds of 60 km/h and 100 km/h for the vehicles on the trunk roads and expressways, respectively.

Noise barrier walls were also set at the expressways. The distance of the walls to the road edges was calculated based on the road lane width information in the Road Traffic Census. The wall height was set to 2.0 m above the road surface. Taking the general noise barrier walls height in Japan's expressways, and assuming that the walls are fully reflective concrete, the reflection loss (loss of energy occurring at each wall) at the inner and outer walls was set to 1 dB.

The road properties and information relating to traffic volume from the Road Traffic Census, and other required parameters mentioned above were linked to the digital road network of DRM in ArcGIS 10.4.2. These data, and the building and ground elevation data were exported to SoundPLAN 8.1, Ono Sokki, Japan.

In SoundPLAN, several parameters needed to be set before beginning the calculation. The assessment methods were L_{den} and L_{night}, in accordance with the END, and also the noise indicators for estimating the health risks. The emission time slices were set to three; day (7 am to 7 pm), evening (7 pm to 11 pm), and night (11 pm to 7 am), representing the A-weighted long-term average noise level measured over a year. The type of ground was set to 0 for hard surfaces (most normal asphalt, concrete).

The calculation settings of sound level at the façades are shown in Table 3.1:

Table 3.1: Calculation settings of sound level at façades in SoundPLAN 8.1, Ono Sokki, Japan.

Parameters	Value
Receiver height	4 m
Reflection order	3
Maximum search radius	1000 m
Maximum reflection distance (receiver)	200 m
Maximum reflection distance (source)	50 m
Allowed tolerance	0.5 dB

The assessment of population exposure to noise is based on the receiver levels at 4 m above the ground in front of the building façades of residential buildings (European Union, 2002). One receiver is assigned in the centre of the façades. The parameter reflection order determines how many reflections shall be calculated. To guarantee sufficient noise map accuracy for strategic noise mapping process, one reflection is necessary (CEDR Project Group Road Noise 2, 2013). The increment in the number of reflections will result in a more accurate sound level, however, the calculation time will also increase.

Therefore, it is necessary to set the optimum number of reflections in which the accuracy of the results and the time required for the calculation are balanced. These were already investigated in the past studies (Fukazawa, 2015; Fukazawa and Toshihito, 2014). The number of reflections of three and more resulted in slight changes in the sound levels when high-rise buildings of 30 m height were lined up at both sides of the road. Based on the results, three was considered the appropriate number of reflections (reflection order) in this study.

The maximum search radius sets how far a source can be from the receiver and still contribute to the noise level at the receiver (SoundPLAN GmbH / SoundPLAN International LLC, 2019). To set this parameter, several thousand receivers were assigned at 500 m distance from the road. The maximum search radiuses were set to 2.0 km (set as the reference), 1.5 km, 1.0 km, and 800 m. The sound levels of the majority of receivers were less than 45 dB L_{night}. Hence, the maximum search distance should be more than 500 m. The maximum search distance of 1000 m was selected because there were small differences in the sound level compared to the reference.

For the maximum reflection distance to the receiver and source, the SoundPLAN calculates the reflections for the reflecting surfaces that are either closer to the receiver than the first parameter entered or closer to the source than the second parameter (SoundPLAN GmbH / SoundPLAN International LLC, 2019). There were significant differences when I applied the shorter reflection distance instead of the recommended values of SoundPLAN. In general, the default values produced sensible results and were set for the parameters.

The allowed tolerance is designed to increase the calculation. It is set to 0.5 dB, which is the value for noise mapping calculation in accordance to the END, and holds for the total result, where only the most important sources are calculated in detail (SoundPLAN GmbH / SoundPLAN International LLC, 2019).

The calculation of sound level at the façades was done with distributed computing, where the process of calculation was distributed to 11 PCs in the network.

3.2.3 Developing façade noise maps

Figure 3.2 displays a schematic diagram showing the process of making a noise map. The annual average metrics of the day-evening-night equivalent sound level (L_{den}) and nighttime equivalent sound level (L_{night}) at each building façade in the whole city area were calculated to estimate the noise-exposed population volume. The calculated values of the receivers in SoundPLAN 8.1 in L_{den} and L_{night} were exported to ArcGIS 10.4.2. The assessment points of the receivers must be at the most exposed façade in accordance with the END to estimate the

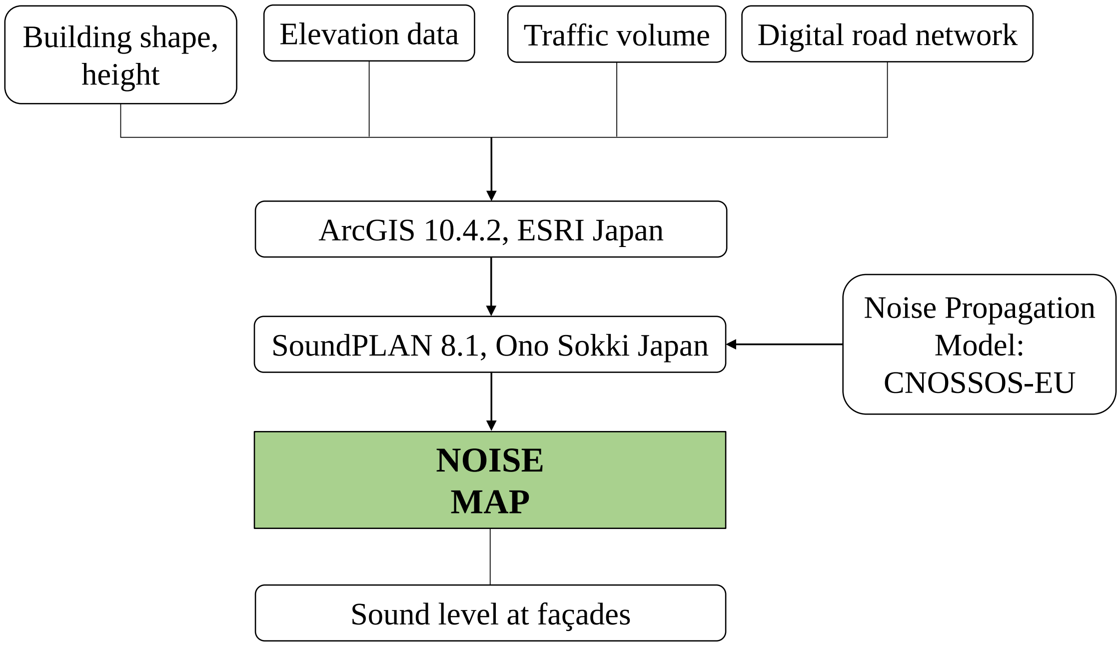

Figure 3.2: Schematic diagram of developing a noise map.

noise-exposed population. Therefore, regardless of the location of rooms in the buildings, the most exposed façade of each identified building signifies the predicted sound levels and health risks of the residents. However, the actual estimation might be lower at other façades.

The maximum L_{den} and L_{night} of each building were then illustrated in the façade noise maps in 5 dB noise bands in accordance with Annex IV paragraph seven of the END. These maps should at least cover the range required by Annex VI of the END, i.e. from < 55 dB to 75 dB for L_{den} and < 50 dB to 70 dB for L_{night} (CEDR Project Group Road Noise 2, 2013).

3.2.4 Developing health risk maps

I developed the health risk maps by converting the sound level to health risks. The study flow is shown in Figure 3.3.

The critical health outcomes with the exposure-response functions (ERFs) as reported in the WHO-EU guidelines WHO-EU (2018) were selected as the noise-induced health risks in our study, namely, high annoyance (HA), high sleep disturbance (HSD), and ischaemic heart disease (IHD). The percentage of highly annoyed (%HA) and highly sleep-disturbed (%HSD)

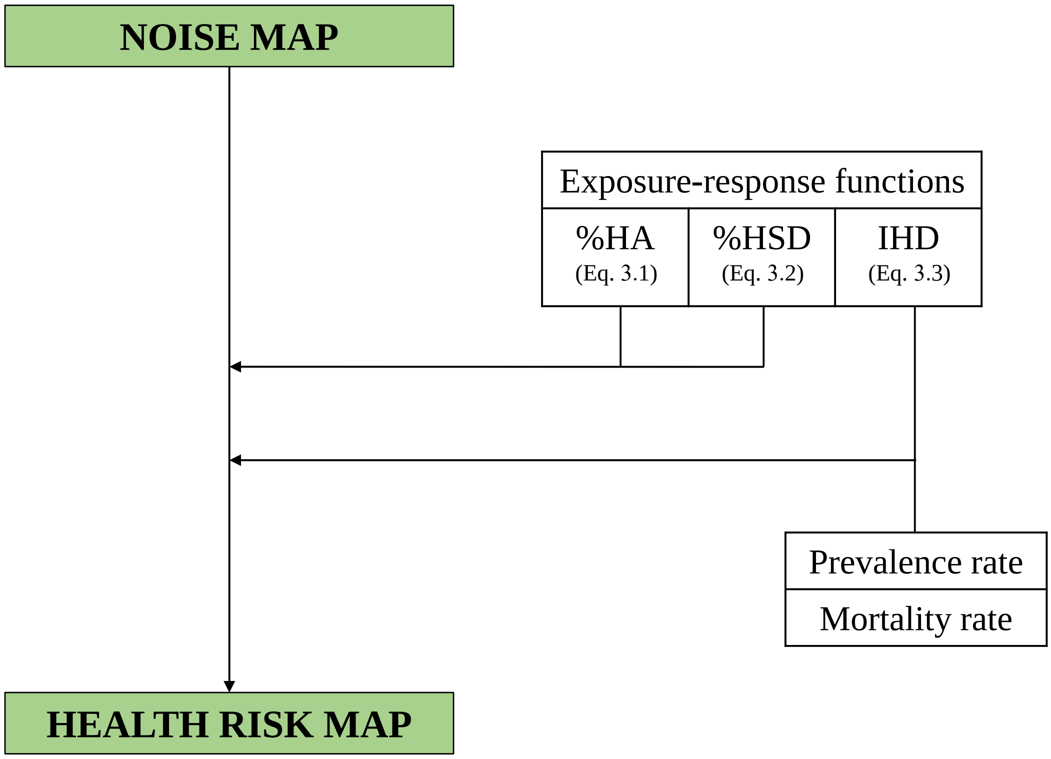

Figure 3.3: Schematic diagram to generate health risk maps from noise maps by applying the exposure-response functions on health impacts due to noise.

was estimated as follows (Equation (3.1) and Equation (3.2), respectively):

$$\%\mathrm{HA} = 78.9270 - 3.1162 \times L_{\mathrm{den}} + 0.0342 \times L_{\mathrm{den}}^2 \tag{3.1}$$

$$\%\mathrm{HSD} = 19.4312 - 0.9336 \times L_{\mathrm{night}} + 0.0126 \times L_{\mathrm{night}}^2 \tag{3.2}$$

where, the range is 40 dB $\leq L_{\mathrm{den}} \leq$ 80 dB for %HA and 40 dB $\leq L_{\mathrm{night}} \leq$ 65 dB for %HSD.

The relative risk (RR) of IHD due to road traffic noise was estimated by using the following equation (Equation (3.3)) across a noise range of 53 dB $\leq L_{\mathrm{den}} \leq$ 80 dB.

$$\mathrm{RR} = 1.08^{(L_{\mathrm{den}}-53)/10} \tag{3.3}$$

In respect of IHD, the prevalence and mortality rates were calculated based on the exposure-response relationships and the population exposed to road traffic noise. The calculations were made by using the population attributable risk with the information of prevalence and mortality rates in Japan using the national statistics data.

I calculated the prevalence rate by multiplying the average prevalence rate in Japan, i.e. 590

per 100,000 people with RR in L_{den}. The prevalence rate was calculated using the number of IHD patients from the 2011 Patient Survey issued by the Ministry of Health, Labour and Welfare (Ministry of Health, Labour and Welfare, 2010). Likewise, the mortality rate was calculated by multiplying the yearly mortality rate, i.e. 113 per 100,000 people with RR in L_{den}. The death statistics were extracted from the 2010 Vital Statistics issued by the Ministry of Health, Labour and Welfare (Ministry of Health, Labour and Welfare, 2010).

The percentages of affected residents, prevalence rates, and mortality rates were integrated into the noise maps to expose the potential morbidity and mortality due to road traffic noise.

3.2.5 Estimation of exposed population and health risks

In estimating the population exposed to noise, I used the number of residents data in each sections in Sapporo City. Figure 3.4 shows the schematic diagram of estimating the exposed population and health risks.

The population of each building was estimated on the assumption that the population in a block area is proportional to the floor area in the block area population. I obtained the population of block areas from the 2010 National Population Census issued by the Statistics Bureau of Japan, Ministry of Internal Affairs and Communications, whereas the data for floor areas of each building were obtained from the geodata (Zmap–AREAII, Zenrin, Tokyo). The number of residents living in specific buildings, P_{bld} was estimated by each section as follows:

$$P_{\text{bld}} = P_{\text{sect}} \times \frac{S_{\text{bld}}}{\sum S_{\text{bld}}} \tag{3.4}$$

where, P_{bld} is the number of residents per building, P_{sect} is the total population in the section, S_{bld} is the building floor area (m^2), and $\sum S_{\text{bld}}$ is the total building floor area in the section (m^2).

The population defined by sound levels at each noise band was multiplied with the ERFs to estimate the number of population exposed to the health risks. In a similar manner with making the health risk maps, the prevalence and mortality rates of IHD in Japan (Ministry of Health, Labour and Welfare, 2010, 2011) were also used in the estimation.

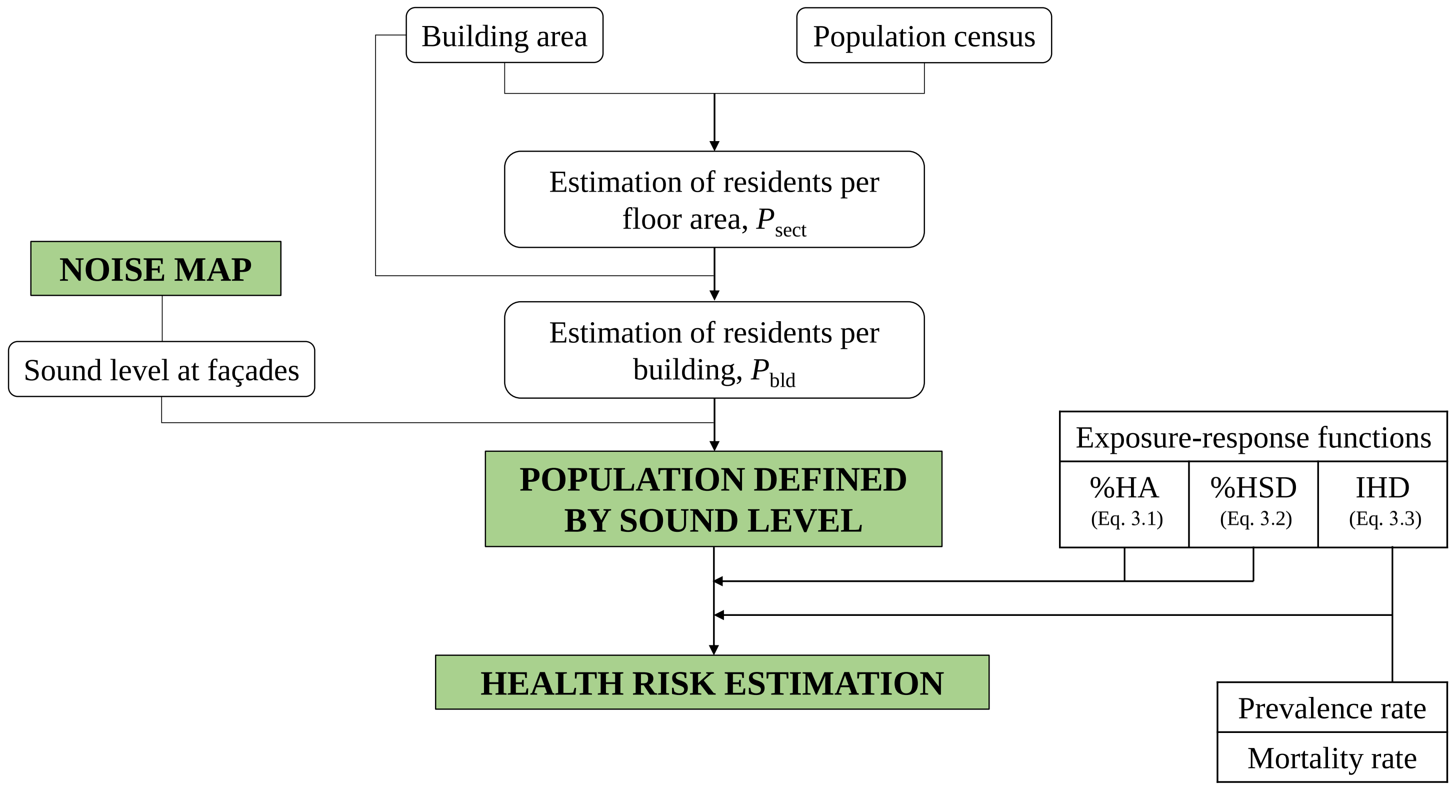

Figure 3.4: Schematic diagram of estimating population and health risks.

3.3 Results and discussion

3.3.1 Façade noise maps

Figure 3.5 shows the scatterplot of the predicted and measured sound levels in the study area. The measured levels were obtained from the field measurements in 2010 conducted by the National Institute for Environmental Studies, Japan (National Institute for Environmental Studies, Japan, 2010). Measurements for the measured sound levels and the Road Traffic Census for the predicted sound levels were carried out on different days. A difference of less than 5 dB between the predicted and measured values was observed.

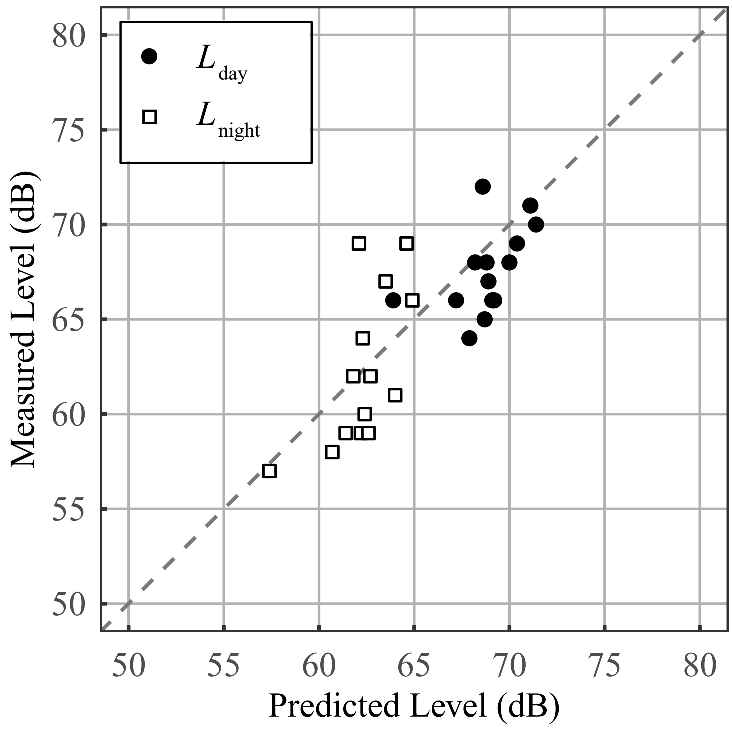

Figure 3.5: Correlation between predicted and measured sound levels (Measured and predicted sound levels survey was carried out on different days). A difference of less than 5 dB between the predicted and measured values was observed.

From the figure, it is evident that while some variations in the results exist due to the differences in the traffic volume during the sound level measurement and the Road Traffic Census, the predicted values show good agreement with the measured values. Therefore, it can be deduced that the prediction results have high reliability.

WHO-EU strongly recommends reducing the road traffic noise below 53 dB for L_{den} and below 45 dB for L_{night}, as 10% and 3% of residents complained of HA and HSD, respectively

(WHO-EU, 2018). The L_{den} of 53 dB is also a threshold of the increase of IHD risk, as shown in Equation 3.3. The façade noise maps of Sapporo City constructed for L_{den} and L_{night} indicated noise levels exceeding the WHO-EU guidelines, which could increase the risk of IHD and HSD to the buildings' populations.

Figure 3.6 and Figure 3.7 show samples of façade noise maps in Sapporo City for L_{den} and L_{night}, respectively. The enlarged views of a section in the façade noise maps in the figures show a highly noise-exposed area adjacent to the interconnected expressways and trunk roads, for which the total average traffic flow was approximately 11.5 million vehicles per year.

In Sapporo City's urban areas, the sound levels are higher than 55 dB L_{den} and 45 dB L_{night}, particularly at buildings that are in close proximity to the expressways and trunk roads. However, in the western part of the city, which has mountainous terrain and a much smaller number of residential buildings, the sound levels are estimated to be lesser than 55 dB L_{den} and 45 dB L_{night}.

From the enlarged view within the façade noise maps, the buildings were estimated to be exposed to high sound levels exceeding 75 dB L_{den} and 60 dB L_{night}. In addition to the buildings being located along the immediate perimeter of the roads, other nearby residential areas were also exposed to road traffic noise levels, i.e. 55 dB L_{den} and 45 dB L_{night} or higher.

Lower noise levels, i.e. lesser than 55 dB L_{den} and 45 dB L_{night} were predicted at the areas with a distance of approximately 100–300 m from the roads. Sound levels that exceed the WHO-EU guidelines of 53 dB L_{den} and 45 dB L_{night} are predominantly evident at buildings near the roads, which is often the case for urban areas. This underlines the seriousness of noise exposure levels to the population as they will increase the health risks. In this area, the number of population that are exposed to sound levels > 55 dB L_{den} and 40 dB L_{night} are 5,978 (37.0%) and 13,275 (82.3%) out of 16,135 population.

The noise map developed here is the first significant-sized map in Japan. A few previous studies only roughly estimated the sound levels in small areas (Bhaskar, A. et al., 2007; Fujimoto, K. and Anai, K., 2002; Oshino, Y. et al., 2006). Despite the availability of traffic and geometrical information in Japan to estimate them, information about the sound level and health risks due to noise exposure was not disseminated.

Noise mapping is a common technique for the graphical representation of sound level distribution, as shown in Figure 3.6 and Figure 3.7. It gives light to the local residents on the state of the acoustic environment, i.e. the degree of noise exposure in their residential areas. While

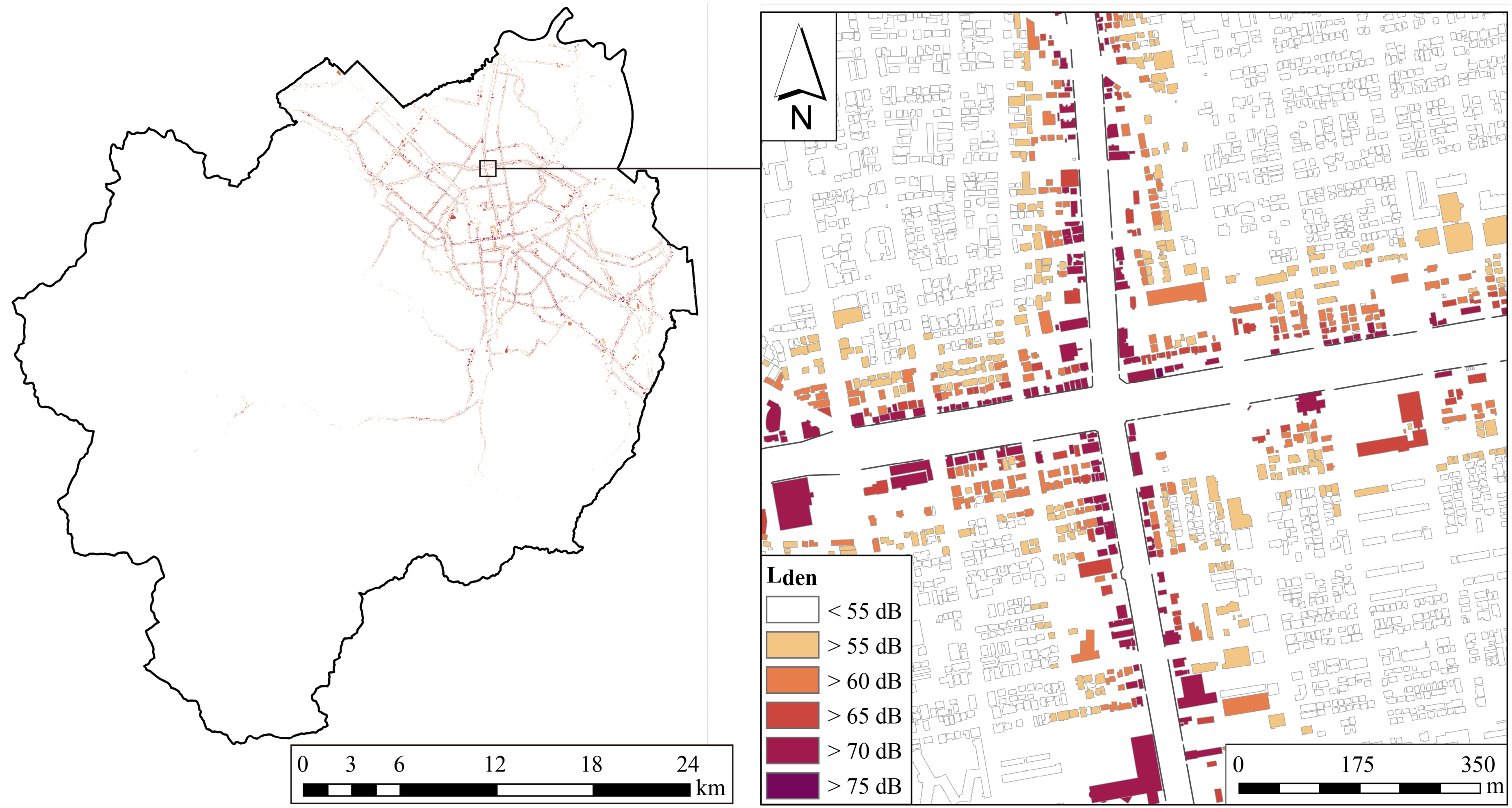

Figure 3.6: An example of the façade noise map of the road traffic noise in Sapporo City in L_{den}. Buildings within the vicinity of the roads show sound levels exceeding the WHO-EU recommended value of 53 dB L_{den}.

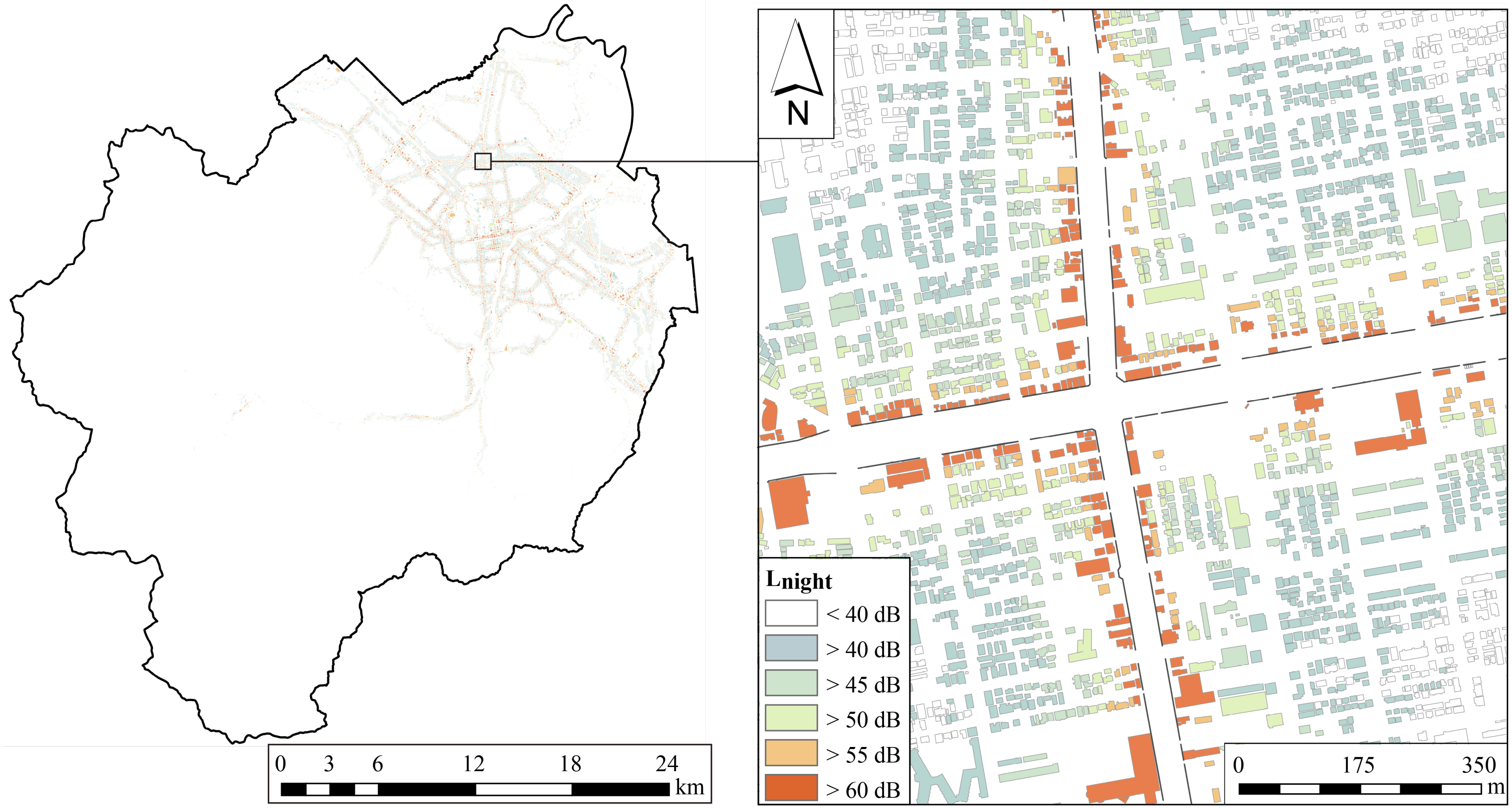

Figure 3.7: An example of the façade noise map of the road traffic noise in Sapporo City in L_{night}. Buildings within the vicinity of the roads show sound levels exceeding the WHO-EU recommended value of 45 dB L_{night}.

the policy makers can estimate the impacts of noise exposure on health using the façade noise maps, the local residents will have difficulty in understanding the health risks from an ordinary noise map that exhibits acoustic intensity instead of the health effects.

Therefore, noise maps are inefficient to prove the danger of environmental noise and may not be convenient to demonstrate the health effects of noise to the local residents. Health risks must be communicated effectively if they are to be understood and acted upon in an appropriate manner. Comprehensive estimation of the health risks is significantly important in communicating the health risks to the public. To communicate the adverse effects of noise to the general public, an alternative tool for risk communication, namely health risk map is necessary.

3.3.2 Health risk maps

Figure 3.8 presents the health risk maps for high annoyance in %HA and high sleep disturbance in %HSD of the same area shown in Figure 3.6 and Figure 3.7. These metrics indicate the probabilities that certain percentages of the population, when exposed to certain levels of road traffic noise, would be highly annoyed or have high levels of sleep disturbance, respectively, at a given spot (Kim, M. et al., 2012). Figure 3.9 exhibits the prevalence and mortality rates per 100,000 people in the risk maps for IHD of the exact same area as in Figure 3.6 to Figure 3.8.

In these health risk maps, the buildings are shown in different shades of colours indicating different risk levels to the residents of the buildings. The colour shades from light to dark correspond to the low- to high-risk levels. The geospatial distribution of the health risks is visualised using the health risk maps, while the mean acoustic intensities are shown in Figure 3.6 and Figure 3.7. For instance, HA of approximately 15.0% instead of 60 dB L_{den} and HSD of approximately 6.0% instead of 55 dB L_{night} were estimated. Further, in terms of the risk of IHD, a prevalence rate of 20 per 100,000 people instead of 57.3 dB L_{den} and 10 deaths per 100,000 people instead of 64 dB L_{den} were approximated.

The lowest value shown in Figure 3.8, i.e. 10%HA is the benchmark level and is reached at 53.3 dB L_{den}. At low noise exposure levels, less than 10% of population was predicted to express high levels of annoyance. However, as the noise level increased, there was a rapid rise in the predicted occurrence of high annoyance level. At approximately 68 dB L_{den}, over a quarter of the population would indicate high level of annoyance with noise. This is illustrated in the figure, where the estimated %HA is more than 25% for people living close to the roads. Also in Figure 3.8, 3% of the population is characterised as highly sleep-disturbed at 45 dB

L_{night}. Higher %HSD of more than 9% (approximately at 60 dB L_{night}) was observed for people living close to the roads.

In Figure 3.9, the highest prevalence and mortality rates were observed for people residing alongside the roads, where the estimated number of patients and deaths was more than 80 and 15 people, respectively, out of 100,000. Even a small shift due to increasing noise exposure may yield a substantial increase in the prevalence and mortality of IHD, as visualised by the shape of the association, which indicates that the risk of IHD increases from above 53 dB L_{den} to 80 dB L_{den} (Figure 2.5).

In this area, the estimated number of population that are highly annoyed and highly sleep-disturbed are 1,187 and 575, respectively. Meanwhile, the estimated prevalence and mortality for IHD are 2.9 and 0.6, respectively.

An enlarged view of the mortality risk map is shown in Figure 3.10 to distinguish the mortality risks of each building's residents. Evidently, the residents of building C with a mortality rate of more than 5 per 100,000 people are less susceptible to death from noise-induced IHD as compared to those living in buildings A and B because residents in buildings A and B have a mortality rate of more than 20 and 10 per 100,000 people, respectively. Therefore, individual health risks can be recognised owing to the spatially risk estimation. Interpreting the health risk maps can be achieved without difficulty and is useful to communicate risks to the local residents.

The mortality risk of noise exposure at the exposed area is relatively high when compared with other causes of death. Table 3.2 and Table 3.3 lists the number of mortalities and mortality rates according to the causes of death in Sapporo City and Japan, respectively, as reported in the 2010 Demographics Statistics issued by the Ministry of Health, Labour and Welfare (Statistics Japan, Ministry of Health, Labour and Welfare, 2010). These statistics can be compared with the health risk maps (see Figure 3.9 and Figure 3.10) to obtain an estimate of the severity of the noise-induced health risks. For instance, in the highly exposed areas, where the mortality rate of IHD is more than 20 per 100,000 people, the health risks due to noise exposure is equivalent to that due to suicide and kidney failure and is significantly higher than that due to leukaemia, traffic accident, tuberculosis, and influenza in terms of the mortality rate. This comparison can educate local residents regarding the gravity of the noise-induced health risks in their houses.

As a substitute to exhibit the adverse health effects of noise, health risk mapping is a dynamic tool that enables easy visualisation of the noise-induced health risks, as exemplified in

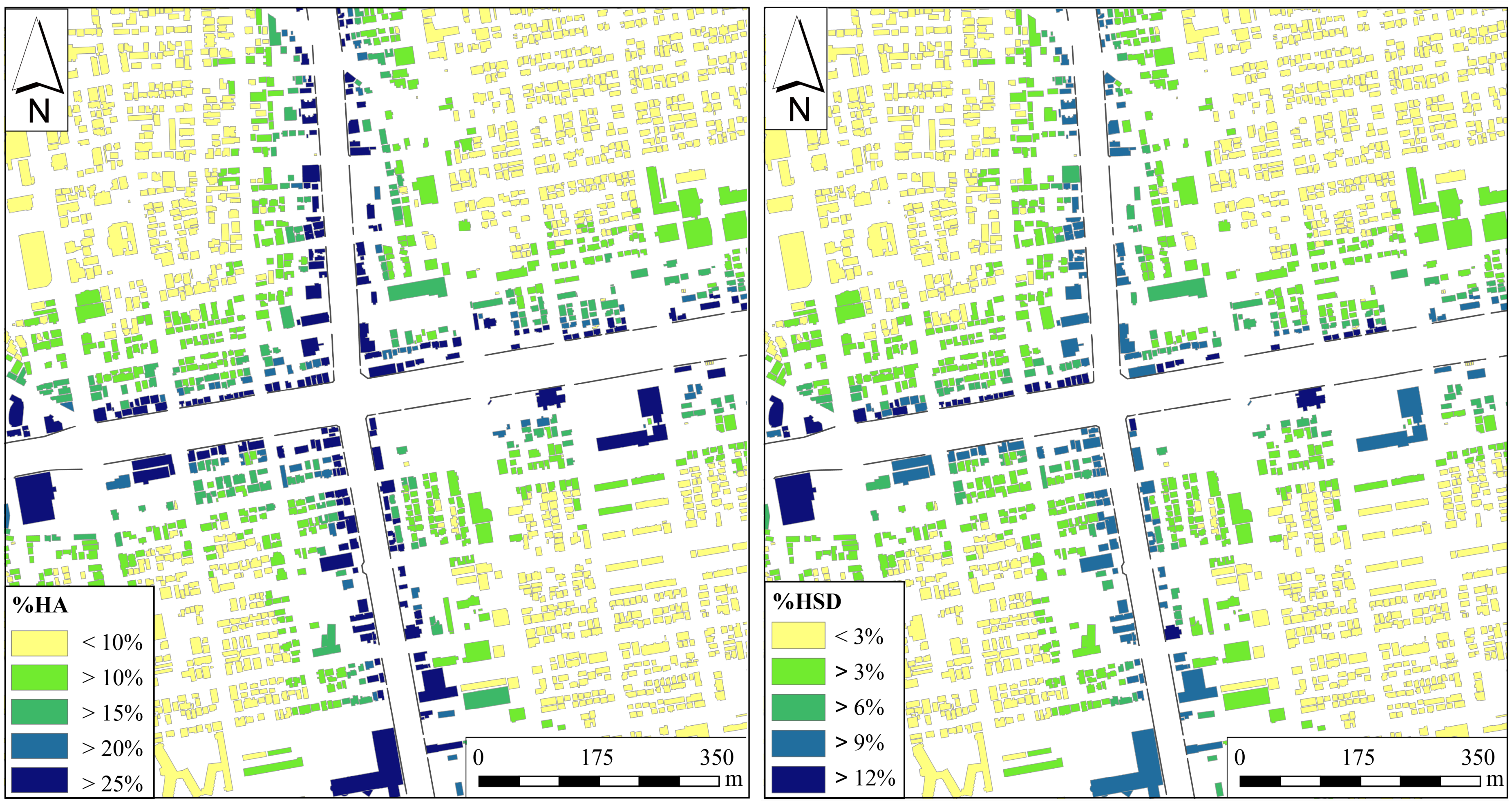

Figure 3.8: An example of the health risk maps for the percentage of people highly annoyed (%HA) (left panel), and highly sleep-disturbed (%HSD) (right panel) at a highly noise-exposed area in Sapporo City.

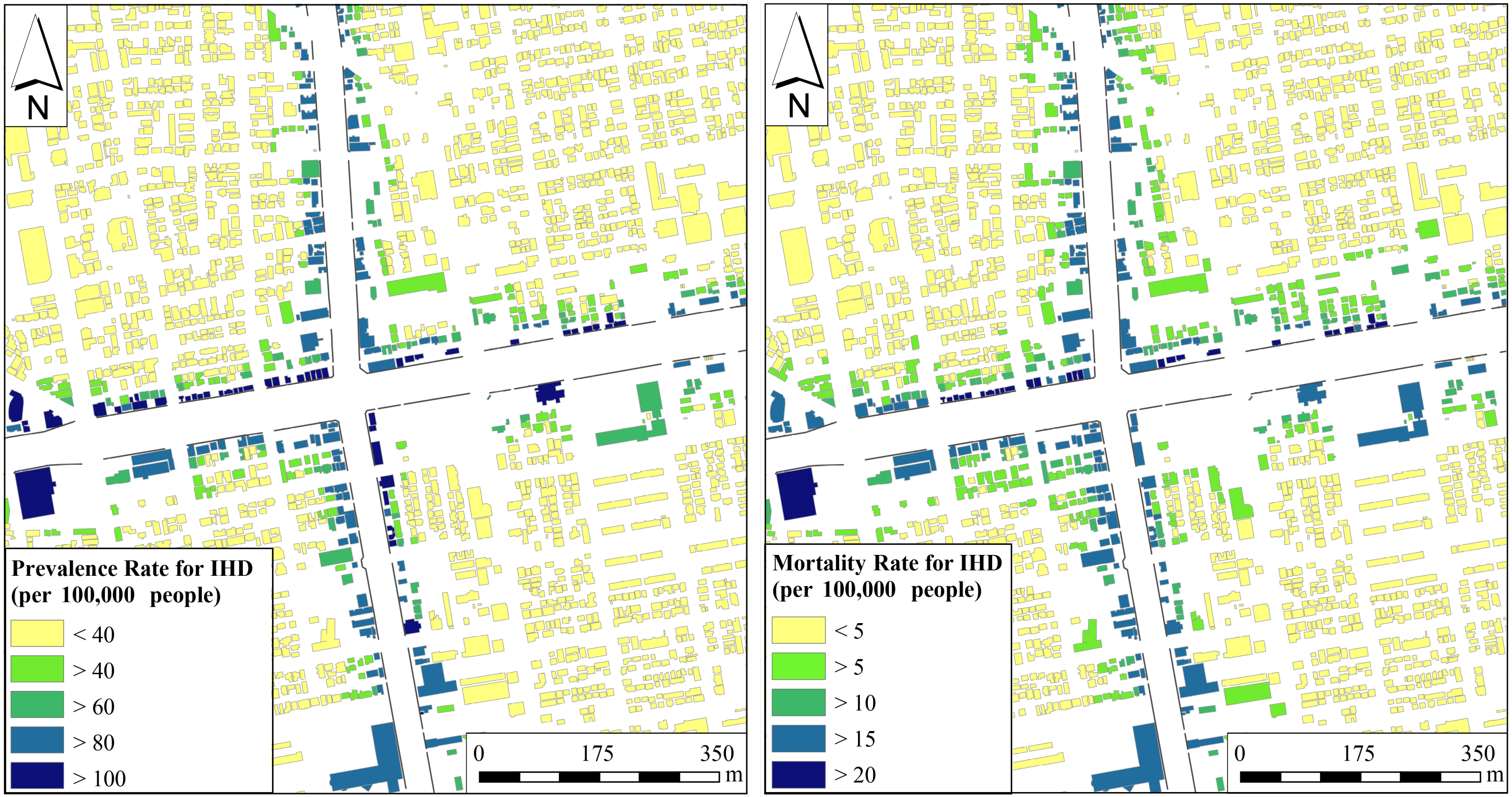

Figure 3.9: An example of the health risk map for the prevalence rate (left panel) and mortality rate (right panel) of the IHD due to noise exposure at a highly noise-exposed area in Sapporo City.

Figure 3.10: Large-scale mortality risk map displaying the mortality rates for different buildings at a highly noise-exposed area in Sapporo City

Table 3.2: Number of deaths and mortality rates due to various causes in Sapporo City for a comparison with the noise-induced health risks.

Cause of death	Number of deaths	Mortality rate*
Cancers	5,256	275.7
Ischemic heart diseases	1,763	92.5
Stroke	1,372	72.0
Suicide	431	22.6
Kidney failure	329	17.3
Leukaemia	103	5.4
Traffic accident	67	3.5
Tuberculosis	26	1.4
Influenza	8	0.4
Total	15,482	812

*per 100,000 person-year

Table 3.3: Number of deaths and mortality rates due to various causes in Japan for a comparison with the noise-induced health risks.

Cause of death	Number of deaths	Mortality rate*
Cancers	353,499	276.0
Ischemic heart diseases	144,075	112.5
Stroke	123,461	96.4
Suicide	29,554	23.1
Kidney failure	23,725	18.5
Leukaemia	8,078	6.3
Traffic accident	7,222	5.6
Tuberculosis	2,129	1.7
Influenza	161	0.1
Total	1,197,012	935

*per 100,000 person-year

Figure 3.8 to Figure 3.10. Ordinary noise maps only illustrate the sound level, thus making it difficult for laymen to interpret the adverse health effects of road traffic noise. On the contrary, visualisation of the health risks through health risk mapping conveys a clear message of the confronting risks as the maps can be easily interpreted by the residents. The maps also highlight the high-risk zones, where noise control or noise mitigation strategies must be executed without delay. This method will greatly facilitate the evaluation of noise-induced health impacts and act as an early detection system to impede health-related problems.

The greatest advantage of the health risk maps is that they can avoid potential misinterpretation of risks from the noise maps by directly showing the health risks. The goal of risk communication is to assist stakeholders in taking risk-based decisions based on a balanced judgment, which results from factual evidences regarding the risks (OECD, 2002; Paek, H.J. and Hove, T., 2020; Sjöberg, L., 2002). Communicating the risks based on ordinary noise maps may lead to misperception of the risks and subsequent inaccurate judgments in those situations by the public.

Another aspect that affects the risk communication is the trustworthiness and credibility of the communicated information (Dransch, D. et al., 2010; Gamhewage, G., 2020; WHO, 2017). Significant public trust can be gained using health risk mapping owing to the transparent and convenient visualisation of the health risk distribution. The health risk maps graphically quantify the risks of each building and can help the public to comprehend the gravity of the health risks due to noise exposure.

The visualisation of health risks through health risk mapping clearly reveals the posing risks to the community. For environmental risk factors, the acceptable 'lifetime risk' is 1 per 100,000 people (Ministry of the Environment, Japan, 1997). Therefore, local residents can realise the risks they are facing by comparing the health risk map for mortality rate due to noise-induced IHD and the acceptable lifetime risk. The health risk maps can be effective in public health risk communication as they can contribute to the knowledge sharing between local communities and help in raising public awareness.

Although we employed the exposure–response relationships established by the WHO-EU, uncertainties remain in the risk estimations. Vulnerable groups of people, i.e. the elderly, new born babies, and those with illnesses, may be more at risk from noise exposure than healthy adults. In addition, the health risk maps heavily depend on several factors, such as the availability and quality of the information about population, exposure, and health, which may contribute to a misclassification of the severity of risks and can, subsequently, affect the reliability of the health risks maps. Therefore, information regarding the reliability should be provided in the risk communication.

Moreover, since the methodology depends on data availability, not all of the noise sources can be considered in estimating the health risks. Therefore, the noise sources should be specified when communicating the risks with the local residents. Graphical representation of risk communication (maps in particular) is acknowledged for its usefulness in several fields, i.e. public health, environmental pollution, and meteorological risk assessment (Dransch, D. et al., 2010; Stieb et al., 2019). Disease mapping and mapping for risk communication regarding various natural hazards, such as wildfires (Mozumder, P. et al., 2009), floods (Hagemeier-Klose, M. and Wagner, K., 2009; Macchione, F. et al., 2019) and volcanic eruptions (Haynes, K. et al., n.d.; Nave, R. et al., 2010), are common practices, and they are evolving with time. Regarding environmental exposures and health risks, there are a few studies that provide health risk information to the public (i.e. asthma and cancer risk in relation to air pollutants (Hammond, D. et al., 2011; Severtson, D.J. and Myers, J.D., 2013)).

So far, only a handful of studies have mapped health risks due to road traffic noise (Hanigan, I.C. et al., 2019; Park, T. et al., 2018). However, those maps are not intended for use in risk communication, where numerically and spatially precise estimations of the health risks are required.

3.3.3 Estimated population exposed to noise, the number of patients, and deaths

In this study, I carried out numerical calculations on the health risks in Sapporo City. It should be noted that this is the first study to reveal health risks due to road traffic noise in a Japanese city.

Table 3.4 and Table 3.5 present the estimated population exposure and health risks attributed to road traffic noise in Sapporo City for L_{den} and L_{night}, respectively, in 5 dB range.

Table 3.4: Estimated population exposure to health risks in Sapporo City due to road traffic noise for L_{den}

	L_{den} (dB)						
	< 55	55–60	60–65	65–70	70–75	> 75	Total
Exposed population	1,430,450	165,981	106,271	102,782	106,750	1,306	1,913,540
Population percentage	74.8	8.7	5.6	5.4	5.6	0.1	100
Highly annoyed							
Percentage affected	0.00	12.82	17.76	24.41	32.77	42.84	—
Number of affected	0	21,277	18,872	25,086	34,978	559	100,773
Ischemic heart disease							
Relative risk	1.00	1.04	1.08	1.12	1.16	1.21	—
Prevalence	0	35	48	72	102	2	257
Mortality	0	7	9	14	19	0	49
Prevalence rate*	0.0	20.8	44.8	69.7	95.6	122.5	—
Mortality rate**	0.0	4.0	8.5	13.3	18.2	23.3	—

*per 100,000 person, **per 100,000 person-year

Table 3.5: Estimated population exposure to health risks in Sapporo City due to road traffic noise for L_{night}

	L_{night} (dB)						
	< 40	40–45	45–50	50–55	55–60	> 60	Total
Exposed population	1,003,873	344,626	196,169	125,726	82,637	160,510	1,913,540
Population percentage	52.5	18.0	10.3	6.6	4.3	8.4	100
Highly-sleep disturbed							
Percentage affected	0.00	2.51	3.51	5.15	7.41	10.30	—
Number of affected	0	8,657	6,893	6,470	6,122	16,532	44,674

The data show that approximately 25% of the total population was exposed daily to high road traffic noise levels, i.e. $L_{\text{den}} > 55$ dB, whereas approximately 48% was exposed to L_{night}

$> 40\,\mathrm{dB}$. The exceedingly high number of exposed people resulted in approximately 100,000 people highly annoyed, 45,000 people highly sleep-disturbed, 260 patients, and 50 deaths due to the IHD.

High annoyance and sleep disturbance are the leading health effects due to road traffic noise in Sapporo City. Although the relationship between annoyance and noise-induced diseases such as cardiovascular diseases remains unclear, road traffic noise would threaten the well-being of a significant proportion of people in the city. The number of patients with sleep disorders due to noise would be very large and important for public health since high sleep disturbance is also regarded as a mild sleep disorder (WHO-EU, 2011).

In addition, as evident from Table 3.4, a significant number of people in the population are potentially at risk of contracting and dying from IHD due to high exposure levels of environmental noise. This indicates that significantly high health risks due to road traffic noise are evident, hence, residents' protection from the noise exposure should not be neglected. With HSD as a risk factor associated in manifesting cardiovascular dysfunctionality (Basner, M. and McGuire, S., 2018; WHO-EU, 2011, 2018) including IHD, this may add up to the number of IHD morbidity and mortality.

The numbers of patients and deaths due to IHD of approximately 260 and 50 eventually results in the total prevalence and mortality rates of 13.4 and 2.6 per 100,000 people. The estimated mortality rates of IHD due to road traffic noise were 1.4 per 100,000 in Japan (Tagusari, J. and Matsui, T., 2020), and 1.9 per 100,000 in European countries (European Environment Agency, 2020). Given that the estimation of the present study is mainly for the urban areas, the result would be consistent with the existing studies. The overall mortality risk of IHD due to road traffic noise in Sapporo City is equivalent to the risk of traffic accidents and higher than tuberculosis and influenza (see Table 3.2). It should also be noted that it is higher than the typical acceptable lifetime risk of environmental risk factors of 1 per 100,000 people (Ministry of the Environment, Japan, 1997). The results provide evidence that the total risk of road traffic noise in Sapporo City cannot be left unaddressed and need to be reduced with raising the priority.

Moreover, as also shown in the risk maps, the affected people are concentrated in the areas highly exposed to road traffic noise. That is, 2/3 of deaths due to noise-induced IHD are concentrated on 10% of the people, who are exposed to L_{den} of higher than 65 dB. The estimated mortality risk at the areas is higher than 10 per 100,000, which is higher than the major causes of deaths (see Table 3.2). Risk communication with local communities is desirable to prevent and/or mitigate the health risks, in particular for the people highly exposed to environmental noise.

3.4 Conclusion

Noise-induced health effects are, without a doubt, a threat to the community. While epidemiological and experimental studies have reported the associations between traffic noise and various adverse health effects, limited attention is given to the mitigation of the health effects. The effects are preventable and by reducing the environmental risk factors, the occurrence of noise-related diseases can be lessened. Similarly, the public should be made aware of the adverse impacts of noise on their health for a better understanding of the risk level they are exposed to. Public awareness on the adverse health effects of noise exposure should also be raised as a key to curb noise pollution, and this is where risk communication is of utmost importance.

In this study, I proposed health risk mapping, which is a potential alternative communication approach to a noise map. This study fills an important gap in the literature through the proposal of health risk mapping as a way of communicating risk with the general public. Health risk mapping uses scientific evidence to quantify the noise impacts on health and to identify and assess health risks for an effective public health risk communication. Because health risks are graphically displayed in health risk maps, it is easier for the public to understand health risk maps than ordinary noise maps.

The proposal to develop health risk maps is not limited to the road traffic noise and health risks mentioned in this study. These maps are also applicable to other types of transportation noise, i.e. railway and aircraft noises, and health risks. If the health effects are provided with the dose-response relationships, the health risks maps can be replicated, i.e. for hypertension and stroke. However, the limitations of reliable data and their analysis and the uncertainty in estimation of the health risks need to be addressed before expanding this method. Hereafter, risk communication using such an easily comprehensible tool can be employed by the local communities to raise awareness in general people on the adverse health effects of noise exposure and to design policies for reducing the noise exposure. The effectiveness of the health risk maps as an alternative risk communication tool should be further investigated, for example, by measuring the risk perception of the pre and post risk communication of the users with the health risk maps.

In addition, this is the first study showing the health risks due to road traffic noise in a Japanese city using recent evidences provided by the WHO-EU. The prevalence and mortality rates of IHD in Sapporo City were estimated to be 13.4 and 2.6 per 100,000 people, respectively. Considering the remarkably large number of people exposed to high noise levels, the burden of disease is substantial despite the relatively small prevalence and mortality rates. Therefore, I infer that noise-induced IHD morbidity and mortality are significantly worrying, and adoption of appropriate prevention measures is of utmost importance. Health risk estimations all over Japan are desirable to evaluate the risks of environmental noise and enforce mitigation policies.

Chapter 4

Mitigation of health risks due to road traffic noise by the transition to electric vehicles

4.1 Introduction

The results of my first study on the health risk mapping of road traffic noise showed that a significant proportion of population in Sapporo City was exposed to the adverse health effects of road traffic noise. Efforts to mitigate health risks are necessary to avoid the consequences.

However, environmental noise is not easily reduced over the short term and requires long-term noise mitigation measures to reduce noise levels on a broader scale (Murphy, E. and King, E. A., 2014) despite noise mitigating policies such as the Environmental Noise Directive (END, 2002/49/EC) by the European Commission. More than half of the global population is now living in urban areas and it is projected to increase to 68% by 2050 (United Nations, Department of Economic and Social Affairs, Population Division, 2020). The urbanisation phenomenon may cause an increasing number of people being exposed to excessive environmental noise and noise-induced health effects. Therefore, in order to protect the public health, there is a need to address this issue before it becomes critical.

There are various noise mitigation approaches that are able to reduce noise and concomitantly reduce its health effects. The details of these mitigation approaches are described in Chapter 2. The most effective noise control measure is by reducing noise at its source, which includes legislation, low-noise road surface, traffic management, low-noise tyres, and low-noise vehicles. Recent social changes in achieving a low carbon society could be a transformative means in reducing road traffic noise: the transition from conventional vehicles to electrical vehicles (EVs).

EVs have emerged in the past few decades as a relevant and promising approach to significantly improve the acoustic environment. EVs are being heralded as an alternative to the conventional vehicles, or internal combustion engine vehicles (ICEVs), and are often designated as the silent vehicles due to their minimal engine noise. EVs almost entirely eliminate noise emission from an engine and could substantially reduce noise in urban areas. These changes can have a major influence on the acoustic environment.

EVs were initially designed and promoted for air pollution control in the cities. The number of EVs on major roads, especially in the Europe, is on the rise and a growing fleet of EVs is expected in the coming decades. Europe is entering the initial adoption phase of electric mobility, which is strongly encouraged by the government stimulus, i.e. promoting various subsidies and incentives to encourage EV ownership. German, for example, has set the target that there will be one million EVs on the roads by 2020; France has a goal of two million EVs on the road by 2020 (Amsterdam Round Table Foundation and McKinsey & Company, 2014); Norway, the leading country in EV usage has targeted that all new cars sold by 2025 should be zero-emission (EVs); and Japan aims to have all new passenger cars be EVs by 2050.

Electric vehicles become an increasingly common sight in the years ahead. Battery EVs account for the majority of these, but plug-in hybrids also play a role over the next ten years before fading as pure electrics continue to get cheaper. On a global scale, only 1 in 250 cars on the road is electric. Meaning, electric vehicles account for 2.2% of the global vehicle market share. The latest figures show that there are 8.5 million EVs in the world and in 2030, the global EV fleet is predicted to be 116 million. Despite the rapid growth, there will be 1.4 billion passenger vehicles on the road in 2030 and EVs account for 8% of these (a rise from current status of 0.7%). This rises to 31% by 2040 as the fleet slowly changes over (*Electric Vehicle Outlook 2020*, 2020).

Currently, the majority of the EV fleet comprises of light vehicles and two-wheelers. However, it is expected that electrification will spread into the heavier vehicle segments, e.g. electric busses and medium and heavy-duty trucks. In 2018, the European electric bus market increased 48% compared to 2017. In 2019, the number of electric bus registration in western Europe tripled. As of 2019, there were 2,300 urban electric buses on European roads. That number is predicted to grow at a rate of 68%, such that by 2025, the number of electric bus fleet will be approximately 17,000 (Geotab Inc., 2020).

With cumulative sales of less than 1 million through 2019, the global market of heavy-duty electric vehicles is still at a very early stage. At the end of 2019, 60 countries on six continents had seen deployment of at least one electric heavy-duty vehicle. Over 98% of the global cumulative sales through 2019 occurred in China. Cumulative heavy-duty electric vehicle sales of more than 2,000 have not been achieved by other market. Outside of China, only the United States and India had over 1,000 cumulative sales of at least 1,000. Over 70% (44 of 60) of the countries with at least one heavy-duty electric vehicle sale had less than 100 cumulative sales (Cui, H. et al., 2020).

According to a report, at the end of 2017 there were 3 million city buses in operation worldwide. Of these, 385,000 belong to the category of electric bus. The incidence on the global fleet is therefore 13% (Bloomberg New Energy Finance, 2018*b*). The total number of electric buses in service is forecast to more than triple to about 1.2 million in 2025, equal to about 50% of the worldwide city bus fleet (Bloomberg New Energy Finance, 2018*a*).

As of today, in Europe, the latest number of electric busses is approximately 4,000 (battery electric buses, plugin hybrids, trolleybus and fuel cell buses). Globally, there are currently 400,000 electric buses in operation, with China dominating the total figure at about 99% (*Electric Vehicle Outlook 2020*, 2020). China has the largest electric bus fleet in operation due to the prioritisation of public transit electrification with subsidies and national regulations since a decade ago.

The growing interest on EVs is mainly due to their environmentally friendly properties in reducing carbon and energy emissions. This recent social change to sustainable electric mobility is one of the initiatives to achieve a low carbon society that will transform the acoustic environment. The sound level reduction of EVs was investigated in some studies; however, the impact of the transition on health risk reduction remains unclear.

In past literature surveys, several studies were conducted on sound level reduction due to the shift to EVs, in which 1–4 dB reduction was estimated (Campello-Vicente, H. et al., 2017; Iversen, L.M. et al., 2013; Kaliski, K. et al., 2012; Lelong, J. and Michelet R, 2001; Ögren, M. et al., 2018; Verheijen, E. and Jabben, J., 2010) in a combination of mixed light EVs and heavy EVs in the vehicle fleet. A handful of experimental studies that investigated primarily on heavy EVs showed more than 10 dB reduction in sound level (Biermann, J.W. and Ruschmeyer, S., 2012; Borén, S. et al., 2016; Pallas, M.A. et al., 2012). Another experimental study based on one EV (Kaliski, K. et al., 2012) indicated that the level of impact for sound level reduction will be a function of proportion of non-electric heavy vehicles, average speed, and proportion of time spent idling, accelerating, and at cruise. However, the reduction in health risks was not investigated.

The present study aimed to elucidate the impact of reducing health risks due to road traffic noise in a future noise-exposure setting by the transition to EVs from ICEVs and reveal the effective traffic conditions. The relationship between the health effect reduction and traffic condition factors, i.e. the percentage of heavy vehicles and traffic speeds according to the CNOSSOS-EU and the exposure-response functions (ERFs) was investigated. Given that the EVs have no internal combustion engine, the propulsion noise was assumed negligible.

To verify the calculation results, two areas in Sapporo City with different traffic conditions were selected. The sound level of road traffic noise in the real traffic flow was calculated by assuming two different situations, namely, traffic flows with ICEVs that represented the current situation and the prospective one with EVs substituting all ICEVs. The health risk reduction was analysed by assuming all vehicles (light and heavy vehicles alike) are replaced with EVs. The exposure-response relationships established in the guidelines of the WHO-EU (WHO-EU, 2018) and the national health statistic and surveys were applied to estimate the exposed population and the noise-induced health risks, i.e. high annoyance (HA), high sleep disturbance (HSD), and ischaemic heart disease (IHD) in Sapporo City, Japan. The reduction of health risks due to the transition to EV was estimated to elucidate the contribution of the mitigation approach.

4.2 Materials and methods

4.2.1 Calculations of the relationship between health risks and traffic conditions

In this study, the harmonised methodological framework for strategic noise mapping in the European Union (EU), i.e. the Common Noise Assessment Method in the EU (CNOSSOS-EU) framework was employed as the noise model for estimating the sound power level and sound propagation. The standard noise prediction methods, including CNOSSOS-EU, assessed the noise emission from the ICEV. The ICEV noise emission in the CNOSSOS-EU model is represented by two main noise sources, namely rolling noise and propulsion noise. The rolling noise increased more than the propulsion noise with increasing speed.

Firstly, the property of EVs as the noise source to estimate the sound level reduction was set. Given that EVs have no internal combustion engine, which is the main noise source of the propulsion noise, the propulsion noise of EVs was assumed negligible in the calculations. Although not included in the CNOSSOS-EU vehicle categories, EVs can be defined in the open vehicle category, which is introduced for future needs. A past study which is in the initial stage of development proposed correction coefficients for the propulsion noise component for light vehicles in all-electric mode in the CNOSSOS-EU method (Pallas, M.A. et al., 2016). However, the results were from experimental data measured on a sample of vehicles, which included pure EVs and hybrid electric vehicles (HEVs) in all-electric mode. HEVs have both the electric motor and conventional internal combustion engine. Pure EVs, or battery electric vehicles, on the other hand, are a subcategory of EVs that exclusively use rechargeable battery packs to power an electric motor. EVs work without internal combustion engines, and as a result, they have no source of propulsion.

The present study considered replacing ICEVs with EVs, hence, the sound power level for propulsion noise component, $L_{WP,i}$ in the CNOSSOS-EU method was assumed to be zero. For calculations that were based on CNOSSOS-EU, the sound power level of light and heavy ICEVs was calculated using Equations 2.10 to 2.15 (see page 43 and 44). This calculation is important to show the dominance of the rolling and propulsion noise, depending on speed. The estimated difference of the total sound power level for the light and heavy ICEVs and EVs was also calculated at 60 km/h and 100 km/h. The 60 km/h and 100 km/h speeds are the typical speed limits in Japan on the trunk roads and expressways, respectively. The purpose of this calculation was to examine the decrease of sound level when EVs were assumed.

The sound power level at traffic speed of 20–100 km/h and heavy vehicle percentage of 0–100% was calculated by the summation of the rolling and propulsion noise in Situations ICEV and EV. The difference of the noise exposure level is assumed to be the same as that of the power level. High annoyance (HA), high sleep disturbance (HSD) and ischaemic heart disease (IHD) were selected as the health effects due to road traffic noise exposure according to the guidelines by the WHO-EU (WHO-EU, 2018) and the exposure-response functions of these health risks, Equations 3.1 to 3.3 (see page 63), are employed. The relationship between the reduction of health risks and traffic conditions was examined based on the calculation results.

4.2.2 Sound level calculation

Two urban areas in Sapporo City, namely, Area A and Area B (see Figure 4.1) were selected to validate the results of the relationship between health risk reduction and traffic conditions. These two areas have different traffic conditions, i.e. high/low percentage of heavy vehicles and low/high speed. In Area A, there are trunk roads with a speed limit of 60 km/h, daily average traffic volume of 15,000–25,000 vehicles, and heavy vehicles percentage of 30–50%. In Area B, there is an expressway with a speed limit of 100 km/h and trunk roads with a speed limit of 60 km/h, the daily average traffic volume is 20,000–90,000 vehicles, and heavy vehicles percentage is 10–30%. Both have an area of 1 km^2. The study was then expanded to the whole agglomeration of Sapporo City to reveal the total mitigation effects in the city.

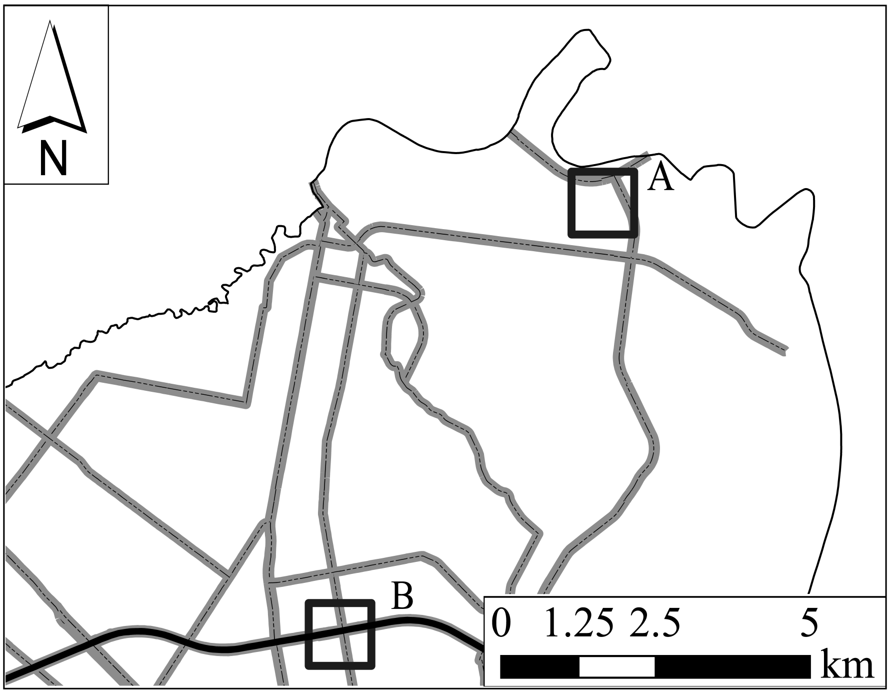

Figure 4.1: Location of the selected areas at northeast of Sapporo City. The solid grey lines are major roads and solid black line is an expressway considered in the present study.

The sound level calculations and health risk estimations were carried out in the two selected areas for two situations: Situation ICEV, where all vehicles are ICEVs, and Situation EV, where all vehicles are EVs. Conditions other than the sound power level were the same in both situations. The sound power level at traffic speeds of 20–100 km/h and percentages of the heavy vehicle of 0–100% were calculated by the summation of rolling and propulsion noise in both situations.

The ground elevation data used was the 10 m grid topographic base map issued by the Geospatial Information Authority of Japan (Geospatial Information Authority of Japan (GSI), Ministry of Land, Infrastructure, Transport and Tourism, 2014). The road traffic volumes of expressways and trunk roads were obtained from the 2010 Road Traffic Census, issued by the Road Bureau, Ministry of Land, Infrastructure, Transport and Tourism of Japan (Road Bureau, Ministry of Land, Infrastructure, Transport and Tourism, 2010). The geometrical information of the selected roads was obtained from the 2015 Digital Road Map Database, issued by the Japan Digital Road Map Association (Japan Digital Road Map Association, 2015). The building geodata were obtained from Zmap–AREAII (Zenrin, Tokyo). All calculations were done in ArcGIS 10.4.2 (ESRI Japan, Tokyo) and SoundPLAN 8.1 (Ono-Sokki, Tokyo). These data and the parameters set in SoundPLAN 8.1 are identical with the first study, which have been described in detail in Chapter 3.

The sound level of the most noisy façade of each building was calculated using the maximum façade sound level of each building, which is recommended to be used to estimate the noise-exposed population by the Environmental Noise Directive (END) (European Union, 2002). Moreover, the dose-response relationships between the façade sound level and health risk are shown in the guidelines (WHO-EU, 2018). This method is also similar with the method used to calculate the sound levels at façades in the first study, and has been described in detail in Chapter 3.

4.2.3 Health risk estimations after the transition to electric vehicles

To estimate the number of population exposed to road traffic noise in each building, we used the information of population in the small areas from the 2010 National Census issued by the Statistics Bureau of Japan, Ministry of Internal Affairs and Communications (Statistics Japan, Ministry of Internal Affairs and Communications, 2010) and the data of floor areas of the buildings in the study area from the geodata (Zmap–AREAII, Zenrin, Tokyo).

We selected three health outcomes from the guidelines (WHO-EU, 2018), namely high annoyance (HA), high sleep disturbance (HSD), and ischaemic heart disease (IHD) since they have high/moderate evidences. In respect of IHD, the prevalence and mortality rate were calculated based on the dose-response relationships and the population exposed to the road traffic noise. The calculation was made using the population attributable risk with the information of the prevalence and mortality rates in Japan at 590 and 113 per 100,000 people, respectively, obtained from the 2011 Patient Survey (Ministry of Health, Labour and Welfare, 2011) and the 2010 Vital Statistics (Ministry of Health, Labour and Welfare, 2010) issued by the Ministry of Health, Labour and Welfare. This method is identical with the method used to estimate the health risks in the first study, and has been described in detail in Chapter 3.

4.3 Results and discussion

4.3.1 The relationship between health risk reductions and traffic conditions

Examples of sound power level of light and heavy ICEVs are shown in Figure 4.2 and Figure 4.3, respectively, in which the speed dependence of rolling noise and propulsion noise contributions was calculated.

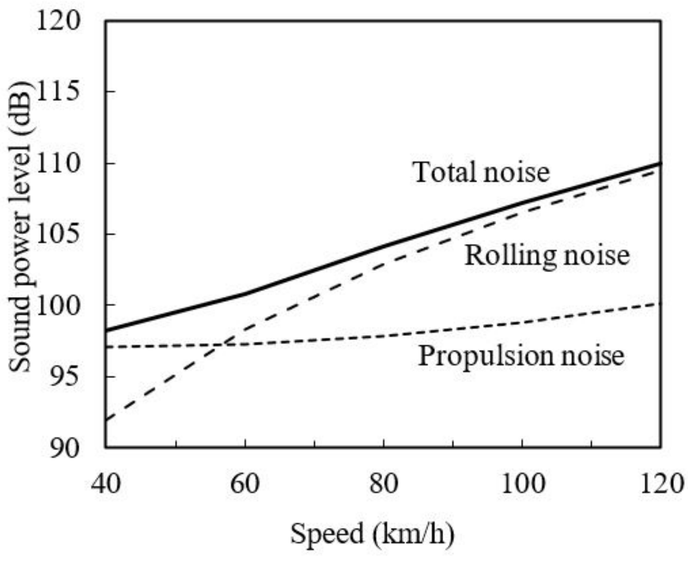

Figure 4.2: An example of sound power level vs vehicle speed. Rolling noise, propulsion noise, and the total noise emission of a light vehicle were calculated with CNOSSOS-EU at an air temperature of 10°C.

In Figure 4.2, the propulsion noise of a light ICEV is dominant at lower speeds, but at higher speeds the rolling noise is dominant. Thus, a light ICEV of similar weight and shape and with the same type of tyres will emit the same noise at higher speeds. In Figure 4.3, the propulsion noise of a heavy vehicle is also dominant at the lower speeds. In urban driving conditions,

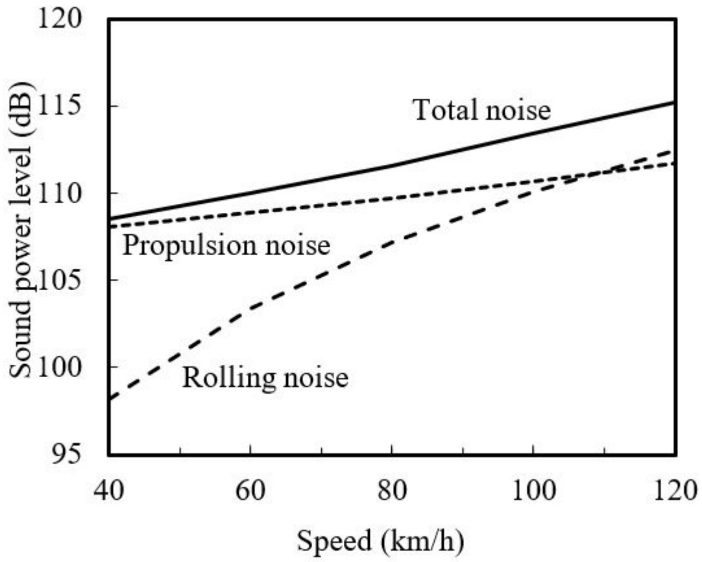

Figure 4.3: An example of sound power level vs vehicle speed. Rolling noise, propulsion noise, and total noise emission of a heavy vehicle were calculated with CNOSSOS-EU at an air temperature of 10°C.

which have lower speed limit, the propulsion noise will have a greater contribution to the total emission.

Figure 4.4 represents the estimated difference of total sound power level for light and heavy ICEVs and EVs calculated at two speed categories of 60 km/h and 100 km/h, in which the sound power level of the EVs was calculated by excluding the propulsion noise. The speed categories were the typical speed limit of trunk roads and expressways, respectively, in Japan.

The sound power level of a light ICEV decreased by 2.6 dB and 0.7 dB at 60 km/h and 100 km/h, respectively, when the propulsion noise was excluded. As for a heavy ICEV, the sound power level decreased by 6.5 dB and 3.3 dB at 60 km/h and 100 km/h, respectively. The sound power level of both light and heavy vehicles showed a marked decrease, particularly at 60 km/h. This reason is shown in Figure 4.2 and Figure 4.3, which indicate that the propulsion noise is dominant at lower speeds. Therefore, when propulsion noise is assumed negligible, the impact of noise reduction will be much higher at lower speeds, particularly for heavy vehicles.

Figure 4.4 also shows that the impact of noise reduction for light vehicles is comparatively smaller than heavy vehicles. Considering the results shown in Figure 4.4, the introduction of higher volumes of heavy EVs will result in an effective noise reduction and presumably, effective health risk reduction. This presumption and the effective traffic conditions for health risk reductions are shown in the following figures.

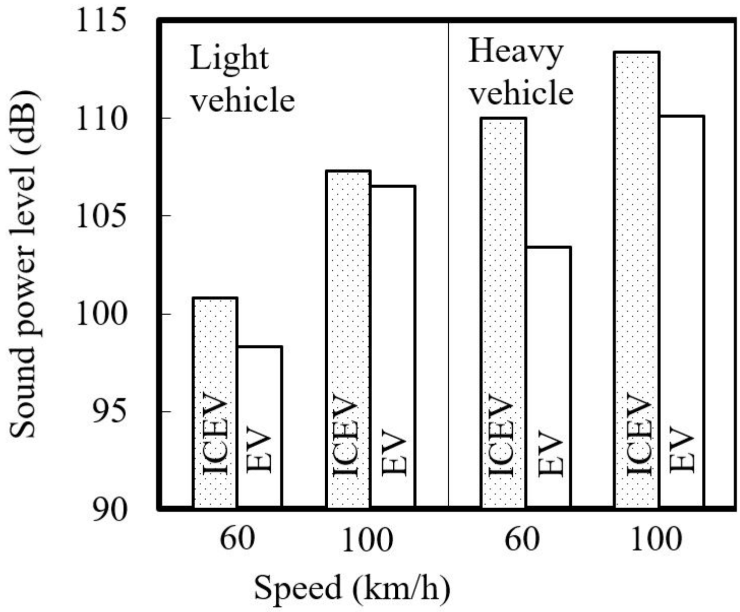

Figure 4.4: Sound power level of a light/heavy ICEV/EV at 60 km/h and 100 km/h with air temperature of 10°C. The sound power level of the EV was calculated by setting the sound power of the propulsion noise to zero.

Figure 4.5 shows the decrease in sound level with traffic condition factors, i.e. percentage of heavy vehicles and traffic speeds when EVs were assumed. The darker regions identify the higher decrease of sound level. The higher the percentage of heavy vehicles, and the lower the traffic speed, the higher the decrease in sound level would be. For example, when the percentage of heavy vehicles is 50%, at 40, 50, and 60 km/h, the decrease in sound level is 9.78, 7.52, and 5.88 dB, respectively. Simultaneously, the health risk reductions would also be higher in these traffic conditions. Therefore, introducing higher percentages of heavy vehicles and regulating to lower traffic speeds will result in a higher sound level decrease.

Figure 4.6 to Figure 4.8 show the relationship between traffic condition, i.e. percentage of heavy vehicles and speeds and health risk reduction for %HSD (percentage of highly sleep-disturbed), %HA (percentage of highly annoyed), and incidence of IHD, respectively. Based on Figure 4.6 (Left), which shows 4.25%HSD at 50 dB L_{night}, the contour reveals a peak risk reduction within the range of -40% – -60% at higher percentages of heavy vehicles and lower speeds. On the other hand, based on Figure 4.6 (Middle) and (Right), the maximum reduction range is within -60% – -80%. The peak risk reduction for %HA and incidence of IHD is -60%– -80% and -80% – -100%, respectively.

The calculation results showed that the higher the percentage of heavy vehicles, and the lower the traffic speed, the more effective the health risk reduction would be. For example,

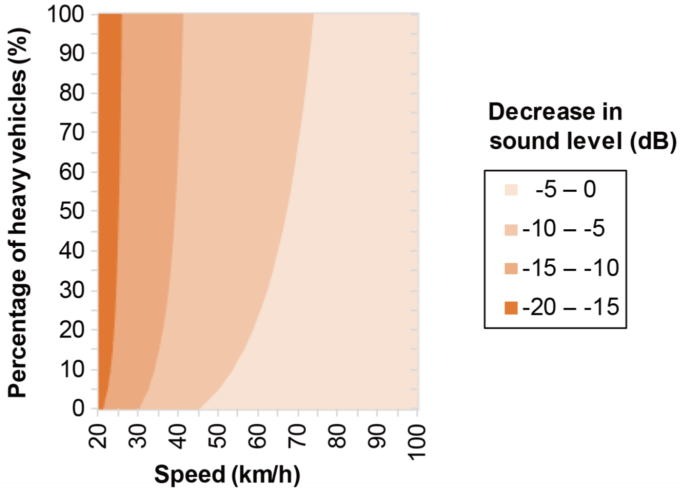

Figure 4.5: Decrease in sound level at different percentages of heavy vehicles and traffic speeds. High sound level reduction is estimated at higher percentages of heavy vehicles and lower traffic speeds.

in Figure 4.7 (Middle), the health risk reduction for %HA at 70 dB L_{den} is 55.0% when traffic speed of 30 km/h and 30% of heavy vehicles in the fleet are assumed; and 66.3% when traffic speed of 20 km/h and 50% of heavy vehicles are assumed.

Figure 4.9, Figure 4.10, and Figure 4.11 respectively show the percentage of highly sleep-disturbed at 40–65 dB L_{night}, percentage of highly annoyed at 40–80 dB L_{den}, and incidence of IHD at 40–80 dB L_{den}. The figures are the estimations of the health risks at 60 km/h traffic speed for ICEV and varying percentages of heavy vehicles for EVs at 0%, 20%, 50%, and 100%.

These figures show that with the transition to EV, the health risk reduction increases with higher percentages of heavy vehicles. Even with 0% of heavy EVs (100% of light EVs) in the traffic flow, there is a marked reduction of health risks. When 20% of heavy EVs are assumed, the health risk reduction doubles that of 0%. There is a slight difference in health risks reduction when 50% and 100% of heavy EVs are assumed.

The same observation could be seen in all figures, which showed that the health risk reduction had the most impact at higher percentages of heavy vehicles and lower speeds. From our calculations, as illustrated in Figure 4.5 to Figure 4.11, the impact of introducing higher percentages of heavy EVs in the vehicle fleet would result in higher decrease in sound level and reduction of health risks. The calculations will be further verified in the actual settings of urban areas in Sapporo City in the following subsections.

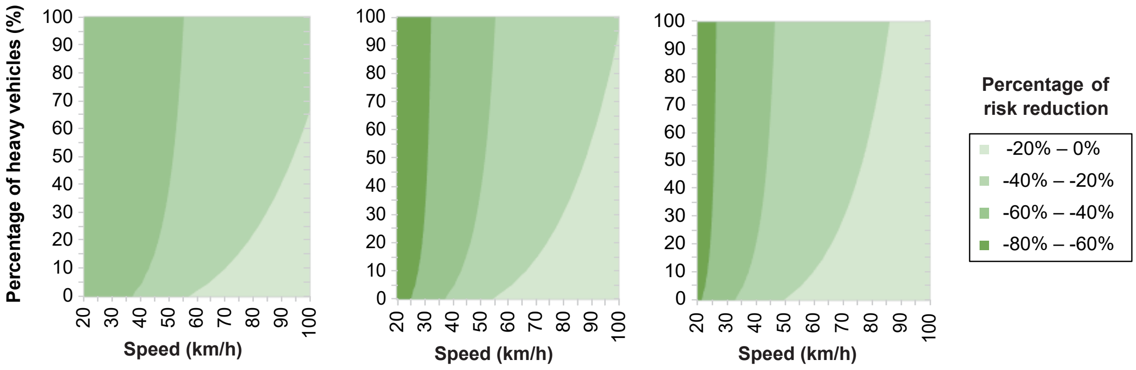

Figure 4.6: Percentage of highly sleep-disturbed (%HSD) reduction (Left) 4.25%HSD at 50 dB L_{night}, (Middle) 8.78%HSD at 60 dB L_{night}, and (Right) 15.82%HSD at 70 dB L_{night} at different percentages of heavy vehicles and traffic speed. High health risk reduction is estimated at higher percentages of heavy vehicles and lower traffic speeds.

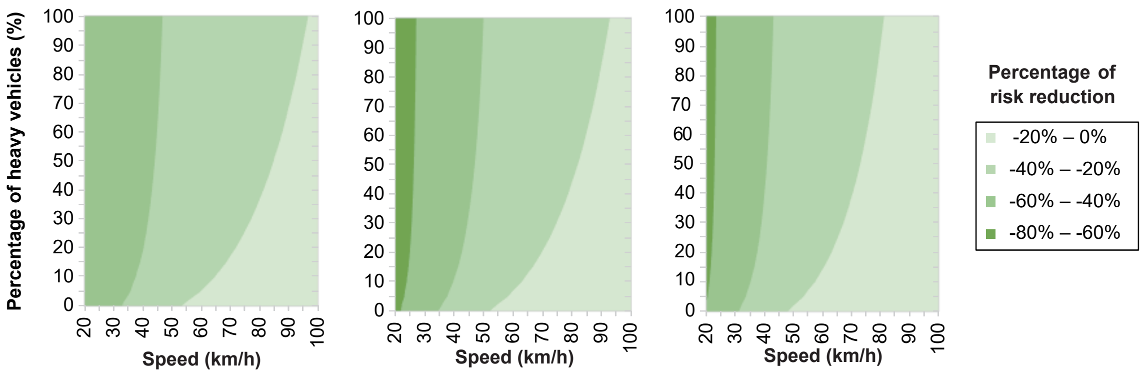

Figure 4.7: Percentage of highly annoyed (%HA) reduction (Left) 15.08%HA at 60 dB L_{den}, (Middle) 28.37%HA at 70 dB L_{den}, and (Right) 48.51%HA at 80 dB L_{den} at different percentages of heavy vehicles and traffic speeds. High health risk reduction is estimated at higher percentages of heavy vehicles and lower traffic speeds.

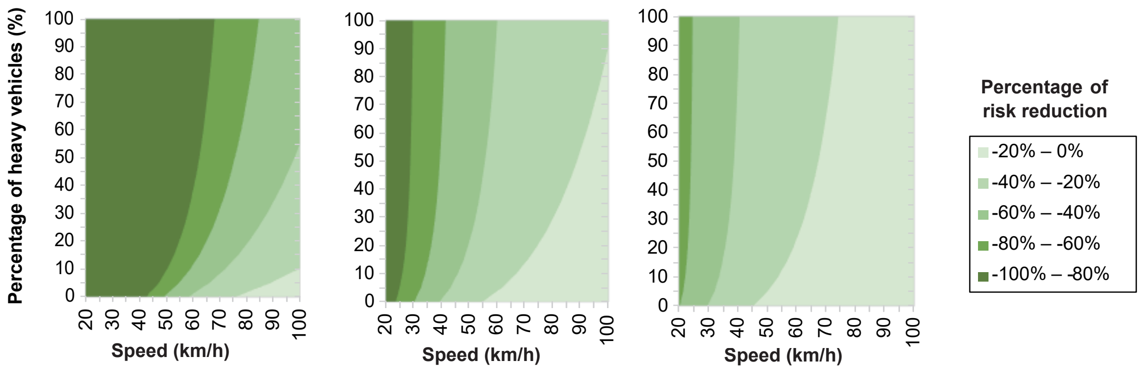

Figure 4.8: Percentage of incidence of ischaemic heart disease (IHD) reduction at (Left) 60 dB L_{den}, (Middle) 70 dB L_{den}, and (Right) 80 dB L_{den} at different percentages of heavy vehicles and traffic speeds. High health risk reduction is estimated at higher percentages of heavy vehicles and lower traffic speeds.

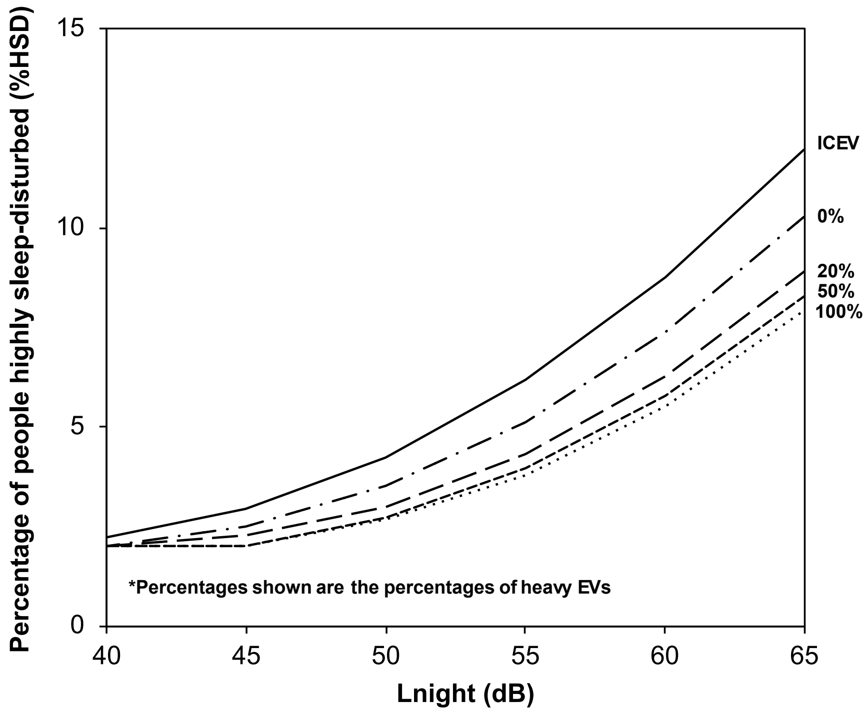

Figure 4.9: Comparison of percentage of highly sleep-disturbed (%HSD) at 60 km/h between ICEV and varying percentages of heavy EV at 40–65 dB L_{night}. The %HSD of EV is estimated at different percentages of heavy vehicles, i.e. 0%, 20%, 50%, and 100%.

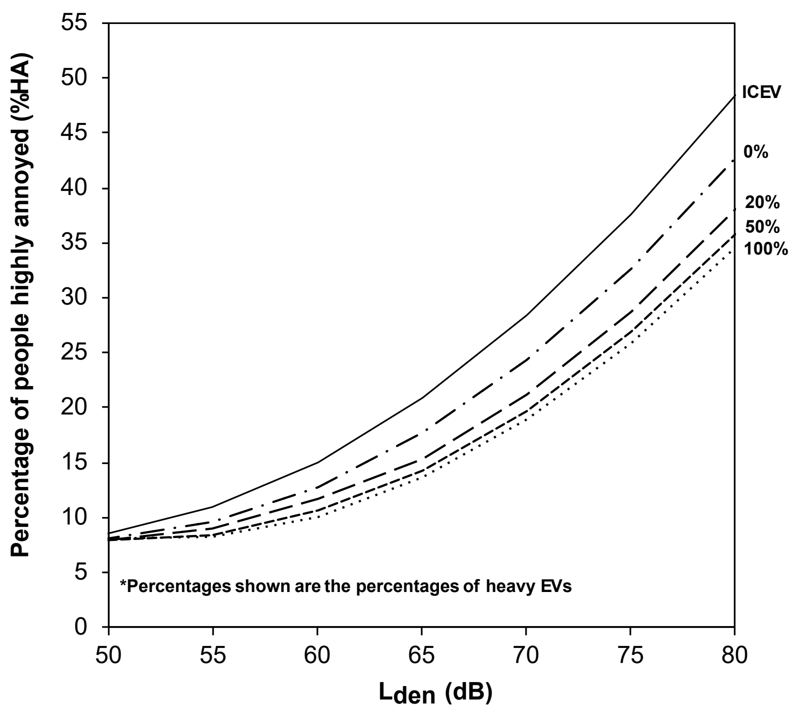

Figure 4.10: Comparison of percentage of highly annoyed (%HA) at 60 km/h between ICEV and varying percentages of heavy EV at 40–80 dB L_{den}. The %HA of EV is estimated at different percentages of heavy vehicles, i.e. 0%, 20%, 50%, and 100%.

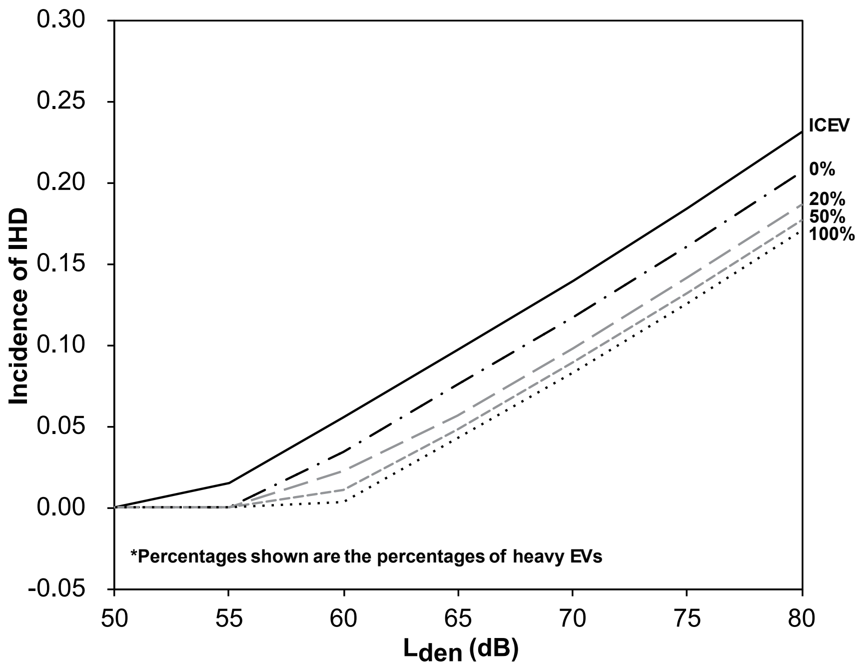

Figure 4.11: Comparison of the incidence of IHD at 60 km/h between ICEV and varying percentages of heavy EV at 40–80 dB L_{den}. The incidence of IHD of EV is estimated at different percentages of heavy vehicles, i.e. 0%, 20%, 50%, and 100%.

4.3.2 Comparison of noise maps and health risk maps by the shift to electric vehicles

Figure 4.12 to Figure 4.15 illustrate the façade noise maps of Area A and Area B in L_{den} and L_{night}, while Figure 4.16 to Figure 4.23 illustrate the health risk maps of Area A and Area B for highly annoyed (%HA), highly sleep-disturbed (%HSD), prevalence and mortality of IHD in 100,000 people. The map on the left represents Situation ICEV where the ICEVs fleet are employed, whereas the map on the right represents Situation EV where EVs fleet are assumed.

In Figure 4.12 to Figure 4.15, the absence of propulsion noise has significantly reduced the sound level in Area A and Area B to approximately 2.5–4.0 dB and 1.5–3.0 dB, respectively. The changes in both L_{den} and L_{night} were clearly more evident in Area A, where the heavy vehicle percentage was high compared to Area B. In Figure 4.16 to Figure 4.23, the transition to EVs has reduced the health risks in both areas.

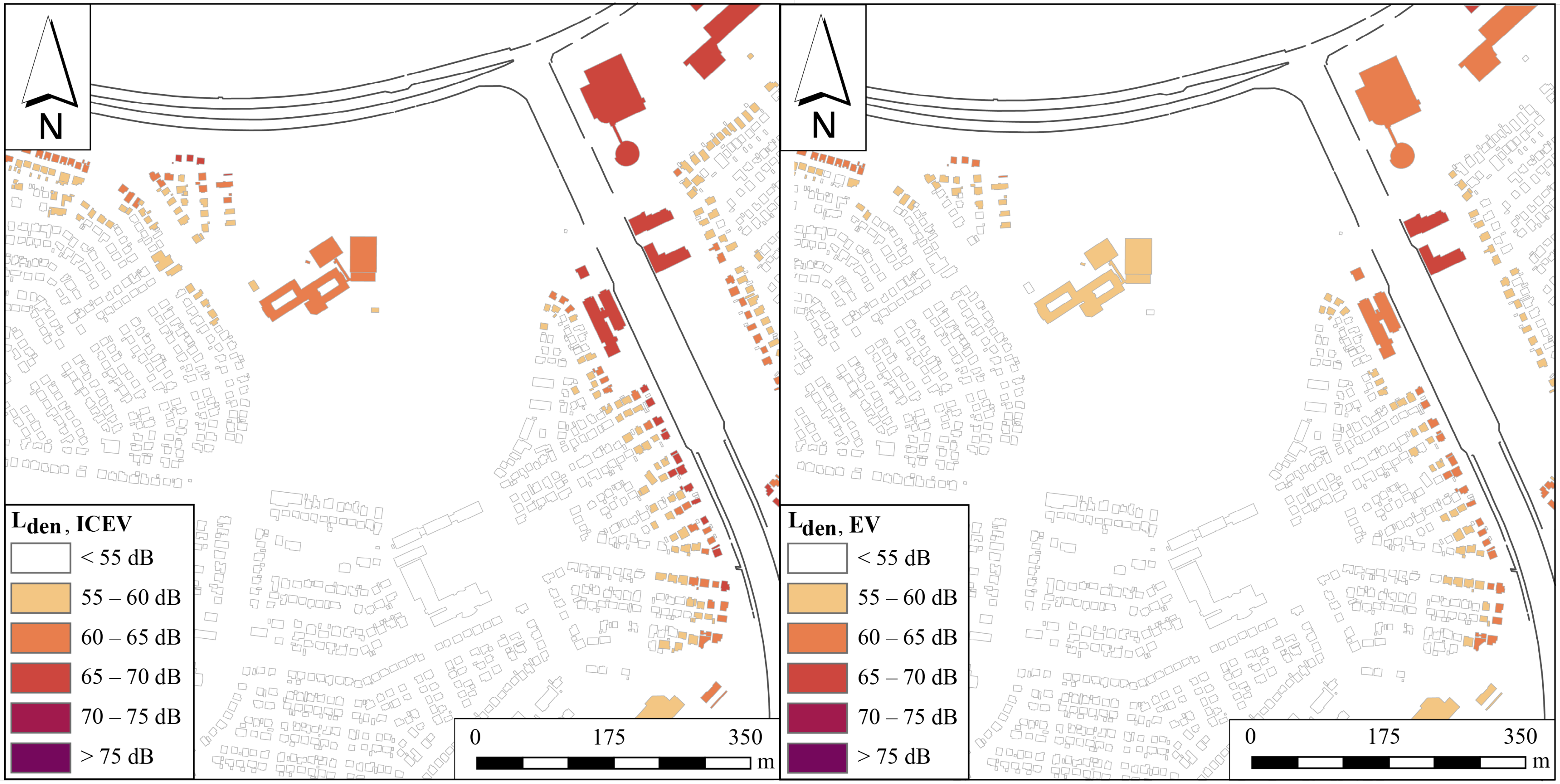

Figure 4.12: Façade noise map in L_{den} in Area A, where the heavy vehicle percentage is relatively high (30–50%). The left and right figures illustrate the situations with ICEVs and EVs, respectively.

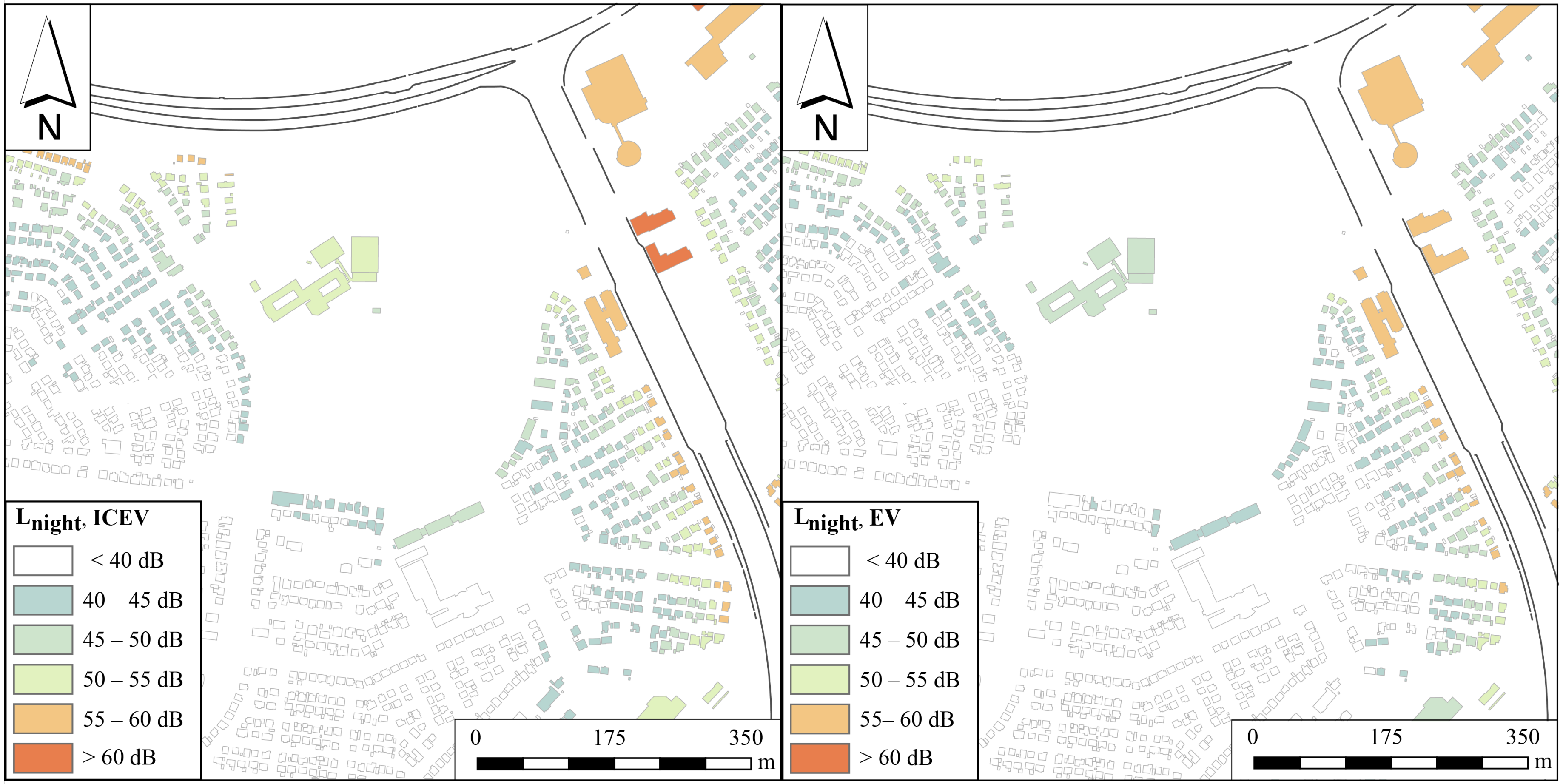

Figure 4.13: Façade noise map in L_{night} in Area A, where the heavy vehicle percentage is relatively high (30–50%). The left and right figures illustrate the situations with ICEVs and EVs, respectively.

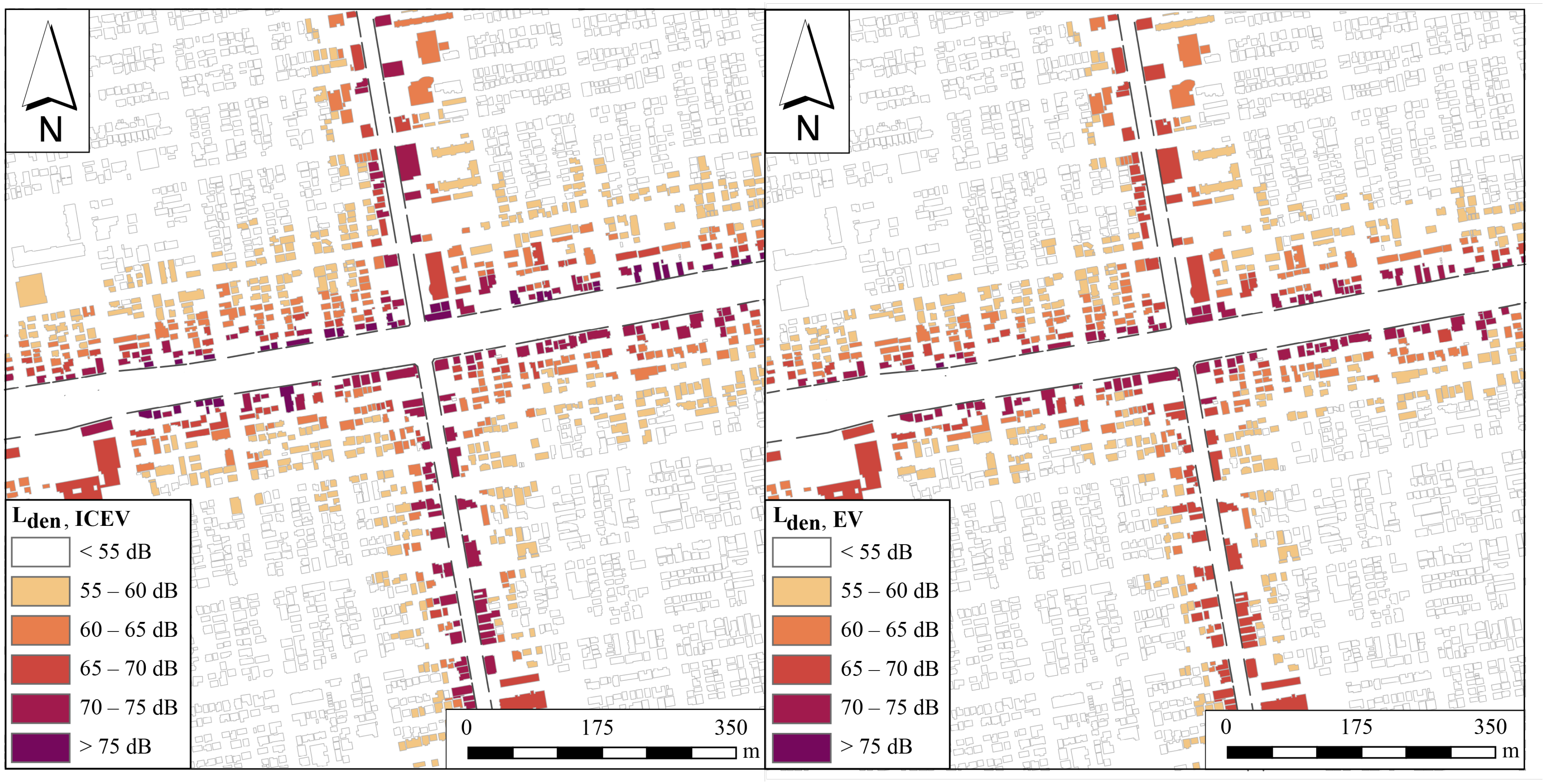

Figure 4.14: Façade noise map in L_{den} in Area B, where the heavy vehicle percentage is relatively low (10–30%). The left and right figures illustrate the situations with ICEVs and EVs, respectively.

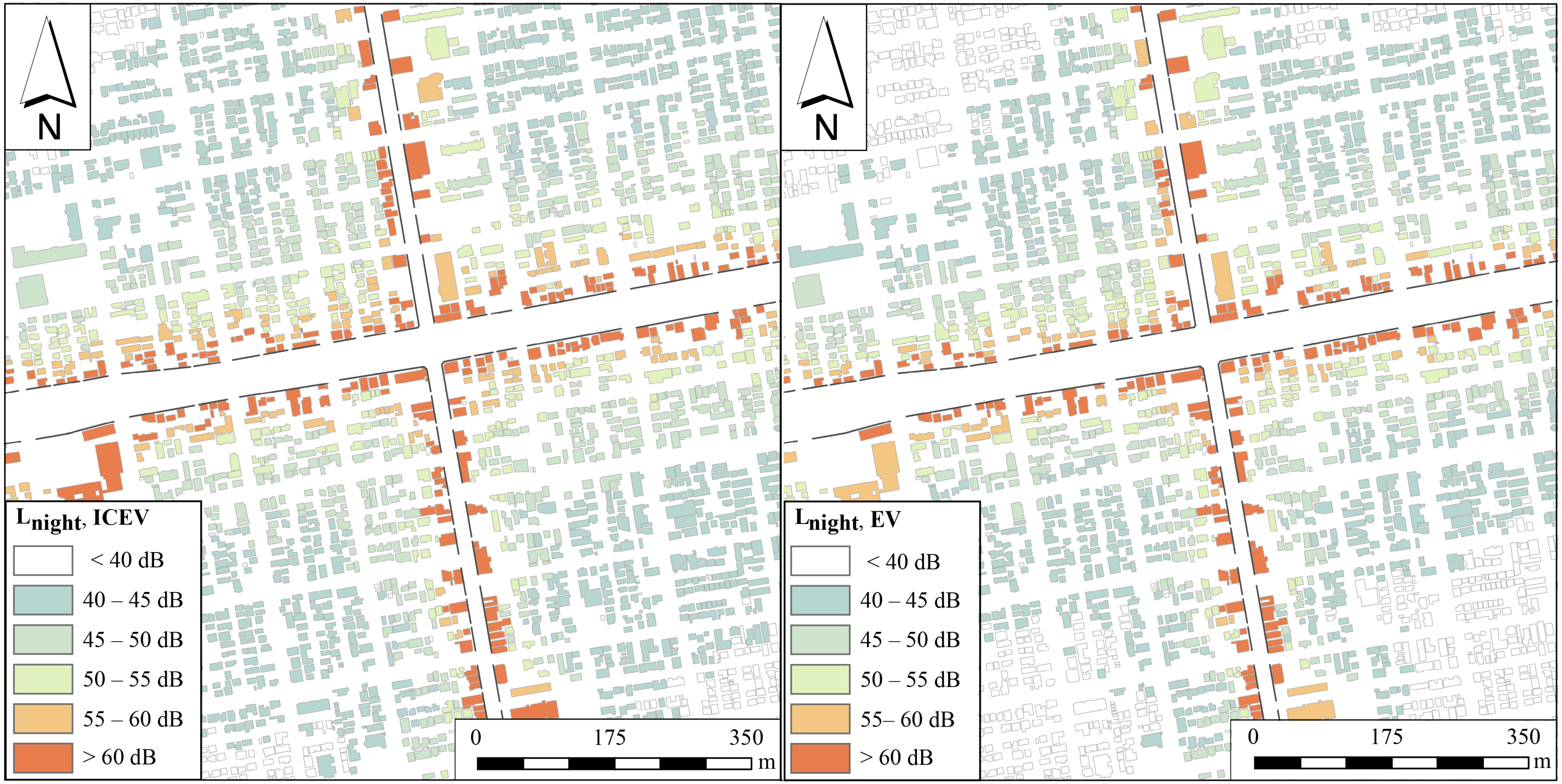

Figure 4.15: Façade noise map in L_{night} in Area B, where the heavy vehicle percentage is relatively low (10–30%). The left and right figures illustrate the situations with ICEVs and EVs, respectively.

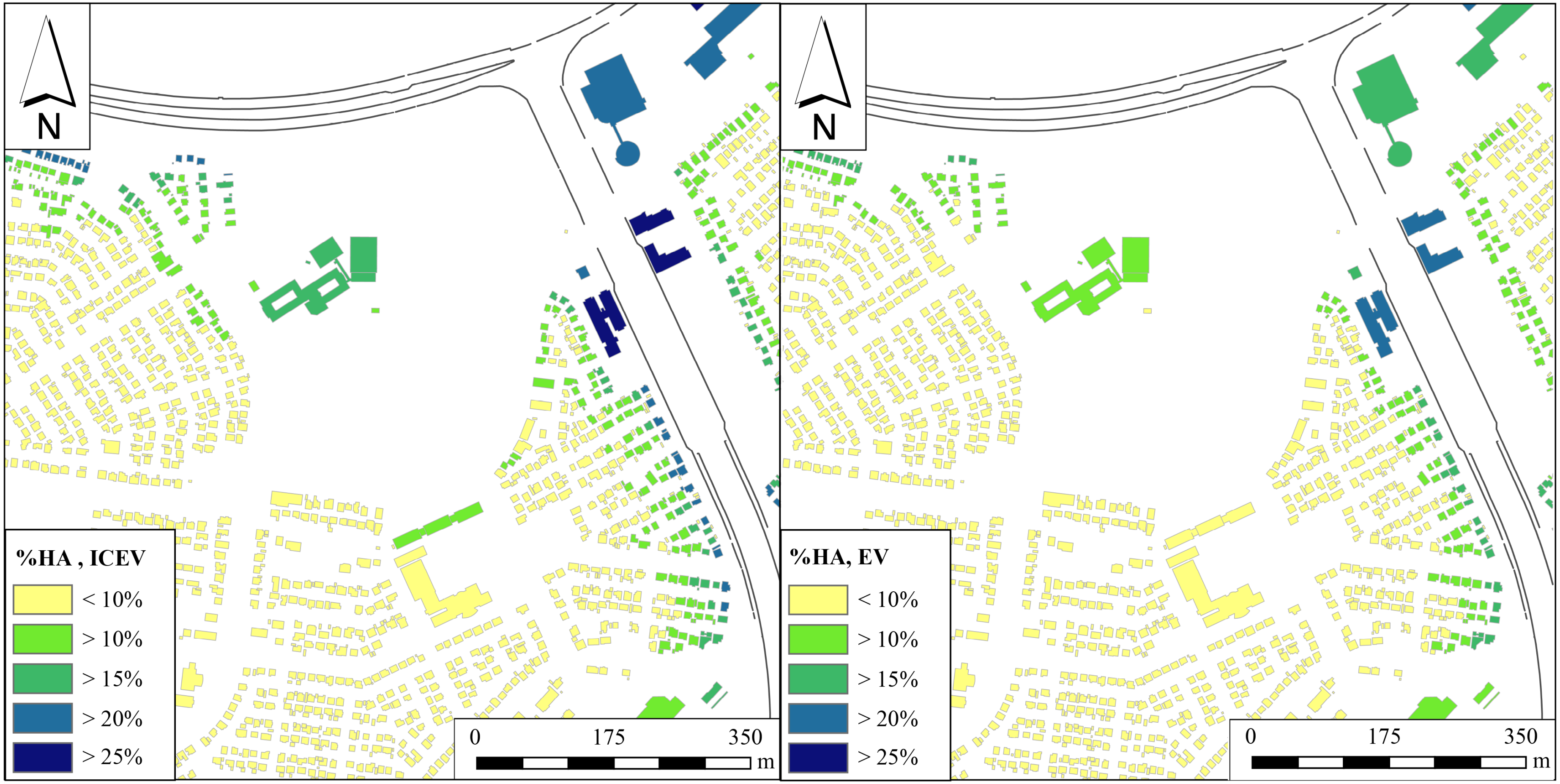

Figure 4.16: Health risk maps for the percentage of people highly annoyed (%HA) in Area A, where the heavy vehicle percentage is 30–50%. The left and right figures illustrate the situations with ICEVs and EVs, respectively.

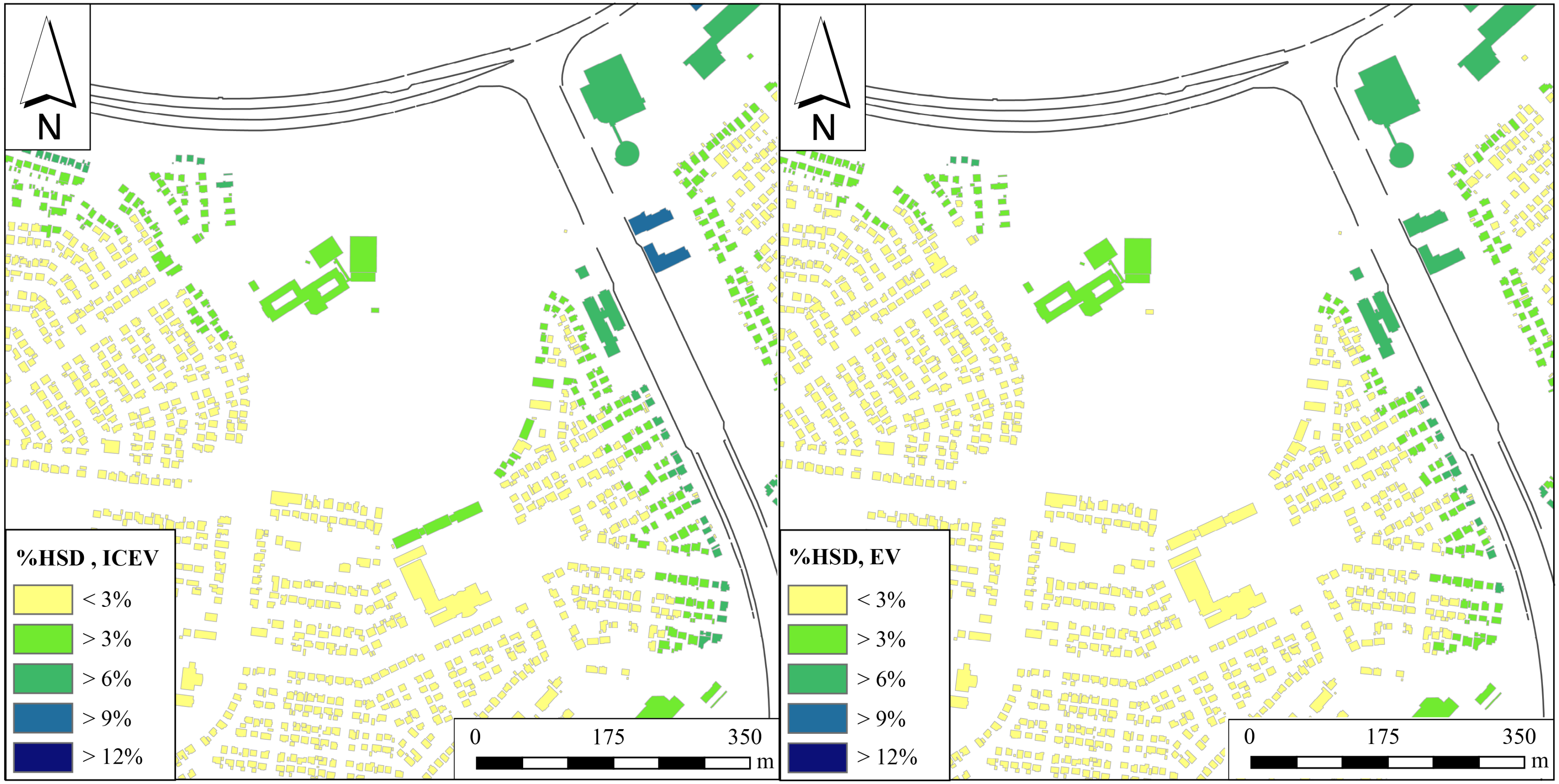

Figure 4.17: Health risk maps for the percentage of people highly sleep-disturbed (%HSD) in Area A, where the heavy vehicle percentage is 30–50%. The left and right figures illustrate the situations with ICEVs and EVs, respectively.

Figure 4.18: Health risk maps for the prevalence rate of ischaemic heart disease (IHD) in Area A, where the heavy vehicle percentage is 30–50%. The left and right figures illustrate the situations with ICEVs and EVs, respectively.

Figure 4.19: Health risk maps for the mortality rate of ischaemic heart disease (IHD) in Area A, where the heavy vehicle percentage is 30–50%. The left and right figures illustrate the situations with ICEVs and EVs, respectively.

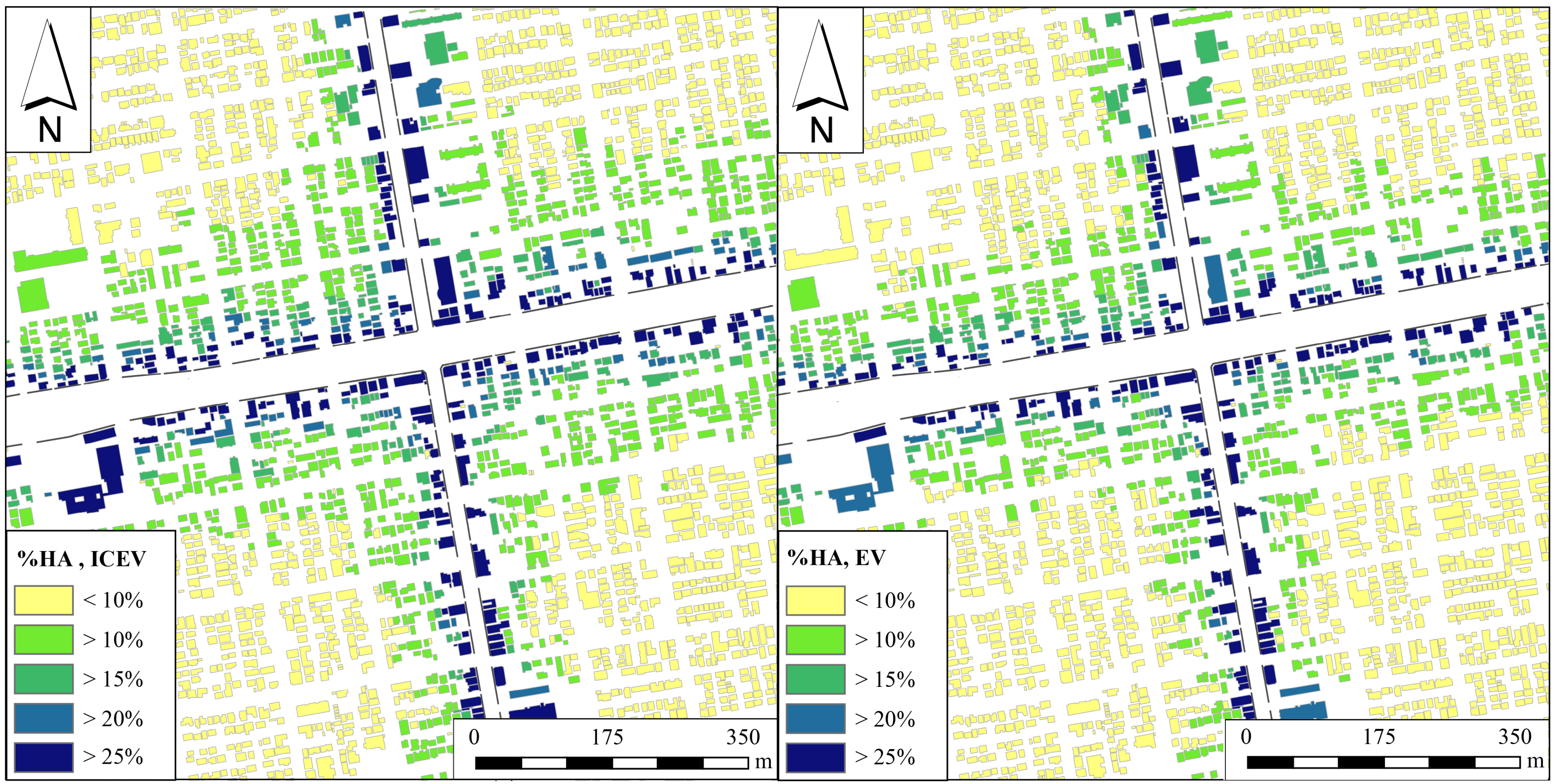

Figure 4.20: Health risk maps for the percentage of people highly annoyed (%HA) in Area B, where the heavy vehicle percentage is 10–30%. The left and right figures illustrate the situations with ICEVs and EVs, respectively.

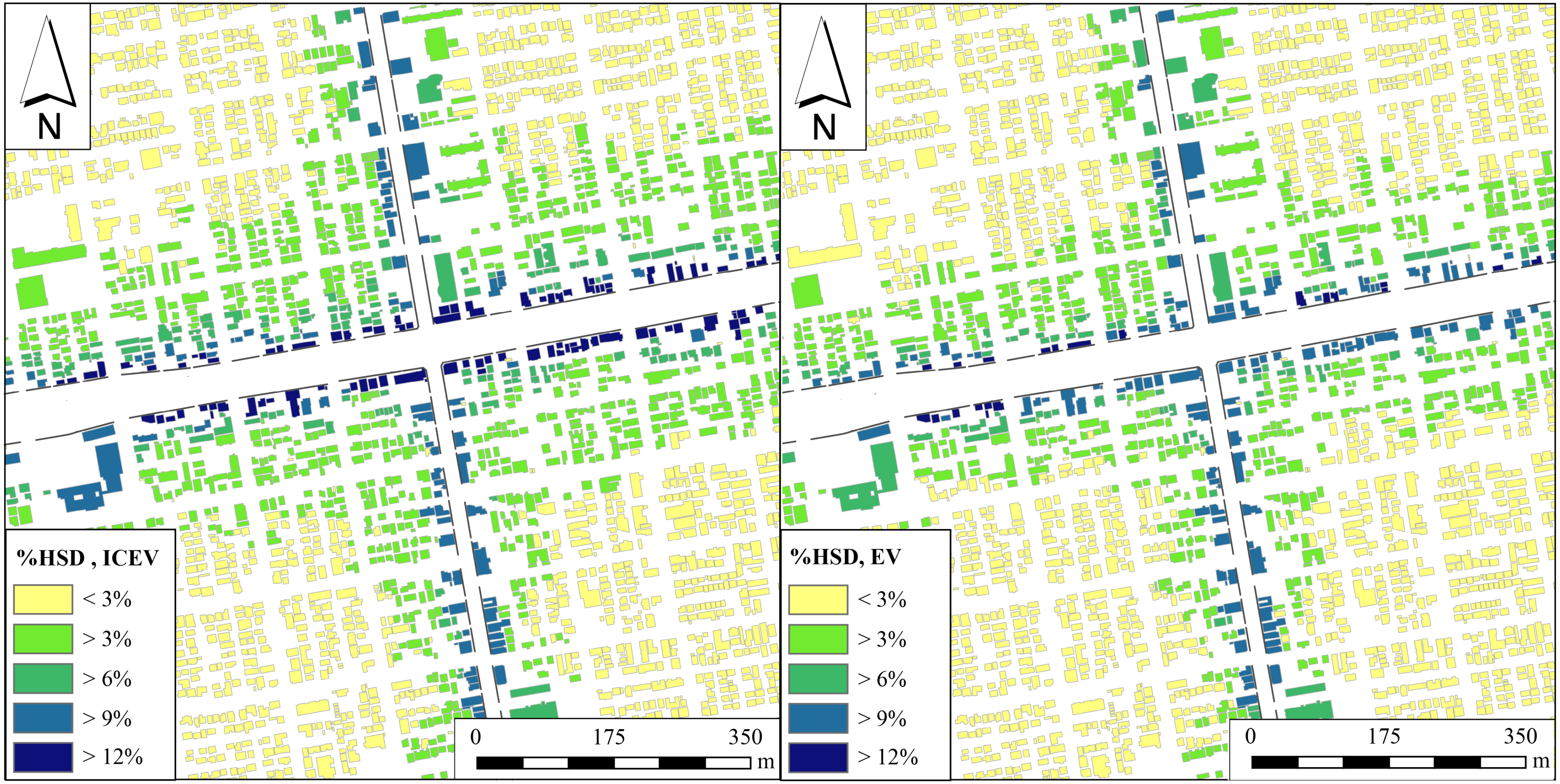

Figure 4.21: Health risk maps for the percentage of people highly sleep-disturbed (%HSD) in Area B, where the heavy vehicle percentage is 10–30%. The left and right figures illustrate the situations with ICEVs and EVs, respectively.

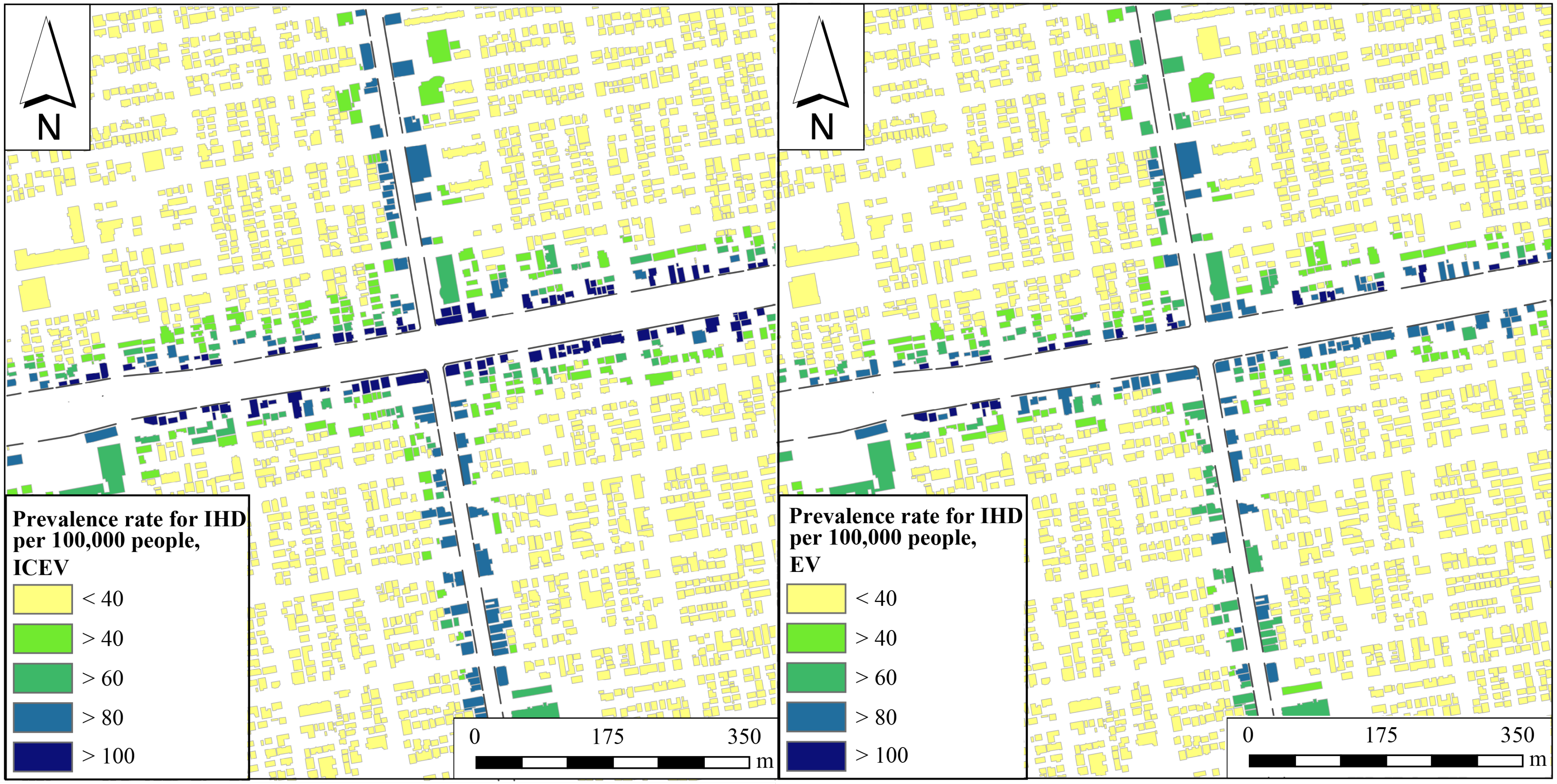

Figure 4.22: Health risk maps for the prevalence rate of ischaemic heart disease (IHD) in Area B, where the heavy vehicle percentage is 10–30%. The left and right figures illustrate the situations with ICEVs and EVs, respectively.

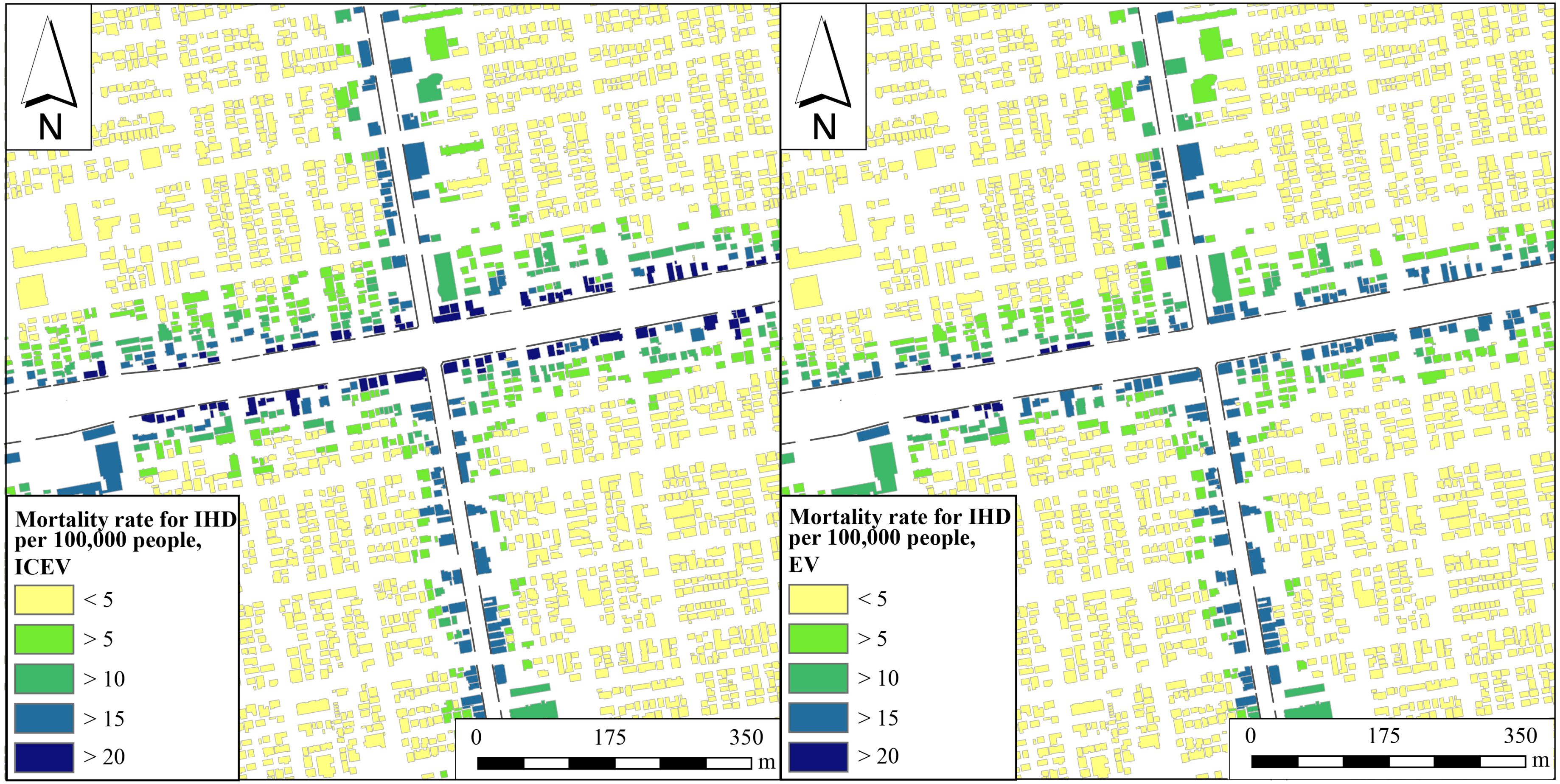

Figure 4.23: Health risk maps for the mortality rate of ischaemic heart disease (IHD) in Area B, where the heavy vehicle percentage is 10–30%. The left and right figures illustrate the situations with ICEVs and EVs, respectively.

4.3.3 Reduction of health risks by the shift to electric vehicles

Table 4.1 to Table 4.4 represent the estimated population exposed to road traffic noise in the two areas stratified by L_{den} and L_{night}. The classification of the population was based on the façade sound levels.

Among the total population in Area A and Area B of 4,341 and 11,933, respectively, the population exposed to excessive road traffic noise level of more than 55 dB in L_{den} were 971 (22.4%) and 4,657 (39.0%) in the situation with ICEVs. During nighttime, respectively, 2,066 (47.6%) and 10,733 (89.9%) residents in Area A and Area B were estimated to be exposed to road traffic noise of more than 40 dB L_{night}. The percentage of exposed population in Area B was relatively higher than Area A due to the higher traffic volumes, travel speeds, and population sizes nearby the roads.

The number of population exposed to > 55 dB L_{den} in Area A and Area B after the transition to EVs was reduced to 580 (13.4%) and 3,700 (31.0%), respectively. A reduction of 40.2% and 20.6% in Area A and Area B, respectively, for noise-exposed population to > 55 dB L_{den} was estimated. For nighttime, the number of exposed population to > 40 dB L_{night} in Area A and Area B after the transition to EVs was reduced to 1,450 (33.4%) and 8,757 (73.4%), respectively. A reduction of 29.8% and 18.4% in Area A and Area B, respectively, for noise-exposed population to > 40 dB $L_{\mathrm{night}}$was estimated.

The transition to EVs significantly reduced the exposed population. For the majority of the categories in Area A, more than half of the population moved to the lower exposed categories. The reduction of the exposed population was relatively smaller in Area B; however, several tens of percent of the population moved to the lower exposed categories.

Table 4.5 to Table 4.12 represent the reduction of the health risks after the transition of all vehicles to EVs. The numbers of IHD patients and deaths, and population suffering from HA and HSD shown in the tables were derived from the exposed population in Table 4.1 to Table 4.4 and the exposure-response relationships established by the WHO-EU (WHO-EU, 2018). The population was stratified using the results of the ICEV situation. I also calculated the prevalence and mortality rates per 100,000 residents, the %HA and the %HSD for the ICEV and EV situations, and obtained the decrease of them. The decrease was derived from the population moved to the lower exposed category for each ICEV category (see Table 4.1 to Table 4.4).

Since the area of the two study areas was limited ($2\times1\ \mathrm{km}^2$) and the total population was about 16,000, the estimate of patients and deaths due to IHD, i.e. around three patients and

Table 4.1: Comparison of the estimated noise-exposed population in Area A in the situations with ICEVs and EVs in L_{den} (dB)

L_{den} (dB),	L_{den} (dB), **ICEV**					
EV	< 55	55–60	60–65	65–70	70–75	> 75
> 75	0	0	0	0	0	0
70–75	0	0	0	0	0	0
65–70	0	0	0	107	21	0
60–65	0	0	51	55	0	0
55–60	0	164	182	0	0	0
< 55	3,370	390	0	0	0	0
Total	3,370	555	233	162	21	0

Table 4.2: Comparison of the estimated noise-exposed population in Area A in the situations with ICEVs and EVs in L_{night} (dB)

L_{night} (dB),	L_{night} (dB), **ICEV**					
EV	< 40	40–45	45–50	50–55	55–60	> 60
> 60	0	0	0	0	0	0
55–60	0	0	0	0	51	128
50–55	0	0	0	88	56	0
45–50	0	0	162	268	0	0
40–45	0	220	477	0	0	0
< 40	2,274	617	0	0	0	0
Total	2,274	837	638	356	107	128

Table 4.3: Comparison of the estimated noise-exposed population in Area B in the situations with ICEVs and EVs in L_{den} (dB)

L_{den} (dB),	L_{den} (dB), **ICEV**					
EV	< 55	55–60	60–65	65–70	70–75	> 75
> 75	0	0	0	0	0	3
70–75	0	0	0	0	250	189
65–70	0	0	0	407	449	0
60–65	0	0	643	203	0	0
55–60	0	1,135	421	0	0	0
< 55	7,275	958	0	0	0	0
Total	7,275	2,092	1,064	610	699	192

Table 4.4: Comparison of the estimated noise-exposed population in Area B in the situations with ICEVs and EVs in L_{night} (dB)

L_{night} (dB),	L_{night} (dB), **ICEV**					
EV	< 40	40–45	45–50	50–55	55–60	> 60
> 60	0	0	0	0	0	913
55–60	0	0	0	0	253	284
50–55	0	0	0	646	423	0
45–50	0	0	1,463	596	0	0
40–45	0	2,961	1,218	0	0	0
< 40	1,199	1,976	0	0	0	0
Total	1,199	4,937	2,681	1,242	675	1,198

one death seemed to be small in the current situation with ICEVs. However, the prevalence and mortality rates were quite high. While the 'yearly' deaths due to IHD were estimated to be 1.55 and 3.58 per 100,000 residents in Area A and B, respectively, the acceptable 'lifetime' risk would be 1 per 100,000 for environmental risk factors (Ministry of the Environment, Japan, 1997). In both study areas, the health risk of IHD due to road traffic noise was quite high as an environmental risk factor.

The population that suffered from HA and HSD (equivalent to mild sleep disorder) in the current situation cannot be neglected even in the small targeted areas, as around 3–7% and 2–3% in total, respectively.

It should be noted that the health risks were unevenly distributed. The impact of road traffic noise-related health risks was considerably higher in areas with high noise exposure. The estimated health risks in the ICEV situation reduced with the shift to EVs. In addition, the prevalence and mortality rates were significantly decreased. This was particularly evident in the higher exposed categories of which the risk of IHD increased with increasing exposure levels. In total, the percentage of the reduction of patients and deaths was 40.9% in Area A and 23.1% in Area B.

With respect to HA, the %HA decreased by 2–10%. Meanwhile, for HSD, the %HSD decreased by 0.5–3%; however, it should be noted that the reduction in the lower exposed group (40–45 dB) would be overestimated because the exposure-response relationship is discontinuous at the threshold. The %HSD was estimated to be zero below the threshold because the calculation method was not provided by the WHO-EU; however, a small proportion of people (less than 2%) should be affected even though noise exposure is below the threshold. Except for the group exposed to 40–45 dB where the risk would be overestimated, the %HSD decreased by 0.5–3% in Area A and 0.5–1.5% in Area B, and the total reduction of the population was estimated to be 22.8% in Area A and 12.0% in Area B.

Table 4.5: Estimated population suffering from high annoyance (HA) in Area A due to road traffic noise.

Current population stratified by L_{den}	L_{den} (dB)	< 55	55–60	60–65	65–70	70–75	> 75	Total
	Exposed population	3,370	555	233	162	21	0	4,341
Current situation with ICEVs	HA population	0	71.1	41.4	39.5	6.8	0	158.8
	%HA	0.00	12.82	17.76	24.41	32.77	–	3.66
After transition to EVs	HA population	0	21.1	32.4	35.8	5.1	0	94.4
	%HA	0.00	3.80	13.90	22.15	24.41	–	2.17
Reduction of the risk	%HA	—	–9.02	–3.86	–2.26	–8.36	—	–1.48

Table 4.6: Estimate of patients with ischaemic heart disease (IHD) in Area A due to road traffic noise.

Current population stratified by L_{den}	L_{den} (dB)	< 55	55–60	60–65	65–70	70–75	> 75	Total
	Exposed population	3,370	555	233	162	21	0	4,341
Current situation with ICEVs	Patients	0.00	0.12	0.10	0.11	0.02	0.00	0.35
	Prevalence rate*	0.00	20.80	44.78	69.70	95.59	0.00	8.12
After transition to EVs	Patients	0.00	0.03	0.06	0.10	0.01	0.00	0.21
	Prevalence rate*	0.00	6.17	26.05	61.22	69.70	0.00	4.80
Reduction of the risk	Prevalence rate*	0.00	–14.64	–18.73	–8.48	–25.89	0.00	–3.32

*Patients per 100,000 residents.

Table 4.7: Estimate of deaths due to ischaemic heart disease (IHD) in Area A due to road traffic noise.

Current population stratified by L_{den}	L_{den} (dB)	< 55	55–60	60–65	65–70	70–75	> 75	Total
	Exposed population	3,370	555	233	162	21	0	4,341
Current situation with ICEVs	Deaths	0.00	0.02	0.02	0.02	0.00	0.00	0.07
	Mortality rate*	0.00	3.96	8.53	13.28	18.22	0.00	1.55
After transition to EVs	Deaths	0.00	0.01	0.01	0.02	0.00	0.00	0.04
	Mortality rate*	0.00	1.18	4.97	11.67	13.28	0.00	0.92
Reduction of the risk	Mortality rate*	0.00	–2.79	–3.57	–1.62	–4.93	0.00	–0.63

*Yearly deaths per 100,000 residents.

Table 4.13 summarises the reduction of the health risks in both areas after the transition to EVs. The difference of the health risk reduction between Area A and Area B was due to the traffic volume of heavy vehicles. As estimated in Figure 4.4, the noise reduction from heavy vehicles was higher than light vehicles. The percentage of heavy vehicles in Area A was twofold higher than that of Area B, and as a consequence, the reduction of health risks was greater compared to Area B.

Table 4.8: Estimated population suffering from high sleep disturbance (HSD) in Area A due to road traffic noise.

Current population stratified by L_{night}	L_{night} (dB)	< 40	40–45	45–50	50–55	55–60	> 60	Total
	Exposed population	2,274	837	638	356	107	128	4,341
Current situation with ICEVs	HSD population	0	21.0	22.4	18.3	7.9	13.1	82.9
	%HSD*	0.00	2.51	3.51	5.15	7.41	10.30	1.91
After transition to EVs	HSD population	0	5.5	17.7	14.0	6.7	9.4	53.3
	%HSD*	0.00	0.66	2.77	3.92	6.22	7.41	1.23
Reduction of the risk	%HSD*	0.00	–1.85	–0.75	–1.23	–1.18	–2.89	–0.68

*Percentage of people highly sleep-disturbed.

Table 4.9: Estimated population suffering from high annoyance in Area B due to road traffic noise.

Current population stratified by L_{den}	L_{den} (dB)	< 55	55–60	60–65	65–70	70–75	> 75	Total
	Exposed population	7,275	2,092	1,064	610	699	192	11,933
Current situation with ICEVs	HA population	0.0	28.2	189.0	148.9	229.1	82.1	917.4
	%HA	0.00	12.82	17.76	24.41	32.77	42.84	7.69
After transition to EVs	HA population	0.0	145.4	168.2	135.4	191.5	63.1	703.7
	%HA	0.00	6.95	15.80	22.20	27.39	32.92	5.90
Reduction of the risk	%HA	—	–5.87	–1.95	–2.21	–5.37	–9.91	–1.79

Table 4.10: Estimate of patients with ischaemic heart disease (IHD) in Area B due to road traffic noise.

Current population stratified by L_{den}	L_{den} (dB)	< 55	55–60	60–65	65–70	70–75	> 75	Total
	Exposed population	7,275	2,092	1,064	610	699	192	11,933
Current situation with ICEVs	Patients	0.00	0.44	0.48	0.43	0.67	0.23	2.24
	Prevalence rate*	0.00	20.80	44.78	69.70	95.59	122.50	18.77
After transition to EVs	Patients	0.00	0.24	0.38	0.37	0.55	0.18	1.72
	Prevalence rate*	0.00	11.28	35.29	61.42	78.94	96.01	14.43
Reduction of the risk	Prevalence rate*	0.00	–9.52	–9.49	–8.28	–16.65	–26.49	–4.34

*Patients per 100,000 residents.

Table 4.11: Estimate of deaths due to ischaemic heart disease (IHD) in Area B due to road traffic noise.

Current population stratified by L_{den}	L_{den} (dB)	< 55	55–60	60–65	65–70	70–75	> 75	Total
	Exposed population	7,275	2,092	1,064	610	699	192	11,933
Current situation with ICEVs	Deaths	0.00	0.08	0.09	0.08	0.13	0.04	0.43
	Mortality rate*	0.00	3.96	8.53	13.28	18.22	23.35	3.58
After transition to EVs	Deaths	0.00	0.04	0.07	0.07	0.11	0.04	0.33
	Mortality rate*	0.00	2.15	6.73	11.70	15.04	18.30	2.75
Reduction of the risk	Mortality rate*	0.00	–1.82	–1.81	–1.58	–3.17	–5.05	–0.83

*Yearly deaths per 100,000 residents.

Table 4.12: Estimated population suffering from high sleep disturbance (HSD) in Area B due to road traffic noise.

Current population stratified by L_{night}	L_{night} (dB)	< 40	40–45	45–50	50–55	55–60	> 60	Total
	Exposed population	1,199	4,937	2,681	1,242	675	1,198	11,933
Current situation with ICEVs	HSD population	0.0	124.0	94.2	63.9	50.0	123.4	455.6
	%HSD*	0.00	2.51	3.51	5.15	7.41	10.30	3.82
After transition to EVs	HSD population	0.0	74.4	82.0	54.2	40.5	115.1	366.2
	%HSD*	0.00	1.51	3.06	4.36	5.99	9.61	3.07
Reduction of the risk	%HSD*	0.00	–1.01	–0.46	–0.78	–1.42	–0.69	–0.75

*Percentage of people highly sleep-disturbed.

Table 4.13: Estimates and reduction of health risks in Situations ICEV and Situation EV in Area A and Area B due to road traffic noise.

Health risks	Area	Situation ICEV	Situation EV	Reduction by the shift to EVs
%HA	A	3.66	2.17	–40.57%
	B	7.69	5.90	–23.29%
%HSD	A	1.91	1.23	–35.72%
	B	3.82	3.07	–19.61%
Prevalence rate of IHD*	A	8.12	4.80	–40.85%
	B	18.77	14.43	–23.12%
Mortality rate of IHD**	A	1.55	0.92	–40.85%
	B	3.58	2.75	–23.12%

*Patients per 100,000 residents.
**Yearly deaths per 100,000 residents.

The findings in Area A were consistent with the calculation results (see Figures 4.6, 4.7, and 4.8), which showed that at 60 km/h and heavy vehicle percentage of 30–50%, 20–40%HA reduction at 60–80 dB L_{den}, 20–40%HSD at 50–70 dB L_{night}, and 20–60% incidence of IHD at 70–80 dB L_{den} was also estimated. Similarly, in Area B, the calculation results showed that at 60 and 100 km/h and heavy vehicle percentage of 10–30%, 0–40%HA reduction at 60–80 dB L_{den}, 0–40%HSD reduction at 50–70 dB L_{night}, and 0–40% incidence of IHD reduction at 70–80 dB L_{den} were estimated.

I observed similar patterns in the percentage of health risk reduction, of which Area A (heavy vehicles percentage of 30–50%) had a much higher reduction percentage than Area B (heavy vehicles percentage of 10–30%). The findings in the verification results in the two areas are consistent with the calculation results, which show that health risk reduction has more impact with higher percentages of heavy vehicles. The area with higher percentages of heavy vehicles contribute to a higher risk reduction (30-40%) compared to the area with lower percentage of heavy vehicles ($\approx 20\%$).

Additionally, the health risk reduction in the total agglomeration of Sapporo City was estimated. The results are shown in Table 4.14 to Table 4.16. The percentages of health risks and population reduced are also shown.

Table 4.14: Estimates of people highly annoyed,%HA, and reduction of %HA in Situation ICEV and Situation EV in Sapporo City.

Exposed population			Situation ICEV		Situation EV		
L_{den} (dB)	Situation ICEV	Situation EV	HA	%HA	HA	%HA	%HA reduced
< 55	1,430,450	1,497,780	0.0	0.0	0.0	0.0	—
55–60	165,981	143,232	21,277	12.8	12,646	7.6	40.6%
60–65	106,271	89,262	18,872	17.8	16,670	15.7	11.7%
65–70	102,782	145,272	25,086	24.4	23,253	22.6	7.3%
70–75	106,750	37,918	34,978	32.8	29,122	27.3	16.7%
> 75	1,306	77	559	42.8	436	33.4	22.1%
Total	1,913,540	1,913,540	100,773	5.3	82,127	4.3	18.5%

Table 4.15: Estimates of people highly sleep-disturbed,%HSD, and reduction of %HSD in Situation ICEV and Situation EV in Sapporo City.

Exposed population			Situation ICEV		Situation EV		
L_{night} (dB)	Situation ICEV	Situation EV	HSD	%HSD	HSD	%HSD	%HSD reduced
< 40	1,003,873	1,176,29	0.0	0.0	0.0	0.0	—
40–45	344,626	258,910	8,657	2.5	4,327	1.3	50.0%
45–50	196,169	162,480	6,893	3.5	6,025	0.4	12.6%
50–55	125,726	107,473	6,470	5.1	5,606	0.7	13.4%
55–60	82,637	101,773	6,122	7.4	5,337	0.9	12.8%
> 60	160,510	106,666	16,532	10.3	14,975	1.0	9.4%
Total	1,913,540	1,913,540	44,674	2.3	36,270	0.4	18.8%

Table 4.16: Estimates and reductions of patients and deaths due to IHD in Situation ICEV and Situation EV in Sapporo City.

Exposed population			Situation ICEV		Situation EV		
L_{den} (dB)	Situation ICEV	Situation EV	Patients	Deaths	Patients	Deaths	Population reduced
< 55	1,430,450	1,497,780	0.0	0.0	0.0	0.0	—
55–60	165,981	143,232	35	7	21	4	40.6%
60–65	106,271	89,262	48	9	37	7	22.5%
65–70	102,782	145,72	72	14	65	12	9.6%
70–75	106,750	37,918	102	19	84	16	17.8%
> 75	1,306	77	2	0	1	0	20.7%
Total	1,913,540	1,913,540	257	49	207	40	19.4%

The estimated percentage of reduction in the agglomeration of Sapporo City was approximately 20% for high annoyance (HA), high sleep disturbance (HSD), and number of patients and deaths of IHD. For the prevalence and mortality rates of IHD, the prevalence rate reduced from 13.4 to 10.8 per 100,000 person and from 2.6 to 2.1 per 100,000 person-year, respectively.

In this study, I employed the CNOSSOS-EU framework to calculate the sound level. The noise emission and propagation of this model are based on the acoustic theory, experimental results, and observations; therefore, the results for ICEVs will not cause major errors. However, the assumption that the propulsion noise from EVs is negligible should be confirmed by further investigation.

I also assumed the speed of all vehicles is 60 km/h in the ordinal trunk roads and 100 km/h in anthe expressway. Actually, according to the Japanese law, heavy trucks are limited to 80 km/h in the expressway. If the speed of heavy vehicles is lower than the calculated value, the effect of reducing health risks will be bigger.

Heavy vehicles have large engines and are thus noisier than light vehicles. The introduction of heavy EVs that eliminate the engine or propulsion noise would significantly reduce the sound power level of a heavy vehicle. The health risks reduction would also be higher when the heavy EVs are introduced in the fleet and is especially evident when the proportion of heavy vehicles is high. In consequence, a higher percentage of heavy vehicles and lower traffic speed would have a rapid and dramatic effect on health risks reduction. This study also showed that there is a significant reduction in health risks even when only light EVs were introduced in the fleet (0% of heavy EVs).

Another limitation of the present study was that my validation was based on data for a Japanese city using the available road traffic information. The health risk reduction results may differ for other areas in Sapporo City, or other cities, depending on the traffic situation. Nevertheless, the main finding, i.e. higher percentages of heavy vehicles and lower traffic speeds would have the greatest impact in reducing health risks will still be valid. This universal concept would help policymakers to consider the effective factors in reducing health risks due to road traffic noise when EVs are introduced in the fleet. For example, regulation of low speeds in urban areas and making policies to support EVs deployment.

Although there is insufficient information on heavy EVs, the reduction of health effects due to road traffic noise will not be negligible because a lot of residents are exposed to the sound level, which can cause high sleep disturbance (mild sleep disorder) at this moment. Even if the decrease of the power level is smaller than my estimation, the total health effects especially on sleep will be mitigated to some extent.

4.3.4 Conclusion

In the present study, the health risk reduction due to road traffic noise by the the shift to EVs were evaluated. The effective traffic conditions are also revealed. The calculations indicated that the health risk reductions were significant when higher proportions of heavy vehicles and lower traffic speeds were assumed. The health risk estimations in the two urban areas also proved that higher percentages of heavy vehicles and lower traffic speeds would contribute to effective health risk reduction.

A significant fraction of the population are at risk of experiencing the adverse health effects of noise; and several tens of percent of reduction in health risks are mitigated with the shift to EVs, particularly in areas with high traffic volume of heavy vehicles and lower traffic speeds. The exposed population and health risks attributed to road traffic noise in Sapporo City and the reduction by the shift to EVs were also estimated. Approximately 20% of health risks reduction was estimated for high annoyance (HA), high sleep disturbance (HSD), and number of patients and deaths due to IHD.

Wide-spreading of EVs is the way forward for a more sustainable transportation alternative to reduce global warming and it will also help mitigate the health effects of road traffic noise to protect the general public. However, transitioning all vehicles to EVs is a long-term goal. Regulating the heavy vehicle speed to a lower speed, especially in nighttime may be effective to mitigate the health risks due to road traffic noise for the time being.

Chapter 5

Conclusion

This work has captured the process in recent years, as environmental noise and health have emerged on the scene and earned a place on the political agenda, especially in Europe. This research took the unprecedented step that will facilitate an 'overlooked' field in Japan and enhance the interdisciplinary nature of environmental research.

The first study focused on the development of health risk maps from noise maps as an alternative tool for public risk communication. The generated health risk maps visualise the distribution of health risks instead of acoustic intensity that is shown in a noise map. Communicating the risk based on ordinary noise maps may lead to misperception of the risks and subsequent inaccurate judgments in those situations by the public. On the contrary, health risk maps directly show the quantitative risks in a graphical format, which can avoid potential misinterpretation of the risks from the noise maps. This would enable the public to comprehend the significance of the health risks due to noise exposure.

With the health risk maps, public trust can be gained owing to the transparent and convenient visualisation of the health risk distribution. Because the health risk maps graphically quantify the risks of each building, this can help the public to comprehend the gravity of the health risks due to noise exposure. Furthermore, the visualisation of health risks through health risk mapping reveals the posing risks to the community. Local residents can realise the risks they are facing by comparing the health risk for mortality rate due to noise-induced ischemic heart disease (IHD) and the acceptable lifetime risk of 1 per 100,000 people (Ministry of the Environment, Japan, 1997).

The health risk maps would contribute to the knowledge sharing with local communities and raising public awareness, hence, can be effective in public health risk communication. By disseminating the health risk maps to the local community, the residents can make informed decisions to protect themselves from the harm of noise, in addition to the measures taken by the government.

The numerical calculations on the health risks in Sapporo City revealed that road traffic noise would threaten the well-being of a significant proportion of people in the city. The overall mortality risk of IHD is higher than the typical acceptable lifetime risk of environmental risk factors of 1 per 100,000 people (Ministry of the Environment, Japan, 1997). The results provide evidence that the total risk of road traffic noise in Sapporo City cannot be left unaddressed and need to be reduced by raising the priority.

In the second study, the health risk reduction by the transition to electric vehicles (EVs) from conventional internal combustion engine vehicles (ICEVs) and the effective traffic conditions were investigated. The introduction of heavy EVs that eliminate the engine or propulsion noise would significantly reduce the sound power level of a heavy vehicle. The health risks reduction would also be higher when the heavy EVs are introduced in the fleet and is especially apparent when the proportion of heavy vehicles is high and at lower traffic speeds.

The calculation results showed that the higher the percentages of heavy vehicles, and the lower the traffic speeds, the more effective the health risk reduction would be. The results in the two areas in Sapporo City with different traffic conditions showed that the area with higher percentages of heavy vehicles and lower traffic speeds contributed to a higher risk reduction compared to the area with opposite traffic conditions. The findings revealed that health risk reductions had more impact with higher percentages of heavy vehicles and lower traffic speeds; and are consistent with the calculation results.

Several tens of percent of health risk reductions were estimated with the shift to EVs in the two areas and in the entire agglomeration of Sapporo City. The study also showed a significant reduction in health risks even when only light EVs were introduced in the fleet (0% of heavy EVs). Therefore, the widespreading of EVs would be a transformative means in mitigating the health effects of the road traffic noise.

For future work, I would like to suggest the following studies:

- Investigating the effectiveness of the health risk maps as an alternative risk communication tool. For example, by measuring the risk perception of the pre and post risk communication of the users with the health risk maps.

- Applying the method to other types of transportation noise, such as railway and aircraft noises, and other health risks.

- Health risk estimations all over Japan to evaluate the risks of environmental noise and enforce mitigation policies.
- Identify additional factors that could further enhance the health risk reduction by the transition to EVs.
- Although EVs are knows as 'quiet' vehicles because they operate without engines and the second study set the propulsion noise as negligible, there is a need to investigate this assumption.

In view of the paucity of such evidence and outcomes, the studies contribute significantly to the body of available evidences. Hopefully, these contributions will hence serve a role in the eventual transition towards achieving zero noise pollution and a healthier living in the future.

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